Public Comment
USFS Waiver
Deadline: 8/24/11 by 12:00 noon

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August 23, 2011

State Water Resources Control Board
c/o Gaylon Lee
Division of Water Quality
1001 I Street, 15th Floor
Sacramento, CA 95814
ForestPlan_Comments@waterboards.ca.gov

Re: Draft Mitigated Negative Declaration and Draft Statewide Conditional Waiver of Waste Discharge Requirements for Nonpoint Source Discharges Related to Certain Activities on National Forest System Lands in California

Dear California State Water Resources Control Board Members:

These comments are submitted on behalf of the Center for Biological Diversity ("Center") regarding the Draft Mitigated Negative Declaration and Draft Statewide Conditional Waiver of Waste Discharge Requirements for Nonpoint Source Discharges Related to Certain Activities on National Forest System Lands in California ("MND") proposed to be adopted by the California State Water Resources Control Board ("SWRCB" or "Board"). These comments are timely filed pursuant to the revised notice issued July 18, 2011 setting noon on August 24, 2011 as the deadline for comments on this matter.

The Center for Biological Diversity ("Center"), a non-profit organization with over 42,000 members, the majority of whom reside in California and is dedicated to protecting imperiled species and their habitats through science, public policy, and the law. The Center represents members of the public who care deeply about water quality in California and the impacts on water quality, wetlands and riparian areas due to activities on the National Forest lands in California.

The Center participated in the stakeholder committee convened by the SWRCB to assist in providing advice to the SWRCB as the Forest Service updated the water quality management plan. As the stakeholder representing aquatic biology, Center staff participated in stakeholder meetings and submitted a comment letter on April 14, 2010, regarding the Forest Service's draft water quality management plan (now re-framed as a handbook). In meetings and in that earlier comment letter, the Center identified several issues of concern which were never addressed in the revised version of the handbook and have not been addressed in any of the documents provided by the SWRCB to date.

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A. Introduction

The Center opposes the SWRCB proposal to approve a waiver with conditions applying to a suite of activities on all National Forest lands within the state of California because the waiver will not protect water quality and the SWRCB has failed to undertake the needed environmental review and instead is attempting to rely on a mitigated negative declaration ("MND") for a project that not only may, but most certainly will, have a significant effect on the environment. By relying on a MND rather than the legally required environmental impact report ("EIR"), the SWRCB action would undermine both the letter and the spirit of the California Environmental Quality Act ("CEQA") because the MND fails to provide any meaningful analysis of significant impacts to water quality and aquatic and riparian resources, fails to address cumulative impacts, and fails to provide any analysis of alternatives that could avoid the significant impacts of the covered activities. Moreover, the SWRCB has failed to show that the alleged mitigation proposed in the MND, which will be outlined in a new Forest Service *handbook* rather than a planning document, is sufficient to reduce the impacts of the many covered activities below a level of significance. The handbook itself is permeated with discretionary language and thus many of the BMPs and other measures are uncertain to occur, and there is no provision for funding for the proffered mitigation measures (even if they were shown to be sufficient).

B. Legal Background.

<u>California Water Code and the Federal Clean Water Act</u>

California Water Code section 13260 requires any person discharging waste or proposing to do so which could affect the quality of the waters of the State to file with the Water Board a report of waste discharge. Water Code section 13269 in turn authorizes the Board to waive a waste discharge requirement if a waiver "is consistent with any applicable state or regional water quality control plan and is in the public interest." A waiver cannot exceed five years but can be renewed. *Id*. Section 13269 was amended in 2005 to add monitoring requirements. Consequently, all waivers must include monitoring that is "designed to support the development and implementation of the waiver program, including but not limited to verifying the adequacy and effectivness of the waiver's conditions." Waivers may be conditioned by the water board.

Under both state and federal law, water quality in California is protected by what is referred to as "antidegradation" policies. The state policy is titled the *Statement of Policy with Respect to Maintaining High Quality Waters in California*, codified in 23 CCR § 2900, and is commonly known as "Resolution 68-16." The federal antidegradation policy is found at 40 CFR §131.12. These state and federal policies are independently enforceable and both require that where surface waters are of higher quality than necessary to protect the designated beneficial uses, the high quality of the waters must be maintained unless otherwise provided by the policies.

The state antidegradation policy applies to groundwater and surface water whose quality meets or exceeds water quality objectives. The federal antidegradation policy applies to surface water regardless of the quality of the water. The state policy also establishes that any activities that result in discharge to high quality waters are required to use the best practicable treatment or control necessary. The state antidegradation policy further establishes that if the discharge, even after treatment, unreasonably affects beneficial uses or does not comply with applicable provisions of Basin Plans, the discharge must be prohibited.

Both the state and federal antidegradation policies acknowledge that minor or repeated activities, even if individually small, can result in violation of antidegradation policies through cumulative effects, especially, for example, when the waste is a cumulative, persistent, or bioaccumulative pollutant.

California Environmental Quality Act

The Legislature enacted CEQA to "[e]nsure that the long-term protection of the environment shall be the guiding criterion in public decisions." *No Oil, Inc. v. City of Los Angeles*, 13 Cal. 3d 68, 74 (1974). The Supreme Court has repeatedly held that CEQA must be interpreted to "afford the fullest possible protection to the environment." *Wildlife Alive v. Chickering*, 18 Cal. 3d 190, 206 (1976) (quotation omitted). CEQA also serves "to demonstrate to an apprehensive citizenry that the agency has, in fact, analyzed and considered the ecological implications of its action." *Laurel Heights Improvement Ass'n v. Regents of Univ. of Cal*. (1988) 47 Cal. 3d 376, 392 ("*Laurel Heights I*"). If CEQA is "scrupulously followed," the public will know the basis for the agency's action and "being duly informed, can respond accordingly to action with which it disagrees." *Id.* Thus, CEQA "protects not only the environment but also informed self-government." *Id.*

CEQA applies to all "discretionary projects proposed to be carried out or approved by public agencies." Pub. Res. Code § 21080(a). CEQA defines a "project" as an action by a public agency that has the potential to cause a physical change in the environment. Guidelines § 15378. CEQA applies to "discretionary projects proposed to be carried out or approved by public agencies, including, but not limited to, the enactment and amendment of zoning ordinances..." Cal. Pub. Res. Code, § 21080(a). The definition of "project" includes the adoption, amendment or revision of a general plan. Guidelines § 15378(a)(1); *Sierra Club v. County of Sonoma* (1992) 6 Cal. App. 4th 1307; *City of Redlands v. County of San Bernardino* (2002) 96 Cal. App. 4th 398

A "project" is "the whole of an action" directly undertaken, supported, or authorized by a public agency "which may cause either a direct physical change in the environment, or a reasonably foreseeable indirect physical change in the environment." Pub. Res. Code § 21065; CEQA Guidelines § 15378(a). Under CEQA, "the term 'project' refers to the underlying activity and not the governmental approval process." *California Unions for Reliable Energy v. Mojave Desert Air Quality Mgmt. Dist*. (2009) 178 Cal. App. 4th 1225, 1241 (*quoting Orinda Ass'n v. Bd. of Supervisors* (1986) 182 Cal. App. 3d 1145, 1171-72). The definition of "project" is "given a broad interpretation in order to maximize protection of the environment." *Lighthouse Field Beach Rescue v. City of Santa Cruz* (2005) 131 Cal. App. 4th 1170, 1180 (internal quotation omitted).

Where, as here, there is a fair argument that the proposed project may have a significant effect on the environment, preparation of an EIR is required. Public Resources Code §§ 21100, 21151; CEQA Guidelines § 15064(a)(1); *No Oil, Inc. v. City of Los Angeles* (1974) 13 Cal. 3d 68, 82. "The fair argument standard is a 'low threshold' test for requiring the preparation of an EIR. . . . It is a question of law, not fact, whether a fair argument exists, and the courts owe no deference to the lead agency's determination. Review is de novo, with a preference for resolving doubts in favor of environmental review." *Pocket Protectors v. City of Sacramento*, 124 Cal.App.4th at 926-28.

CEQA defines a "significant effect" as a "substantial, or potentially substantial, adverse change." Public Resources Code § 21068. An action has a significant effect if it "has the potential to degrage the quality of the environment." *Azuza land Reclamation Company, Inc. v. Main San Gabriel Basin Watermaster* (1997) 52 Cal.App.4th 1165, 1192. The CEQA guidelines require a mandatory finding of significance for a project with "possible environmental effects which are individually limited but cumulatively considerable." "Cumulatively considerable" means that the incremental effects of a project are considerable when viewed in connection with the effects of past projects, the effects of other current projects, and the effects of probable future projects." CEQA Guidelines § 15065.

Before reaching a decision on the project, the lead agency must consider any comments submitted. Pub. Resources Code, § 21091, subd. (d)(1); CEQA Guidelines, § 15074 (b)). Under CEQA, an EIR must be prepared even if the lead agency can point to substantial evidence in the record supporting its determination that no significant effect will occur. *Architectural Heritage Assn. v. County of Monterey* (2004) 122 Cal. App. 4th 1095, 1110. The lead agency may not dismiss evidence because it believes that there is contrary evidence that is more credible. *Pocket Protectors v. City of Sacramento* (2005) 124 Cal. App. 4th 903, 935. Either there is substantial evidence showing the possibility of a significant environmental effect or there is not. If there is, then the lead agency must prepare an EIR. *Architectural Heritage Assn*., 122 Cal. App. 4th at 1109-1110. Importantly, the "fair argument" test "establishes a low threshold for initial preparation of an EIR, which reflects a preference for resolving doubts in favor of environmental review." *Id.* at 1110. Notably, even where the ultimate goal of an action may be to improve the environment, an EIR may be needed. CEQA Guidelines § 15063 (b)(1) (where a project may cause significant effect on the environment "regardless of whether the overall effect of the project is adverse or beneficial" the agency shall prepare an EIR).

By contrast, mitigated negative declarations are appropriate only when there is no substantial evidence in light of the whole record before the public agency that the project, as revised, may have a significant effect on the environment. Pub. Resources Code, § 21064.5; see also § 21080, subd. (c); CEQA Guidelines §§ 15006, subd. (h), 15064, subd. (f)(2), 15070, subd. (b), 15369.5. Likewise, a MND is no good if it does not ensure that measures designed to mitigate impacts are "fully enforceable through permit conditions, agreements, or other measures." (Pub. Resources Code § 21081.6, subd. (b).) "The purpose of these requirements is to ensure that feasible mitigation measures will actually be implemented as a condition of development, and not merely adopted and then neglected or disregarded." (*Fed'n of Hillside & Canyon Ass'ns v. City of Los Angeles* (2000) 83 Cal.App.4th 1252, 1261.) Moreover, there must exist “a monitoring program to ensure that the mitigation measures are implemented.” *Federation of Hillside & Canyon Ass’ns v. City of Los Angeles,* Cal.App.4th at 1261.

Further, CEQA requires the preparation of environmental review documents “as early as feasible in the planning process to enable environmental considerations to influence project program and design and yet late enough to provide meaningful information for environmental assessment.” *Laurel Heights I*, 47 Cal.3d at 395; see also CEQA Guidelines § 15004(b). The purpose of CEQA is to provide decision-makers and the public with environmental information before decisions are made, not after. As the California Supreme Court observed in *Laurel Heights I*, “[i]f post-approval environmental review were allowed, [CEQA analyses] would likely become nothing more than post hoc rationalizations to support action already taken. We have expressly condemned this [practice].” 47 Cal. 3d at 394 (citation omitted). Accordingly, “public agencies shall not undertake actions concerning the proposed public project that would have a significant adverse effect or limit the choice of alternatives or mitigation measures, before completion of CEQA compliance.” CEQA Guidelines § 15004(b)(2). In particular, an agency shall not “take any action which gives impetus to a planned or foreseeable project in a manner that forecloses alternatives or mitigation measures that would ordinarily be part of CEQA review of that public project.” CEQA Guidelines § 15004(b)(2)(B).

CEQA makes clear that one of its fundamental purposes is to “provide public agencies and the public in general with detailed information about the effect which a proposed project is likely to have on the environment.” Public Resources Code § 21061. A failure to include adequate information in the review process is therefore reviewed as a matter of law, without deference to the agency. *See Vineyard Area Citizens for responsible Growth, Inc. v. City of Rancho Cordova* (2007) 40 cal.4th 412, 435. Under CEQA, the agency must “use its best efforts to find out and disclose all that it reasonable can.” *San Franciscans for Reasonable Growth v. City and County of San Francisco* (1984) 151 Cal.App.3d 61, 74.

C. The Proposed Project May Significantly Effect the Environment and An EIR is Required

Before authorizing an activity that will cause reasonably foreseeable significant direct and indirect physical changes in the environment of this magnitude the Board must prepare an EIR. Many of the activities covered by the proposed statewide waiver will impact sensitive resources that may be significantly adversely affected by the activities including but not limited to threatened and endangered species and other imperiled species and habitats.

Below is a summary of just a few of the significant effects of livestock grazing, logging, and ORV use in the National Forests on California’s riparian and aquatic ecosystems and species.

1. Off-road vehicle effects on water quality and aquatic and riparian species and habitats

Off-road vehicle (ORV) use negatively affects many different ecosystems in California within the National Forests and downstream from the forests, including the California Desert, the

Sierra Nevada, coastal areas, and the Great Basin (Shore 2001). Impacts include degraded water quality and harm to native biodiversity. Because so many species are dependent upon water bodies and riparian zones and these habitats are scarce in many parts of California, these fragile ecosystems can be particularly damaged by ORV use (Sarr et al. 2005). ORV use in aquatic ecosystems such as wetlands, bogs and swamps can cause ruts to form and change natural hydrological patterns (Kassar 2009).

ORVs cause soil compaction, destroy soil crusts, and erode soils, which in turn limit the soil's ability to support vegetation (Ouren et al. 2007; Adams 1982). The loss of vegetative cover results in decreased water filtration into water bodies and increased precipitation runoff. This runoff causes even more erosion and leads to higher levels of sedimentation, turbidity, and pollutants in water bodies. Pollutants enter aquatic systems from deposition of ORV emissions and gasoline spills.

ORVs cause numerous impacts on riparian wildlife, including physiological damage and behavioral changes from excessive noise, habitat perturbations and destruction, increased stress and mortality, direct mortality from collisions, and altered behavior and dispersal patterns (Ouren et al. 2007). These effects can lead to negative impacts on local population size, productivity, and survivorship. Soil disturbance from roads and trials and the ongoing and increasing use of those roads and trails could cause significant impacts to soil surfaces and result in increased siltation and hydrocarbons entering water ways and further impairing water quality. Snowmobile use can similarly lead to deposition of hydrocarbons and other chemicals on snow which eventually enter water ways and impair water quality. [1]

Some references documenting riparian ecosystem impacts from ORV use:

- Adams, J.A., Endo, A.S., Stolzy, L.H., Rowlands, P.G., and Johnson, H.B. (1982). Controlled experiments on soil compaction produced by off-road vehicles in the Mojave Desert, California
- Barton, D. C., & Holmes, A. L. (2007). Off-Highway Vehicle Trail Impacts on Breeding Songbirds in Northeastern California
- Kassar, C., (2009). Environmental impacts of ORVs on the Rubicon Trail.
- Ouren, B. D. S., Haas, C., & Melcher, C. P. (2007). Environmental Effects of Off-Highway Vehicles on Bureau of Land Management Lands
- Shore, T. (2001). Off-Road to Ruin.
- Welsh, H. H., & Ollivier, L. M. (1998). Stream Amphibians As Indicators of Ecosystem Stress:a Case Study From California's Redwoods.
- Wilshire, H.G., Nakata, J.K., Shipley, Susan, and Prestegaard, Karen. (1978). Impacts of vehicles on natural terrain at seven sites in the San Francisco Bay area
- Arnold, J. and T. Koel. 2006. Effects of Snowmobile Emissions on the Chemistry of Snowmelt Runoff in Yellowstone National Park. Final Report. Fisheries and Aquatic Sciences Section. Center for Resources. Yellowstone National Park. Wyoming. YCR-2006-1.

[1] The Center incorporates by reference the comments filed by Snowlands Network and Winter Wildlands Alliance on August 16, 2011 and the attachments provided regarding the impacts of snowmobile use on water quality.

- Adams, S.E. "Effects of Lead and Hydrocarbons From Snowmobile Exhaust on Brook Trout (Salvalinus fontinalis)," Transactions of the American Fisheries Society. Vol. 104, No. 2 (1974), pp. 363-373. (abstract)

Some species affected by or at risk from ORV use include but are not limited to:

- Amphibians: negative impacts from sedimentation (Welsh and Ollivier 1998)
- Birds dependent upon riparian areas: abandon nests close to ORV trails (Barton and Holmes 2007)
- Aquatic species: see references below regarding sedimentation from logging, and grazing

2. Livestock Grazing and Range Management Activities effects on riparian vegetation, water quality, and riparian and aquatic resources

Ecological costs of livestock grazing in the Western United States include biodiversity loss; lowered population densities for many species; ecosystem function disruption, including nutrient cycling and succession; altered community organization; and altered habitat physical characteristics (Fleischner 1994). These costs are magnified in riparian ecosystems, which are often the most biologically rich in arid and semiarid regions, as livestock congregate in these habitats (Fleischner 1994). Livestock congregate near riparian areas for water, succulent forage, and shade (Belsky et al. 1999). In the Western United States, livestock grazing has damaged ~80% of riparian and stream ecosystems (Belsky et al. 1999). Livestock congregating in riparian areas causes trampling and overgrazing of streambanks, soil erosion and sedimentation in streams, streambank instability, negative impacts on water quality, aridity, and increased temperatures. These impacts have decreased habitat for riparian plants, cold-water fish, and wildlife, causing declines or local extinction of many native species. These changes, in turn, can cause large-scale impacts in adjacent and downstream ecosystems.

Numerous studies have documented negative impacts of grazing on California's riparian ecosystems. These impacts have included effects on riparian vegetation, water quality, and riparian biodiversity, among others.

In California's oak savanna–annual grassland, annual total herbaceous cover in springs and creeks was negative over time under moderate grazing as compared to light or no grazing (Jackson and Allen-Diaz 2006). There were also negative effects of grazing on vegetation structure in wet meadows in Sequoia National Park (Holmquist et al. 2010). In California's foothills, researchers have found that the concentration of cattle along stream banks during the dry season caused a significant increase in bare ground, and that cattle trails facilitate sediment transport into stream channels (George et al. 2004). Herbaceous cover at springs and creeks in the Sierra Nevada foothills declined with moderate grazing as compared to lightly grazed and ungrazed plots (Allen-Diaz and Jackson 2000).

Manure from cattle grazing washes into or is directly deposited into lakes and streams in Sierra Nevadas, introducing harmful microorganisms and nutrients (e.g. nitrogen, phosphorus) (Derlet et al. 2010). These nutrients enhance algae growth, leading to eutrophication of otherwise naturally oligotrophic aquatic ecosystems. Studies have documented high instances of contamination of lakes and streams by *Escherichia coli* in cattle-grazing areas in the Sierra Nevada in California (Derlet and Carlson 2006; Derlet et al. 2008). Similar impacts have been found from pack animals used in these areas of the national forests. Contamination of rangeland creeks by steroids, in some cases at levels high enough to impact fish in central California (Kolodziej and Sedlack 2007).

Streams in grazed areas had decreased fish biomass as compared to non-grazed areas in California's Golden Trout Wilderness (Knapp & Matthews 1996). Lower insect family richness has also been documented in lightly and moderately grazed wetlands as compared to ungrazed springs in the Sierra Nevada foothills (Allen-Diaz et al. 2004). As compared to ungrazed sites, significantly lower riparian avian abundance and richness was found in grazed locations in California and surrounding states (Tewksbury et al. 2002). Input of manure has also been documented to impact aquatic insect populations in California coastal stream systems (Del Rosario et al. 2002).

Some studies documenting impacts of grazing on riparian ecosystems in California include but are not limited to:

- Allen-Diaz, B., & Jackson, R. D. (2000). Grazing Effects on Spring Ecosystem Vegetation of California's Hardwood Rangelands.
- Allen-Diaz, B., Jackson, R. D., Bartolome, J. W., Tate, K. W., & Oates, L. G. (2002). Long-term grazing study in spring-fed wetlands reveals management tradeoffs.
- Barton, D. C., & Holmes, A. L. (2007). Off-Highway Vehicle Trail Impacts on Breeding Songbirds in Northeastern California.
- Belsky, A. J., Matzke, A., & Uselman, S. (1999). Survey of livestock influences on stream and riparian ecosystems in the western United States.
- Derlet, R. W., & Carlson, J. R. (2006). Coliform bacteria in Sierra Nevada wilderness lakes and streams: what is the impact of backpackers, pack animals, and cattle?
- Derlet, R. W., Ger, K. A., Richards, J. R., & Carlson, J. R. (2008). Risk factors for coliform bacteria in backcountry lakes and streams in the Sierra Nevada mountains: a 5-year study.
- Derlet, R. W., Goldman, C. R., & Connor, M. J. (2010). Reducing the impact of summer cattle grazing on water quality in the Sierra Nevada Mountains of California: a proposal.
- U. S. Fish & Wildlife Service, (1985). Revised Recovery Plan for the Paiute Cutthroat Trout.
- George, M. R., Tate, K. W., Larsen, R. E., Gerlach, J. D., & Fulgham, K. O. (2004). Cattle grazing has varying impacts on stream-channel erosion in oak woodlands.
- Holmquist, J. G., Schmidt-Gengenbach, J., & Haultain, S. a. (2010). Does Long-term Grazing by Pack Stock in Subalpine Wet Meadows Result in Lasting Effects on Arthropod Assemblages?
- Jackson, R. D., & Allen-Diaz, B. (2006). Spring-fed wetland and riparian plant communities respond differently to altered grazing intensity.
- Knapp, R., & Matthews, K. (1996). Livestock Grazing, Golden Trout, and Streams in the Golden Trout Wilderness, California: Impacts and Management Implications.
- Kolodziej, E. P., & Sedlak, D. L. (2007). Rangeland grazing as a source of steroid hormones to surface waters.

- Kondolf, G. M. (1994). Livestock grazing and habitat for a threatened species: Land-use decisions under scientific uncertainty in the White Mountains, California, USA.
- Ouren, B. D. S., Haas, C., & Melcher, C. P. (2007). Environmental Effects of Off-Highway Vehicles on Bureau of Land Management Lands:
- Fleischner, T. L. (1994). Ecological Costs of Livestock Grazing in Western North America
- del Rosario, R. B., Betts, E. A, & Resh, V. H. (2002). Cow Manure in Headwater Streams: Tracing Aquatic Insect Responses to Organic Enrichment.
- Shore, T. (2001). Off-Road to Ruin.
- Tewksbury, J., Black, E., Victoria, N. N. U. R., Logan, D., & Dobkin, S. (2002). Effects of anthropogenic fragmentation and livestock grazing on western riparian bird communities.
- Welsh, H. H., & Ollivier, L. M. (1998). Stream Amphibians As Indicators of Ecosystem Stress: A Case Study From California's Redwoods.

Some species affected by or at risk from grazing include, but are not limited to:

- Salmonid species, including coho (silver) salmon (*Oncorhynchus kisutch*), cutthroat trout (*Salmo clarki*), and steelhead trout (*Salmo gairdneri*): at risk of impacts from increased sedimentation in important spawning and nursery areas (Burns 1972; Hicks et al. 1991). Suspended sediment has caused gill damage and increased stress response in coho salmon in the laboratory (Lake and Hinch 1999).
- Endangered Coho salmon (*Onchorhynchus kisutch*): spawn in watersheds in Marin and Sonoma county at the same time during which high concentrations of steroids were detected in streams (Kolodziej and Sedlack 2007)
- Yosemite toad: a rare amphibian in high elevation meadows in the central Sierra Nevada; at risk of trampling by cattle in National Forest rangeland and the impacts of trampling on habitat.
- California golden trout (Knapp and Matthews 1996)
- Lens-pod Milk-vetch, *Astragalus lentiformis:* a rare endemic plant only found in one part of Plumas National Forest in the Sierra Nevadas. This plant is susceptible to trampling, and most occurrences are located in grazing allotments (Derlet et al. 2010).
- Riparian birds, including song sparrow, fox sparrow, western wood-pewee, American robin, hairy woodpecker, green-tailed towhee (Tewksbury et al. 2002) and the dwindling Willow Flycatchers (Sanders and Flett 1989)
- Paiute cutthroat trout, *Oncorhynchus clarki seleniris*: increased sedimentation in aquatic habitat from grazing implicated in habitat limitations (USFWS 2004; Kondolf 1994)
- Amphibians are subject to negative impacts from increased sedimentation (Welsh and Ollivier 1998).

3. Logging and other timber management activities effects on water quality and aquatic and riparian species and habitats

Timber management can alter riparian systems in several ways (Chamberlin et al. 1991). It can affect streamflow though changing the water balance or the rate at which water enters streams. Roadbuilding, falling, yarding, and building can have particularly strong effects on watershed hydrology and streamflow. Logging can also alter watershed hydrology through increased snow deposition, which advances the rate and timing of snowmelt. Logging can also affect water quality by altering variables such as temperature, suspended sediment, dissolved oxygen, and nutrients. It can increase sedimentation in stream channels, altering the shape of the channel and potentially affecting aquatic species and habitat.

Logging roads and skid trails accelerate runoff from slopes adjacent to streams and can increase the content of soil in the water (Chamberlin et al. 1991). In Casper Creek in northwestern California, logging has been associated with an 89% increase in suspended sediments during storm events as compared to undisturbed conditions (Lewis 1998).

In northern California, Newbold et al. (1980) found lower diversity and density of macroinvertebrate fauna in commercially logged streams as compared to unlogged streams. Mcgurk and Fong (1995) also found reduced macroinvertebrate diversity associated with forest management and logging roads in California's Sierra Nevada and Klamath mountain ranges. Bulldozers used for road building and in streams for debris removal increased sedimentation in several California trout and salmon streams (Burns 1972). In a review of the effects of increased sedimentation on aquatic ecosystems from forest management, Anderson (1996) noted effects on fish behavior, physiology, and population as well as effects on fish habitat (including spawning, rearing, food production, and summer and overwintering habitats).

In a study examining impacts of timber harvest in sensitive aquatic habitats in the Caspar Creek watershed in California, Cafferata and Spittler (1998) noted that storm sequences combined with road construction and logging resulted in numerous landslides. In this same area, logging has created a series of gullies, increasing suspended sediment yields and peakflow runoff to streams (Reid et al. 2009). It has also caused significant increases in streamflows (Keppeler & Ziemer, 1990).

Some studies documenting impacts of logging on riparian ecosystems in California include but are not limited to the following:

- Anderson, P. G. (1996). Sediment generation from forestry operations and associated effects on aquatic ecosystems.
- Burns, J. W. (1972). Some effects of logging and associated road construction on Northern California streams.
- Bury, R. B. (2008). Low thermal tolerances of stream amphibians in the Pacific Northwest: Implications for riparian and forest management.
- Chamberlin, T. W., Harr, R. D., & Everest, F. H. (1991). Timber harvesting, silviculture, and watershed processes.
- Harvey, B. C., White, J. L., & Nakamoto, R. J. (2009). The Effect of Deposited Fine Sediment on Summer Survival and Growth of Rainbow Trout in Riffles of a Small Stream.
- Hicks, B. J., Hall, J. D., Bisson, P. A., & Sedell, J. R. (1991). Responses of Salmonids to Habitat Changes.
- Keppeler, E. T., & Ziemer, R. R. (1990). Logging effects on streamflow: Water yield and summer low flows at Caspar Creek in northwestern California.
- Lake, R. G., & Hinch, S. G. (1999). Acute effects of suspended sediment angularity on juvenile coho salmon (*Oncorhynchus kisutch*).

- Lewis, J. (1998). Evaluating the Impacts of Logging Activities on Erosion and Suspended Sediment Transport in the Caspar Creek.
- Mcgurk, B. J., & Fong, D. R. (1995). Equivalent roaded area as a measure of cumulative effect of logging.
- Reid, L. M., Dewey, N. J., Lisle, T. E., & Hilton, S. (2010, April 15). The incidence and role of gullies after logging in a coastal redwood forest.
- Shaw, E. A., & Richardson, J. S. (2001). Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival.
- Stoddard, M. a, & Hayes, J. P. (2005). the Influence of Forest Management on Headwater Stream Amphibians At Multiple Spatial Scales.
- Suttle, K. B., Power, M. E., Levine, J. M., & McNeely, C. (2011). How Fine Sediment in Riverbeds Impairs Growth and Survival of Juvenile Salmonids.
- Cafferata, P. H., & Spittler, T. E. (1998). Logging Impacts of the 1970's vs. the 1990's in the Caspar Creek.
- Sarr, D., Odion, D., Hibbs, D., Weikel, J., Gresswell, R., Bury, B., Czarnomski, N., et al. (2005). Riparian Zone Forest Management and the Protection of Biodiversity.

Some species affected by or at risk from logging include but are not limited to:

- Salmonid species, including coho (silver) salmon (*Oncorhynchus kisutch*), cutthroat trout (*Salmo clarki*), and steelhead trout (*Salmo gairdneri*): at risk of impacts from increased sedimentation in important spawning and nursery areas (Burns 1972; Hicks et al. 1991; Harvey et al. 2009; Shaw and Richardson 2001; Suttle et al. 2011). Suspended sediment has caused gill damage and increased stress response in coho salmon in the laboratory (Lake and Hinch 1999). Loss of riparian vegetation alters the light and temperature conditions of streams, which in turn changes primary and secondary production, emergence times of salmonid fry, and survival of juveniles (Hicks et al. 1991). Past timber harvesting and road construction have caused declines in diversity and abundance of salmonid species in the Pacific Northwest (Sarr et al. 2005).
- Cold-water stream amphibians, including two salamander endemic to the Pacific northwest, *Rhyacotriton variegates* and *Dicamptodon tenebrosus*, are at risk from increased temperatures associated with logging (Bury 2008). The Pacific Northwest is home to three endemic families of amphibians restricted to cool-water streams: tailed frogs (Ascaphidae); torrent salamanders (Rhyacotritonidae); and Pacific giant salamanders (Dicamptodontidae). These families are also susceptible to substrate in stream waters and unfavorable habitat alterations caused by logging (Stoddard and Hayes 2005; Sarr et al. 2005). Amphibians are also subject to negative impacts from increased sedimentation (Welsh and Ollivier 1998).
- Stream macroinvertebrates: lower diversity and density associated with logging (Newbold et al. 1980; Mcgurk and Fong 1995; Sarr et al. 2005)

Moreover, the potential for water consumption by the covered activities such as range management activities (livestock grazing), timber harvest and fuel reduction activities, road upgrading and storm-proofing, and others is wholly unaddressed in the MND. Particularly in the arid areas of the state the use of water for these activities could exacerbate water quality problems associated with reduced water availability including inability to support cold water habitats and sedimentation. As a result the water consumption for such activities could also cause additional significant impacts to aquatic and riparian species. The MND fails to look at these issues as direct and indirect impacts of the proposed waiver or in a cumulative impacts context by watershed and/or regionally.

There is substantial evidence to support a fair argument that approval of the proposed project may cause significant environmental impacts, including but not limited to biological resources and water quality. Given this evidence, the Board must prepare an EIR for the proposed project, even if the agency has been presented with other evidence suggesting that the proposed project will not have any significant impact. *See, e.g., No Oil, Inc. v. City of Los Angeles*, 13 Cal. 3d 68 (1974); CEQA Guidelines § 15064(f)(1).

As shown above, the proposed project may have significant direct and indirect impacts on listed species and other riparian and aquatic species as well as other wildlife species and water quality; therefore, an EIR is required. *See, e.g.,* CEQA Guidelines §15065(a)(1) (mandatory findings of significance). Impacts to habitat for rare flora and fauna are significant under section 15065 and require full evaluation under CEQA. *See Mira Monte Homeowners Association v. Ventura County*, 165 Cal.App.3d 357, 363-364.

In summary, because the proposed statewide waiver will have significant effects on the environment (as detailed in these comments and others), including biological resources, water resources, and water quality, the Board is required to prepare an EIR.

D. The MND Relies on an Improper Baseline

The CEQA Guidelines define the project baseline as "the physical environmental conditions in the vicinity of the project, as they exist . . . at the time environmental analysis is commenced." (Guidelines, §15125). The MND misconstrues what this means in order to assert that "many of the activities and impacts discussed do not require full environmental analysis . . . because many of the activities permitted under the Proposed Statewide Waiver are already part of the environmental baseline."

The Board's position is contrary to the general purpose of CEQA which "is to be interpreted in such manner as to afford the fullest possible protection to the environment within the reasonable scope of the statutory language." (*Association for a Cleaner Environment v. Yosemite Community College Dist.* (2004) 116 Cal. App. 4th 629, 638.) The assertion is also contrary to what is meant by "baseline" in the CEQA realm – the Board is conflating ongoing activities with the condition of the existing environment in such a way as to reduce, not afford, protection to the environment and must therefore be rejected.

Here, the Board must assess the current *condition* of the state's waterways – the baseline – and then must work from there to assess how the activities it is authorizing as part of the project will additionally harm that condition. Then, the project must be mitigated to reduce any significant harm. To properly assess the "existing physical conditions in the affected area," "the real conditions on the ground," (*Communities for a Better Environment*, 48 Cal.4th at 321), the Board must determine what the state of the environment is where the project will be taking place.

Current activities of course will in part determine the current conditions – e.g., in areas where logging or grazing or ORV use is currently or did occur, the current environmental condition will be different than in areas where it is not occurring. After determining the current condition, the Board must then analyze how future logging, future grazing, and future ORV use will harm water quality. Thus, while in some forest areas activities such as logging and grazing are occurring right now, that does not in any way undo the fact that what is being analyzed by this project is the *future* actions that fall under these category of activities. The fact that some grazing and some logging and some ORV use is *currently* occurring right now is meaningless to the analysis of future activities; instead, it is only relevant for determining the current condition of the forests. Put another way, current actions should be considered because they influence the current condition, and hence baseline, for this project, but they in no way reduce the need for analysis of how future logging and future grazing and future ORV use will cause harm to water quality.

Moreover, it appears that the Board is arguing that future logging and future grazing and future ORV use are part of an ongoing activity that is already occurring and therefore do not cause additional harm. This is wrong. Just because grazing is occurring today at a particular site in no way allows the harm caused by future grazing at that site to be ignored as some sort of "ongoing" harm that is wedded to the baseline. That makes no sense as the future grazing is what is being analyzed as part of this project and therefore that future grazing must be assessed in relation to the existing condition on the ground. If the future grazing will cause significant impacts to the current condition, then an EIR is necessary. Thus, because the Board misapplies the baseline CEQA requirement to pretend that future harm can be ignored as part of ongoing harm, the Board's MND must be rejected as it defies both the purpose of CEQA and the baseline requirement itself.

E. Direct, Indirect, and Cumulative Impacts are Not Adequately Identified or Analyzed.

There is no question that the project will have significant direct and foreseeable indirect impacts on the environment. As a result, this is the time when a full environmental review should be conducted for the impacts of the project as a whole. The time for complete CEQA review of this proposed project is now, when environmental considerations still can inform the decision, and before the Board takes any steps that could foreclose any potential alternatives or mitigation measures. *Laurel Heights I*, 47 Cal.3d at 394-95; CEQA Guidelines § 15004(b)(2)(B). It does not matter for purposes of CEQA that the Board or any other public agency may need to render some later decision with regard to the specific project approvals. *See Fullerton Joint Union High Sch. Dist. v. State Bd. of Educ.* (1982) 32 Cal. 3d 779, 795. The Board cannot defer evaluation of environmental impacts until after project approval or skirt the required procedure for public review and agency scrutiny of potential impacts. *Sundstrom v. County of Mendocino* (1988) 202 Cal.App.3d 296, 307-09.

Because the project may have significant direct and indirect impacts on many environmental resources including water quality and rare and imperiled wildlife and plant species, an EIR is required. *See, e.g.,* CEQA Guidelines §15065(a)(1) (mandatory findings of significance). Impacts to habitat for rare flora and fauna are significant under section 15065 and require full evaluation under CEQA. *See Mira Monte Homeowners Association v. Ventura County*, 165 Cal.App.3d 357, 363-364.

Neither the handbook nor the MND even provide a list of the many rare, threatened and endangered species that currently inhabit and depend on water resources and water quality in areas that will be directly impacted by the activities proposed to be covered in the statewide waiver. Some of these species include but are not limited to: Southern Steelhead; Spring-run Chinook Salmon; Pink Salmon; Chum Salmon; Longfin Smelt; Santa Ana speckled dace; green sturgeon; Coho Salmon; Summer Steelhead; Sacramento Splittail; Santa Ana Sucker; Mountain yellow-legged frog; Sierra Nevada yellow-legged frog, Yosemite toad, California red-legged frog, Southwestern willow flycatcher, least Bell's vireo, and many others.

Even looking only at those species that may be directly and indirectly affected by impacts to water quality—that is without even examining the many other rare and endangered species are directly in the footprint of the activities that would be included within the statewide waiver—there are clearly many listed and rare species that may be significantly impacted by the activities approved under the waiver.

The draft waiver also, in conclusory fashion, asserts that the Category A activities are insignificant and have a "low likelihood of impacts to water quality, and as such, require no additional conditions", including for example:

> Routine annual road and OHV trail maintenance, such as culvert cleaning and low impact replacement/modification/upgrading outside of designated riparian zones, road surface improvements (paving, patching, blading, gravel surfacing), brushing, ditch cleaning and cross drain cleaning;
>
> Dispersed camping, camping in developed recreation sites, use of non-motorized trails, fence building, and similar low-impact, dispersed activities

In fact, these activities can indeed cause significant impacts to water quality and the MND has not shown that those impacts will be reduced below a level of significance. For example, Central Valley Regional Water Quality Control Board adopted a Cleanup and Abatement Order (CAO) No. R5-2009-0030 on April 23, 2009 due to dispersed camping and ORV use along the Rubicon trial which lead to unregulated human wastes disposal and excessive sediment runoff causing unacceptable impacts to water quality.

The draft waiver acknowledges that the Category B activities are at least a "moderate risk," including the following:

> Pre-Commercial thinning in designated riparian zones, or using heavy equipment, or with burning.
>
> Vegetation management, particularly prescribed burns, mechanical mastication, and the use of hand crews, adjacent to streams and drainages, or other situations or locations where likelihood of discharge exists.

Range management activities.

Understory or pile burning within designated riparian zones.

Activities conducted by hand crews in designated riparian zones and that pose a risk of discharge.

Road upgrading and storm-proofing where there is potential for discharge.

Construction of new roads (not subject to state-wide stormwater permit).

Motor vehicle trails and their use.

NPS activities associated with mining (e.g., roads, pads, cleared areas as described in finding 38(b)).

Timber harvest and fuel reduction activities, including forest restoration projects and research and demonstration projects on fuel reduction.

Watershed projects, including but not limited to instream restoration projects and legacy NPS remediation.

Although the draft waiver admits these activities can cause significant impacts, the waiver and draft MND fail to show that the proffered conditions will reduce those impacts to a level below significance. Indeed, the draft MND relies largely on general policy statements, goals, and objectives provided the Forest Service regarding the maintenance and improvement of wildlife habitat and aquatic ecosystems in concluding that the USFS guidance, the WQMH, and the waiver conditions "will ensure any impacts to biological resources in the project area are mitigated to less than significant." (MND at 40.) Not only does experience show this has not been the case in the past, but there is no basis provided in the MND to believe that such impacts, which are not even fully identified will in fact be mitigated under the USFS guidance, the handbook, or the waiver conditions. For all of these reasons and others, an EIR is necessary.

Moreover, the Waiver Application for Category B Activities is significantly flawed and will not ensure that activities that significantly impact water quality are fully assessed. The process as described burdens the regional water boards by *requiring* that they respond to any notice of intent ("NOI") from the USFS within 30 days whether or not sufficient staff time is available. In addition, by requiring the regional board evaluation only *after* the USFS approval, the process creates undue bureaucratic momentum towards approving the projects and undermines the regional board's ability to ensure a full range of alternatives have been considered to avoid significant impacts. Similarly, placing the burden on the regional board to provide factual information and a reasoned analysis in order to deny application of the waiver to a particular Category B project in response to an NOI, reverses the burden which should be on the USFS to show that the proposed project described in the NOI will not adversely impact water quality or other resources. In contrast, the proposed policy allows the USFS to provide only general statements about the compliance with the waiver conditions in "general terms" and how the proposed project "fits within the basic strategy for watershed improvements."

F. The MND Violates the Water Code and CEQA

The monitoring offered in the MND is invalid because it violates the California Water Code and CEQA. Section 13269 of the Water Code mandates that all waivers must include monitoring that is "designed to support the development and implementation of the waiver program, including but not limited to verifying the adequacy and effectiveness of the waiver's conditions." Here, however, the monitoring offered does not meet the Water Code's standard because it does not act to verify the waiver's conditions. Instead, the monitoring is so vague and ambiguous that it is impossible to tell if the monitoring will even occur let alone achieve meaningful oversight of the adequacy and effectiveness of the waiver's conditions. Moreover, the Board is attempting to rely on vague monitoring requirements of another agency by relying on Forest Service statements that are not described because in some instance they have not even been determined – in other words, the Board is relying on another agency to implement monitoring that does not even exist.

For instance, logging activities (e.g. logging roads) affect water quality by discharging sediment and are especially deleterious to salmonids.[2] The only way to make certain that road sediment will not injure salmonids, and thereby not violate the "take" provisions of the ESA and CESA, is to prevent harmful discharge of sediment into waterways that contain or impact salmonids. This is especially true in light of the fact that sediment discharge is a classic example of the severe degradation that "can result from individually minor but collectively significant [operations] taking place over a period of time."[3] CEQA Guidelines, § 15065. "[T]housands of

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[2] *See, e.g.,* NMFS Fine Sediment Presentation (3/1/11) (Dan Wilson), "Summary of Selected Information on the effects of fine sediment on anadromous salmonids," and citations therein including Harvey, B. C., J. L. White, and R. J. Nakamoto. 2009. The Effect of Deposited Fine Sediment on Summer Survival and Growth of Rainbow Trout in Riffles of a Small Stream. North American Journal of Fisheries Management 29: 434–440, 2009; Jensen, D.W. , Steel, E. Ashley , Fullerton, Aimee H. and Pess, George R. 2009. Impact of Fine Sediment on Egg-To-Fry Survival of Pacific Salmon: A Meta-Analysis of Published Studies', Reviews in Fisheries Science, 17: 3, 348 —359; Lake, R.G. and S.G. Hinch. 1999. Acute effects of suspended sediment angularity on juvenile coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 56: 862–867; Moyle. P.B. and J. J. Cech Jr. 2004. Fishes. An Introduction to Ichthyology. Fifth edition. Prentice Hall; Newcombe, C.B. and C.B. MacDonald. 1991. Effects of Suspended Sediments on Aquatic Ecosystems. North American Journal of Fisheries Management 11:72-82; Shaw, E.A, and J.S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival. Can. J. Fish. Aquat. Sci. 58: 2213–2221 (2001); Suttle. K B., M. E. Power, J.M. Levine, and C. McNeely. 2004. How Fine Sediment in Riverbeds Impairs Growth and Survival of Juvenile Salmonids. Ecological Applications, Vol. 14, No. 4 , pp. 969-974; Ward, B.R., P.A. Slaney, A.R. Facchin, and R. W. Land 1989. Sized Based Survival in Steelhead Trout (*Oncorhynchus mykiss*): Back-calculating lengths from Adults' Scales Compared to Migrating Smolts at the Keogh River British Colombia; Waters T. F. (1995) Sediment in Streams: sources, biological effects and control. American Fisheries Society Monograph 7

[3] NMFS Fine Sediment Presentation (3/1/11) (Dan Wilson), ("Small increases in fine sediment significantly increases the likelihood of take in multiple [salmonid] life stages by: Reduces growth and therefore decreases probability of ocean survival; Suffocates eggs in redds; Increases stress; Increases mortality.")

relatively small sources of pollution [can] cause a serious environmental . . . problem."[4] Thus, in order to adequately protect salmonids, the information necessary to fully assess whether, and to what extent, road operations will impact sediment discharge must be fully disclosed and explained.[5] Moreover, not only NMFS, but Cal Fire, the Department of Fish and Game, and the North Coast Regional Water Quality Control Board as well, have each explained that there is a very real and deleterious connection between logging road operations and sediment discharge into California waterways.[6] Thus, without provisions that explain and demonstrate how harmful sediment discharge will actually be avoided, the waiver fails to prevent significant impacts, thus inhibiting CEQA's substantive mandate that all significant impacts of a project be avoided or mitigated.[7]

One example of this shortcoming is the monitoring report's statement that "Each national forest will conduct road patrols to the extent allowed by weather, safety, and road conditions during and after major storms to detect and correct road drainage problems that could affect water quality." This is ambiguous to the point of being meaningless – this requirement nowhere states *how* the Forest Service will "detect and correct road drainage problems" and therefore fails to demonstrate that the Forest Service actually will *be able to* detect and correct the problem. "Conclusory statements do not fit the CEQA bill." *Californians for Alternatives to Toxics v. Dept. of Food & Agric.*, 136 Cal.App.4th at 17. Therefore, until the Board provides, and explains, monitoring that actually ensures the harmful impacts of sediment discharge will be detected and corrected, the monitoring fails as a matter of law. Moreover, nowhere is it even stated how the Forest Service will ensure that funding exists to carry out the detection and correction. For monitoring to have any meaning of course requires that it be fully implemented and therefore the state must demonstrate that funding exists for this monitoring.

Furthermore, the Handbook in which this monitoring requirement originates states that "The Regional Hydrologist will develop a template road patrol protocol and each national forest will use the template to develop its road patrol plan. Road patrol plans will describe conditions under which road patrols are appropriate, safety precautions, and monitoring, corrective, and reporting procedures." Again, this is meaningless because it does not explain what the "template road patrol protocol" will look like or consist of and, thus, is entirely conclusory and speculative

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[4] *Kings County Farm Bureau v. City of Hanford* (1990) 221 Cal.App.3d 692, 718

[5] *See Joy Road Area Forest & Watershed Assn. v. California Dept. of Forestry & Fire Prot.* (2006) 142 Cal.App.4th 656, 667 ("[T]he cumulative impact analysis must be substantively meaningful. A cumulative impact analysis which understates information concerning the severity and significance of cumulative impacts impedes meaningful public discussion and skews the decisionmaker's perspective concerning the environmental consequences of the project, the necessity for mitigation measures, and the appropriateness of project approval.")

[6] *See, e.g.*, "North Coast Regional Water Quality Control Board Presentation on Water Quality Requirements and Operations on Saturated Soils (3/1/11)"; "Information on Wet Weather Log Hauling and Impacts to Water Quality, Pete Cafferata, CAL FIRE, March 2011"

[7] Pub. Res. Code § 21002.1(b); CEQA Guidelines, § 15252 ("The document used as a substitute for an EIR or negative declaration in a certified program shall include at least the following items: . . . Alternatives to the activity and mitigation measures to avoid or reduce any significant or potentially significant effects that the project might have on the environment . . . .")

in terms of actually being effective monitoring. The Board and Forest Service need to develop the protocol, and then explain it, *as part of* the decisionmaking process so that the public and decisionmakers can gauge whether the protocol will offer any meaningful monitoring. *Association of Irritated Residents v. County of Madera* (2003) 107 Cal.App.4th 1383, 1391-1392 ("A prejudicial abuse of discretion occurs if the failure to include relevant information precludes informed decisionmaking and informed public participation, thereby thwarting the statutory goals of the EIR process.") To simply state that a protocol will be developed obviates the ability of the public and decisionmakers to scrutinize and assess whether the protocol will actually be effective.

Thus, until the Board and Forest Service offer monitoring that is actually explained and justified as to its existence and effectiveness, the MND fails as a matter of law. This is especially true given that the BMPEP monitoring that the Board otherwise relies on, in the MND's own words, "leaves many critical questions unaddressed."

Moreover, it appears that the Board has failed to adequately include trustee agencies in the process particularly (1) the California Department of Fish and Game which is the trustee agency for fish, wildlife, and plants that may be affected by the many covered activities and the permitting agency for any activities resulting in streambed alteration (Fish & Game Code §§ 711.7(a), 1600, 1602-1603), and (2) the State Lands Commission which is the trustee agency for public trust lands underlying many of the waterways at issue in the waiver that will be directly impacted by many of the covered activities. *See, e.g., National Audubon Society v. Superior Court (1983) 33 Cal.3d 419, 435; Center for Biological Diversity, Inc. v. FPL Group, Inc.* (2008) 166 Cal.App.4th 1349; *National Audubon Society v. Superior Court* (1983) 33 Ca1.3d 419.) CEQA requires that trustee agencies be consulted regarding whether an EIR is needed and on a negative declaration. Pub. Res. Code § 21080.3; CEQA Guidelines, 14 CCR § 15063(g). Unfortunately, there is no evidence in the MND or draft waiver that the Board recognizes its responsibility to ensure that the public trust resources impacted by this project (including the bed and banks of navigable waterways as well as fish and wildlife), which resources are overseen by other state agencies and commissions as trustees, will be adequately protected.

G. The Proposed Mitigation Measures Are Inadequate under CEQA Because They are not certain and are not Fully enforceable

A MND violates CEQA if it does not ensure that measures designed to mitigate impacts are "fully enforceable through permit conditions, agreements, or other measures." (Pub. Resources Code § 21081.6, subd. (b).) "The purpose of these requirements is to ensure that feasible mitigation measures will actually be implemented as a condition of development, and not merely adopted and then neglected or disregarded." (*Fed'n of Hillside & Canyon Ass'ns v. City of Los Angeles* (2000) 83 Cal.App.4th 1252, 1261.) Moreover, there must exist "a monitoring program to ensure that the mitigation measures are implemented." (*Federation of Hillside & Canyon Ass'ns v. City of Los Angeles,* Cal.App.4th at 1261.)

It is the lead agency's duty to provide mitigation and monitoring that is enforceable and clear. The Forest Handbook violates this fundamental CEQA requirement because it contains numerous statements that are not explicit requirements and instead are merely guidance. Indeed, much of the Handbook contains language that lacks mandatory terms such as "shall" or "must." Moreover, in many instances, the Handbook contains requirements that are vague or unenforceable. For instance, when a requirement uses "should" instead of "shall," or simply lacks mandatory language at all, the requirement is meaningless from an enforcement standpoint. Vague language is much the same in that it provides no meaningful guidance and therefore precludes enforceability. The Handbook does this in many instances – below are examples but are not exhaustive of the problem:

- "The design **should** consider the size and distribution of natural structures (snag and down logs) as a means of preventing erosion and sedimentation." (unenforceable)
- "Mitigations or changes needed to stabilize slopes and project or improve stream courses will be incorporated into the harvest unit design." (vague as it does not identify what the mitigation will actually be or how it will be enforced)
- "Where the harvest impacts cannot be reduced to a low or moderate level with treatments, then the harvest units **should** be avoided or harvest methods modified, or both" (unenforceable)"
- "During the timber sale planning process, the interdisciplinary team will identify and recommend limited operating periods." (vague and unenforceable)
- "Unsuitable forest lands will not be harvested until they can be harvested without irreversible or unmitigable resource effects. If the team determines that current or prospective logging methods would result in irreversible or unmitigable watershed effects, then the line officer **should** reclassify the area to unsuitable forest land and defer harvesting." (vague (does not identify what the mitigation will actually be or how it will be enforced) and unforceable)
- "Equipment will not be operated when ground conditions are such that excessive damage will result." (vague)
- "During the timber sale planning process and/or during sale appraisal, the interdisciplinary team will identify criteria for selecting treatment areas or classes of areas for special treatment and document them in the environmental assessment." (vague)
- "The sale administrator handbook section on Skid Trails and Firelines contains guidelines for spacing of cross drains, construction techniques, and cross drain heights. The sale administrator **should** use these guidelines on the ground to identify site-specific preventive work that is required of the purchaser." (vague and unenforceable)
- "**To the extent possible**, ensure drainage features are fully functional before the start of the local winter season (such as November 16 to March 31) or before the start of runoff-inducing precipitation events." (unenforceable)
- "For fish-bearing streams, the water drafting rate **should** not exceed 350 gallons per minute" (unenforceable)

Vague and unenforceable provisions are also illegal because they do not guarantee that significant impacts will in fact be avoided. The above examples, as well as much of the Handbook overall, make plain that the BMPs may or may not prevent significant impacts. It is impossible to know because *no one* knows whether the BMPs will all be implemented and enforced, and similarly, no one knows what the BMPs actually require in light of their vague nature in many instances. For example, when the BMPs offer language like "Mitigations or changes needed to stabilize slopes and protect or improve stream courses will be incorporated into the harvest unit design," it is not possible to determine whether significant impacts will actually be avoided because no one knows what the actual mitigations or changes will be. In short, vague, non-specific BMPs preclude the public and decisionmakers from being able to determine the efficacy of the BMPs. As a result, the Board cannot lawfully rely on the provisions in the handbook and the MND to assure that all impacts will be reduced below a level of significance. The Board cannot avoid its obligation to prepare a legally adequate EIR for this project.

In sum, it is not clear from the MND, waiver conditions, or the handbook that the Board intends to or could hold the Forest Service accountable for meeting water quality standards or improving the condition of impaired water segments. Rather, the handbook and waiver are permeated with vague and discretionary language that point towards an (unlawful) expectation that the Forest Service implementation of the BMPs will not be rigorous and lack of funding or Forest Service staffing may be used to excuse the Forest Service from meeting needed water quality standards. Thus the proffered mitigation does not provide an adequate basis for issuance of a waiver or reliance on an MND.

H. The Failure to Evaluate Alternatives Violates CEQA

The Board has not provided sufficient information about feasible alternatives to comply with CEQA. Pursuant to CEQA and the guidelines, "public agencies shall not undertake actions concerning the proposed public project that would have a significant adverse effect or limit the choice of alternatives or mitigation measures, before completion of CEQA compliance." CEQA Guidelines § 15004(b)(2). In particular, an agency shall not "take any action which gives impetus to a planned or foreseeable project in a manner that forecloses alternatives or mitigation measures that would ordinarily be part of CEQA review of that public project." CEQA Guidelines § 15004(b)(2)(B).

As noted above, the time for complete CEQA review of this proposed project is now, when environmental considerations still can inform the decision for the proposed statewide waiver, and before the Board takes any steps that could foreclose any potential alternatives or mitigation measures. *Laurel Heights I*, 47 Cal.3d at 394-95; CEQA Guidelines § 15004(b)(2)(B). It does not matter for purposes of CEQA that the Board, or any other public agency may need to render some later decision with regard to any site specific projects. *See Fullerton Joint Union High Sch. Dist. v. State Bd. of Educ.*, 32 Cal. 3d 779, 795 (1982). The Board cannot defer evaluation of environmental impacts of this decision until after the proposed waiver is approved or skirt the required procedure for public review and agency scrutiny of potential impacts. *Sundstrom v. County of Mendocino* (1988) 202 Cal.App.3d 296, 307-09.

CEQA requires that public agencies should not approve projects as proposed if there are feasible alternatives or feasible mitigation measures available which would substantially lessen the significant environmental effects of such projects. *See* Public Resources Code § 21002. In this case, alternatives that should have been considered and fully analyzed but were not, include but are not limited to, the following alternatives: regional waivers with conditions that take into account the vastly different ecosystems within the public lands managed by the Forest Service across the State; statewide or regional permits with enforceable requirements instead of a waiver with conditions; and/or requiring the Forest Service to adopt a water quality management plan that is enforceable under federal law rather than rely on a handbook which provides guidance but may not be enforceable in federal or state court. Additional alternative that should have been considered include, but are not limited to, requiring different BMPs for example: providing limits on the number of low-water ORV trail stream crossings in each watershed; excluding all livestock from all wetlands and riparian areas; limiting the number of acres of soil disturbance in each watershed to prevent excess siltation; prohibiting activities that cause loss of riparian vegetation and impair water quality in stream segments that provide habitat for endangered or threatened aquatic species, etc.

Please do not hesitate to contact me if you have any questions about these comments or the attached reference materials. Thank you for considering these comments.

Sincerely,

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STREAM AMPHIBIANS AS INDICATORS OF ECOSYSTEM STRESS: A CASE STUDY FROM CALIFORNIA'S REDWOODS

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Abstract. Road construction of the Redwood National Park highway bypass resulted in a large accidental infusion of fine sediments into pristine streams in Prairie Creek State Park, California, during an October 1989 storm event. This incident provided a natural experiment where we could measure, compare, and evaluate native stream amphibian densities as indicators of stream ecosystem stress. We employed a habitat-based, stratified sampling design to assess the impacts of these sediments on the densities of aquatic amphibians in five impacted streams by comparing them with densities in five adjacent, unimpacted (control) streams. Three species were sampled in numbers sufficient to be informative: tailed frogs (*Ascaphus truei,* larvae), Pacific giant salamanders (*Dicamptodon tenebrosus,* paedomorphs and larvae), and southern torrent salamanders (*Rhyacotriton variegatus,* adults and larvae). Densities of amphibians were significantly lower in the streams impacted by sediment. While sediment effects were species specific, reflecting differential use of stream microhabitats, the shared vulnerability of these species to infusions of fine sediments is probably the result of their common reliance on interstitial spaces in the streambed matrix for critical life requisites, such as cover and foraging. Many stream-dwelling amphibians are highly philopatric and long-lived, and they exist in relatively stable populations. These attributes make them more tractable and reliable indicators of potential biotic diversity in stream ecosystems than anadromous fish or macroinvertebrates, and their relative abundance can be a useful indicator of stream condition.

Key words: Ascaphus truei; *bioindicators; California;* Dicamptodon tenebrosus; *ecosystem stress; redwood ecosystem;* Rhyacotriton variegatus; *sedimentation; stream amphibians.*

INTRODUCTION

The condition of the physical habitat is critically important in stream (lotic) ecosystems and can change more easily and quickly than in most other ecosystems (Power et al. 1988). Sedimentation of aquatic ecosystems is a common outcome of many land management activities, including timber harvesting, road building, mining, and grazing (Meehan 1991, Reid 1993, Waters 1995). Consequently, stress due to increased sedimentation is one of the most common causes of ecological dysfunction in lotic ecosystems (Waters 1995). The negative impacts of sediments on stream-dwelling organisms, including fishes, stream and benthic invertebrates, and periphyton, are well documented (Newcombe and MacDonald 1991, Meehan 1991, Waters 1995). However, few studies have examined the direct effects of sediments on stream-dwelling amphibians (see Hall et al. 1978, Hawkins et al. 1983, Bury and Corn 1988, Corn and Bury 1989).

In the developing lexicon of ecosystem ''health'' (see Suter 1993 for a critique of the health analogy applied to ecosystems), there is consensus that ''unhealthy'' or stressed ecosystems manifest common symptoms of degradation (Godron and Forman 1983, Odum 1985, Steedman and Regier 1987). Among these symptoms of ecosystem dysfunction are: (1) alteration in biotic community structure to favor smaller life forms; (2) reduced species diversity, (3) increased dominance by ''*r*'' selected species, (4) increased dominance by exotic species, (5) shortened food-chain length, (6) increased disease prevalence, and (7) reduced population stability (Rapport 1992). While stressed ecosystems do not always manifest all of the above symptoms, in the majority of cases, most do appear (Rapport et al. 1985). The major challenge in ecosystem diagnosis is to identify early warning signs of incipient pathology (Rapport 1992, Rapport and Regier 1995). Odum (1992) noted that ''the first signs of environmental stress usually occur at the population level, affecting especially sensitive species'' (see also Rapport and Regier 1995). Such sensitive species are obvious candidates for indicator species. The use of indicator species is fraught with pitfalls and must be based on precise definitions and procedures to be effective and credible (Landres et al. 1988). However, the approach of finding and monitoring early indicators of ecosystem stress has the advantage of shortening the relatively slow response time of the whole ecosystem to stress by shifting attention to the much quicker response time of sensitive species (Rapport 1992). Such indicators would ideally have the combined attributes

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of being holistic, early warning, and diagnostic (Rapport 1992). Furthermore, these indicators need to be abundant and tractable elements of the system whose natural perturbations can be distinguished from states indicative of ecosystem dysfunction.

Amphibians are thought to be sensitive to perturbations in both terrestrial and aquatic environments because of their dual life histories, highly specialized physiological adaptations, and specific microhabitat requirements (Bury 1988, Vitt et al. 1990, Wake 1990, Olson 1992, Blaustein 1994, Blaustein et al. 1994*a*, Stebbins and Cohen 1995). During their aquatic stages, many stream-dwelling amphibian larvae are highly specialized in their uses of lotic microhabitats for both foraging and cover. Such specialized adaptations can render them susceptible to even minor environmental changes that alter their ability to seek cover from predators and to forage for phytoplankton, zooplankton, insects, and other invertebrates. In lotic habitats these specializations are shared with early life stages of both anadromous and freshwater fishes, as well as many stream invertebrates. Amphibians are relatively long-lived compared with invertebrates and fishes (e.g., Moyle 1976, Groot and Margolis 1991). Daugherty and Sheldon (1982*a*) reported a tailed frog with a known age of 14 yr, and Hairston (1987) reported longevity records for six families of salamanders that ranged from 10 to 55 yr. Amphibians are also highly philopatric compared to most fishes (see Daugherty and Sheldon 1982*b*, Welsh and Lind 1992), can occur in relatively stable numbers (Hairston 1987), and are readily sampled. Thus, we believe they are potentially more tractable and reliable environmental indicators than these other taxa. Few studies have been designed specifically to examine the responses of amphibians to environmental perturbations in aquatic ecosystems (but see Moyle 1973, Hall et al. 1978, Hawkins et al. 1983, Hayes and Jennings 1986, Corn and Bury 1989, Welsh 1990, Blaustein et al. 1994*b*). In this paper we report the results of a study of amphibian population responses to alterations of the physical habitat in streams due to abnormal infusions of fine sediments and evaluate the use of amphibians as indicators of stream ecosystem dysfunction.

The primary challenge with indicator species, or any study where causal arguments are being made about shifts in presence or abundance, lies in separating any natural fluctuations in numbers from those attributable to anthropogenic environmental stresses (Pechmann et al. 1991, Blaustein 1994, Blaustein et al. 1994*b*, Pechmann and Wilbur 1994). The coast redwood (*Sequoia sempervirens*) ecosystem (Zinke 1977) where our study was conducted is self-perpetuating and in a late-seral or old-growth stage (i.e., in a steady state; Bormann and Likens 1979; see also Franklin and Hemstrom 1981, Veirs 1982). Based on the resistance–resilience model of ecosystem stability (Waide 1995), the coastal redwood ecosystem is among the most stable on the planet, and even the relatively dynamic lotic environment (Power et al. 1988) within late seral redwood forest is comparatively stable. Contrasting the potential life-spans of the native amphibians relative to that of the trees that define this ecosystem, it is certainly a highly stable environment from the perspective of the amphibians. We believe that it is reasonable to assume that in such a stable system, natural population perturbations within the amphibian assemblage would be minimized, and marked changes in their numbers over a short period of time could confidently be considered an indication of ecosystem dysfunction. Even with metamorphosis and the consequent movement of individuals from aquatic to terrestrial environments, populations of long-lived species with multiyear larval periods would remain relatively stable. Any pulses of newly hatched larvae entering the system could easily be accounted for in analysis by removing the first year class if that were appropriate given the question being addressed. While we can offer no direct evidence from the Pacific Northwest in support of our assumption of stable amphibian populations in stable environments, there are relevant data from forested ecosystems of the eastern United States. Hairston (1987) indicated that stream salamander populations from the Appalachian Mountains (*Desmognathus* spp.) have remained stable for up to seven years (length of time studied). He also reported stable populations in pond and terrestrial environments (see also Hairston and Wiley 1993), and concluded that salamander populations are apparently minimally affected by stochastic events, unless these events are destructive of the habitat (Hairston 1987).

A combination of natural and anthropogenic events during the fall of 1989 created a natural experiment, which afforded us an opportunity to test the response of amphibians to ecosystem stress in streams of an old-growth redwood ecosystem. The Redwood National Park bypass project was a large highway construction project adjacent to the eastern border of Prairie Creek Redwoods State Park, Humboldt County, California. This area received >12.7 cm of precipitation during a major storm 20–23 October 1989, which resulted in large infusions of sediments from the ongoing road construction into seven stream channels in the Prairie Creek drainage. The fine sediment layer deposited on affected streambeds measured 0.3–5.0 cm in depth (Anonymous 1991).

Here we provide an analysis of the effects of this combination of shallow mass wasting and surficial erosion (hereafter the erosion event) on densities of the three most abundant native, stream-dwelling amphibians in five of these streams. Our approach was to examine and compare these densities with those of the same species in five unimpacted (control) streams in the same basin. We also examined fine-scale microhabitat relationships within the unimpacted streams to help interpret any differences in amphibian numbers

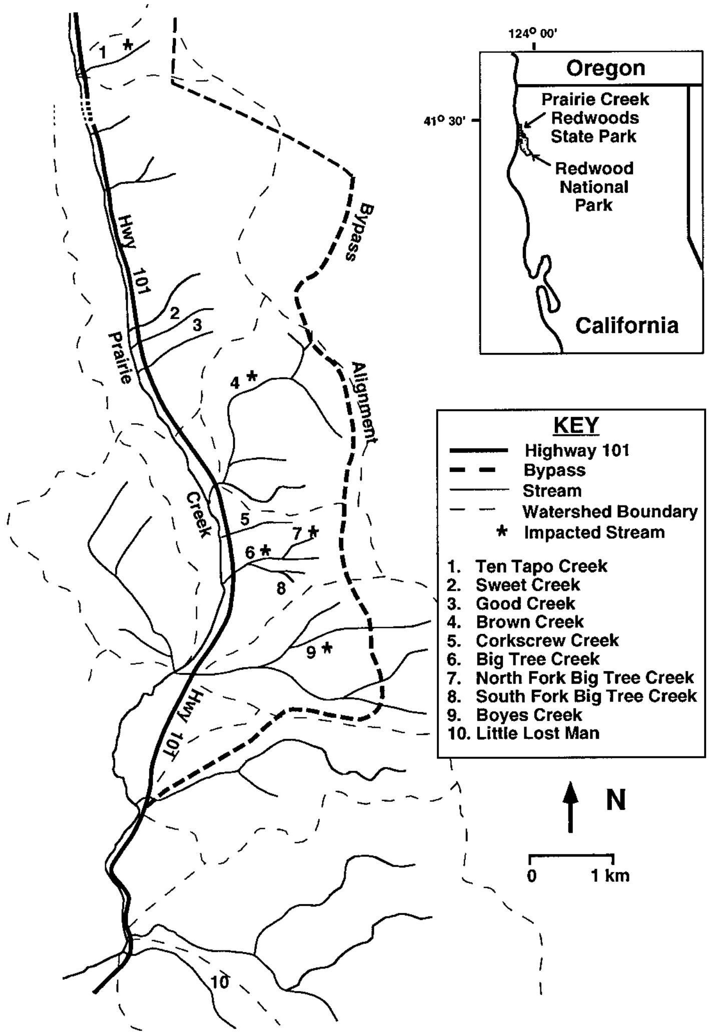

FIG. 1. Locations of impacted (*) and unimpacted streams in Prairie Creek State Redwoods and Redwood National Park, Humboldt County, California. All drainages were sampled for amphibians from June through August 1990. (Modified from Welsh et al. 1997.)

that might be revealed between the impacted and unimpacted sets of streams.

Site and species accounts

For our study of the impacts of the erosion event on the amphibian community we selected the five of seven streams affected by the event that drained westward into Prairie Creek (Anonymous 1991). Our five control streams were selected from those unimpacted streams in the same drainage, with a similar westward aspect, that were interspersed among the impacted streams (Fig. 1). The two sets of streams (five unimpacted and five impacted) were of similar size and orientation, and vegetative cover. Of the total set of 10 streams, nine were located within Prairie Creek Redwoods State Park and one control stream (Little Lost Man Creek) was located in the same drainage basin in adjacent Redwood National Park (Fig. 1).

Three species of amphibians were sufficiently abundant in these streams to enable our study.

Pacific giant salamander.—The larval and paedomorphic forms of this salamander are strictly aquatic and general accounts of their habitat describe them as bottom dwellers in mountain streams, lakes, and ponds (Nussbaum et al. 1983, Leonard et al. 1993) where they are often found under cobble-size substrates (Parker 1991, Welsh 1993). This salamander can be extremely abundant in small streams of the Pacific Northwest, accounting for as much as 99% of the predator biomass in such systems (Murphy and Hall 1981, Hawkins et al. 1983). Larvae of this species typically require two complete summers of growth before metamorphosis occurs (Leonard et al. 1993).

Tailed frog.—Welsh (1993) summarized the niche for the larval tailed frog as ''. . . clear, cool, fast-flowing streams in coniferous forests of the Pacific Northwest.'' Conditions within streams with larvae ''. . . consisted of fast current over coarse gravel, pebble, cobble, or boulder substrates, with little fine sediment'' (Welsh 1993). These conditions included intermediate to high water velocity and cold water temperatures (Welsh 1990, 1993; see also H. H. Welsh and A. J. Lind, *unpublished manuscript*). The strong association with fast-flowing, cold water habitats probably reflects the evolutionary history of this frog (sensu Holt 1987). Tailed frogs are unique among temperate anurans in being specifically adapted to these unusual and extreme conditions (cf. deVlaming and Bury 1970, Gradwell 1971, Claussen 1973, Brown 1975). Larvae from lowland populations of the tailed frog typically require 1–2 yr before metamorphosis occurs (Leonard et al. 1993).

Southern torrent salamander.—General descriptions of the habitat of this small, secretive salamander indicate that it occurs in and along small streams, spring heads, and seepages (Anderson 1968, Nussbaum and Tait 1977, Nussbaum et al. 1983, Good and Wake 1992, Welsh 1993, Welsh and Lind 1996). Larval individuals can be found in the loose substrates of small streambeds. Adults are both stream and streamside dwellers, occurring where water flows through a matrix of unsorted rock substrates (J. Baucom, *personal communication*). Typical habitats include the splash zones of rocky tumbling brooks and waterfalls. Adults often occur side-by-side with larvae within coarse substrates in streams (Welsh and Lind 1992, 1996). The southern torrent salamander has a four and one-half to five year larval period (Leonard et al. 1993).

METHODS

From June to August 1990, we sampled five impacted (subjected to a mass sediment infusion) and five unimpacted streams. Our study design assumed that amphibian community composition and densities in the unimpacted streams resembled the composition and densities present in the impacted streams had the erosion event not occurred. The similarities and proximity of these 10 streams, the stability of the coast redwood ecosystem, and the lack of any documented historical perturbations that impacted any of these streams prior to the highway construction project, all support this assumption. We alternated sampling between impacted and unimpacted streams to ameliorate the effects of any recruitment of newly hatched larval amphibians on the density estimates. In addition, we tested the supposition that the two stream sets were geomorphically similar (see *Methods: Comparisons of physical habitat*).

Habitat typing of streams

Our sampling design was stratified by mesohabitat type (e.g., pool, run, riffle, and other types; Welsh et al. 1997). Prior to sampling for amphibians, each stream was mapped from Highway 101 east to its headwaters (Fig. 1). The mapping included the subdivision and classification of streams at the level of geomorphological reach type (braided, alluvial, or confined) and stream mesohabitat composition (Appendix).

Comparisons of physical habitat in unimpacted and impacted streams

We lumped similar mesohabitat types into five composite categories (after Hawkins et al. 1993), in order to increase sample sizes and simplify analyses: (1) all pools, including main channel, backwater, and secondary channel pools; (2) glides and runs; (3) riffles; (4) step runs; and (5) step pools. These five categories are hereafter referred to as the primary mesohabitat types (Appendix).

In order to insure that any differences in amphibian densities detected between the unimpacted and impacted streams could not be attributed to differences in stream reach type (alluvial, braided, or confined) or differences in the composition of primary mesohabitat types, we tested for differences in these parameters between the two stream sets. We performed unpaired Student's t tests (Zar 1995) of the mean proportions of stream length by reach type and primary mesohabitat type for each set of streams. The significance level (α) was set at 0.05 with a Bonferroni adjustment (Stevens 1986) applied for multiple tests (α for mesohabitat type tests = 0.01; α for reach type tests = 0.017).

To evaluate sediment loads in each stream we sampled the pool mesohabitats where fine sediments (<2 mm) tend to collect (Lisle and Hilton 1992). Fine sediment depths were measured at three locations in each pool bowl (the upstream end, the middle, and at the downstream end) (Appendix), with the three measurements averaged for analysis. We also visually estimated the percentage of embedded coarse substrate at the pool tail (Appendix). The two pool sediment variables were employed to evaluate differences in fine sediments between the two sets of streams but were not used in the analyses of amphibian densities. Unpaired Student's t tests were used to test differences in the mean sediment depth and the mean percentage of pool tail substrate embedded for each set of streams (α = 0.05).

Amphibian sampling

Stream habitats for amphibian sampling were selected using a random systematic design based on stream length and ratios of primary mesohabitat types along each stream (Welsh et al. 1997). Working from west to east (upstream) and beginning at Highway 101 (Fig. 1), we sampled the first unit of every mesohabitat type encountered, then a randomly selected unit of each type between the second and the sixth, then every fifth unit of each type thereafter. This provided a proportional sampling effort of each mesohabitat type relative to its availability in each stream.

Within each selected stream mesohabitat unit, we systematically placed one or more amphibian sampling units (cross stream belt transects) based on habitat length, placing one belt transect for every 10 m of habitat (Fig. 2). Belt transects (hereafter belts) were 0.6 m wide and extended from bank to bank so that

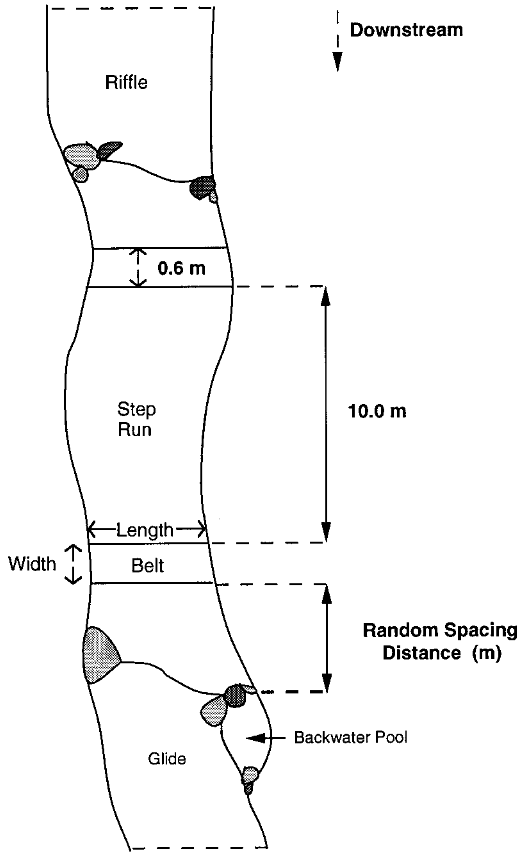

FIG. 2. Schematic representation of random-systematic belt placement within selected mesohabitats (see *Methods: Amphibian sampling*). Modified from Welsh et al. (1997).

sampling unit length varied with stream width. The length of each mesohabitat unit was divided by the total number of belts desired (approximately one every 10 m) to determine exact spacing, and a random distance between 0 and 10 m was used to determine placement of the first belt (Welsh et al. 1997). Each belt was then thoroughly searched for amphibians. The area was first scanned for visible animals and then all cover objects were removed working from bank to bank and upstream until the entire area was searched. Animals were captured using a metal mesh net, identified, sexed (if possible), measured (snout–vent and total length), and released after sampling was completed. Cover objects were returned to their original positions. We are confident that our searches captured all amphibians present in the open, and probably most of those under the first layer of large substrate (>16 mm diameter).

Three species were detected and sampled in numbers sufficient for statistical analyses: larval and paedomorphic (animals with larval morphology and sexual maturity) Pacific giant salamanders (*Dicamptodon tenebrosus*), larval tailed frogs (*Ascaphus truei*), and larval and adult southern torrent salamanders (*Rhyacotriton variegatus*). We did not differentiate larval and paedomorphic Pacific giant salamanders, and they were combined for analysis. Only six adult tailed frogs were captured. Because of this small sample and their primarily terrestrial habitat associations, they were omitted from the analyses. Four adult torrent salamanders were found, and because they occur in the same aquatic microhabitats as the larvae, the two life stages were combined for analyses. Histograms of snout–vent length indicated that our sampling occurred after the recruitment of Pacific giant and southern torrent salamander larvae, and before the recruitment of tailed frog larvae to our stream set.

Biotic and abiotic measurements associated with amphibian sampling

In order to characterize fine-scale or microhabitat attributes associated with amphibian captures, we estimated or measured 28 microhabitat parameters associated with the individual belt samples (Fig. 2) (Appendix: microhabitat attributes).

Statistical analyses

We used Statistical Analysis System (SAS version 6.12; SAS Institute 1997) to conduct all data analyses. In contrast to the stricter $\alpha = 0.05$ used in testing for differences in geomorphology and pool fine sediment levels among the sets of streams, we set $\alpha = 0.10$ for our analysis of variance (ANOVA), analysis of covariance (ANCOVA), and correlation analysis. This moderate α provides a criterion more appropriate for the detection of ecological trends and it increases statistical power (Toft and Shea 1983, Toft 1991) (see Schrader-Frechette and McCoy [1993] for a thorough justification and evaluation of this methodological approach in ecology). Dependent variables were natural log-transformed, and some independent variables were arcsine-transformed, to meet the assumptions of normality and homogeneity of variance.

Analysis of variance.—We used partial hierarchical ANOVA to test for differences in densities of each amphibian species (the dependent variables) between impacted and unimpacted streams. Within each impact category (impacted and unimpacted) there are five streams, and within those streams five mesohabitat types are possible. This method permits us to partition the total variability into three components while adjusting for unequal sample sizes within the different levels. The unit of analysis was the mesohabitat unit (i.e., mesohabitat types within streams within impacts). The effect for impact was calculated using streams within impact as the mean square error (MSE). The effects for mesohabitat type and impact by mesohabitat type interaction were calculated using mesohabitat type within stream within impact as the MSE. The mean squares were calculated using Type I sums of squares (SS) as all mesohabitats were sampled in proportion to their occurrence in the population (Milliken and Johnson 1984). This was not the case with the overall model

F; thus it was not used to determine model significance. The following null hypotheses were tested:

H_{0_1}: There are no significant differences between impact and no impact for any of the three species;

H_{0_2}: There are no significant differences among mesohabitat types for any of the three species;

H_{0_3}: There is no interaction between mesohabitat type and impact for any of the three species.

When ANOVA provided evidence of differences among mesohabitats, we used Tukey's studentized multiple range test to compare the means.

Analysis of covariance.—In order to more closely examine the effects of specific fine sediment parameters (Appendix: fine aquatic substrates) on individual species we employed ANCOVA. We used this method to look for evidence of other possible effects of the erosion event that were not measured during our sampling (e.g., chronic suspended sediment load, bed instability) that may be indirectly related to sediment transport. This allowed us to adjust the ANOVA models by each fine sediment variable measured (Appendix). The model structure for the ANCOVA is the same as that of the ANOVA described (partial hierarchical) and employed Type I SS. The following null hypotheses were tested:

H_{0_1}: There are no significant differences between impact and no impact for any of the three species, when densities are adjusted by each of the fine sediment covariates;

H_{0_2}: There are no significant differences among mesohabitat types for any of the three species, when densities are adjusted by each of the fine sediment covariates;

H_{0_3}: There are no significant differences for the interaction of mesohabitat type and impact for any of the three species, when densities are adjusted by each of the fine sediment covariates.

The five fine sediment parameters consisted of two visual estimates of substrate composition, one estimate of substrate condition, and two measures of fine sediment derived from grab samples collected immediately adjacent and upstream of the belts (Appendix: fine aquatic substrates). In order to simplify the ANCOVA by eliminating redundancy among closely related variables, we used correlation analysis to select one variable from those pairs that described a similar parameter (percentage fines and silt volume, $r = 0.466$, $P < 0.0001$; percentage sand and sand volume, $r = 0.303$, $P < 0.0001$). From each of these pairs we chose the variable with the highest correlation with our dependent variables (percentage fines), or if the significant *r* values were equivocal relative to the dependent variables, we chose the measured variable (sand volume).

Significant covariates were determined using Type III sums of squares. Only those ANCOVA results with a reduced error variance for our tests were meaningful. Consequently, only those models with a decreased overall MSE were evaluated further. ANCOVA models that failed to reduce the MSE over the ANOVA or had a nonsignificant *F* for the covariate, failed to explain additional effects beyond those detected in the ANOVA.

Correlation analysis.—We performed correlation analyses of 28 microhabitat attributes measured or estimated within each belt sample (Appendix). We restricted this analysis to those data from belts in the control streams with captures of each of the three species in order to address the question "what measured or estimated microhabitat variables best characterized the fine-scale ecological relationships of the resident amphibians under pristine stream conditions?"

Results

Comparisons of physical attributes between impacted and unimpacted streams

We surveyed and habitat typed 3.6 km of impacted streams and 3.2 km of unimpacted streams (Fig. 1). Comparisons of mean proportions of stream length by reach type and primary mesohabitat type indicated that there were no significant differences between the impacted and unimpacted sets of streams (Table 1). Assuming that the relative amount of available habitat is a reasonable indicator of the number of organisms that may be supported there (Southwood 1977, 1988), we consider that this lack of difference in geomorphological composition supported our assumption that the amphibian assemblages in the two sets of streams probably would have had similar species composition and densities had the erosion event not occurred.

Mean fine sediment (<2.0 mm) depths in the impacted pools ranged from 0.1 to 25.0 cm compared with 0.0–4.0 cm in the unimpacted pools (Fig. 3). Percentage embeddedness of pool tails ranged from 10 to 100% in the impacted streams and from 0 to 85% in the unimpacted streams. Tests between the two sets of streams for both the mean sediment depth in pool bowls, and the percentage of substrate embeddedness at the pool tails, showed significantly greater amounts of sediment in the impacted streams (Fig. 3). This clearly demonstrates an impact effect of the 1989 erosion event still remained when we sampled in 1990.

Comparisons of amphibian densities between impacted and unimpacted streams

We sampled a total of 267 belts in 179 mesohabitat units, with 93 habitat units (137 belts) in the impacted streams and 86 habitat units (130 belts) in the unimpacted streams. We captured a total of 540 amphibians; larval and paedomorphic individuals of the Pacific giant salamander were the most common ($n = 296$), followed by larval tailed frogs ($n = 205$), and larval and adult southern torrent salamanders ($n = 39$).

Analysis of amphibian densities.—Densities of the three species varied by mesohabitat type and impact (Fig. 4). The Pacific giant and southern torrent salamanders showed significant differences for impact

TABLE 1. Comparison of reach types and mesohabitat composition for 10 streams sampled for aquatic amphibians in Prairie Creek Redwoods State Park and Redwood National Park, Humboldt County, California, 1990.

	Reach types			Mesohabitat types				
Stream	Alluvial	Braided	Confined	All pools	Glide/run	Riffle	Step run	Step pool
Unimpacted streams								
Corkscrew	0	0	100.0	5.3	2.0	41.5	9.7	41.5
Good	0	67.7	32.3	4.4	5.6	40.6	46.2	4.2
Little Lost Man	67.7	0	32.3	7.6	1.4	3.4	37.0	51.1
S. fork Big Tree	0	0	100.0	10.8	6.2	14.9	0.0	68.1
Sweet	0	3.2	96.8	3.0	0.7	36.6	39.1	21.0
$\bar{x}$	13.5	14.2	72.3	6.2	3.2	27.4	26.4	37.2
	(14.0)	(13.0)	(16.0)	(1.4)	(1.1)	(7.7)	(9.1)	(11.0)
Impacted streams								
Big Tree	6.0	27.7	66.3	20.6	3.9	19.3	11.4	45.6
Boyes	84.9	15.1	0	19.9	4.8	19.6	30.2	18.0
Brown	0	0	100.0	26.2	8.5	20.7	22.0	24.3
N. fork Big Tree	0	20.2	79.8	3.7	0.0	12.0	4.2	80.2
Ten Tapo	81.9	18.1	0	18.0	0.0	14.7	12.2	56.4
$\bar{x}$	34.6	16.2	49.2	17.7	3.4	17.3	16.0	44.9
	(20.0)	(4.6)	(21.0)	(3.7)	(1.6)	(1.7)	(4.6)	(11.0)
t	−0.87	−0.14	0.87	−2.88	−0.12	1.29	1.03	−0.48
P†	0.41	0.89	0.41	0.02	0.91	0.27	0.33	0.64

Note: Percentage of stream length by reach and mesohabitat type, mean, and standard error (in parentheses) are reported. Comparisons of impacted and unimpacted streams were made using Student's *t*.†

† Significant *t* probability values were interpreted using a Bonferroni adjustment (Stevens 1986).

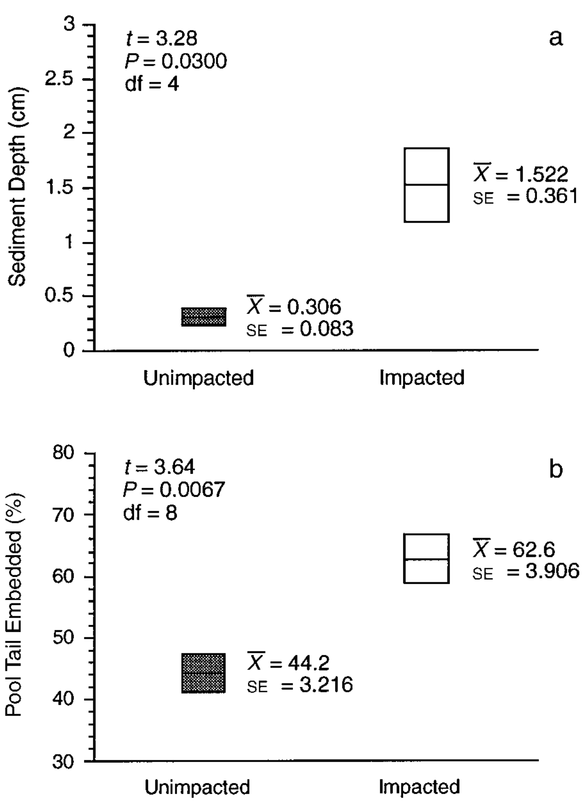

FIG. 3. Comparisons of sediment depths (a) and pool tail embeddedness estimates (b) in impacted and unimpacted streams. Boxes indicate means (from three measures of sediment depth [cm] along the central axis at the top, middle, and bottom of each pool bowl), ± 1 SE.

(sedimentation); in all cases the densities in unimpacted streams were greater (Table 2a). The tailed frog and southern torrent salamander showed significant differences among mesohabitat types (Table 2a). There was also a significant interaction between impact and mesohabitat type for the tailed frog (Table 2a).

In the impacted streams, there were no significant differences among mesohabitat types for the Pacific giant and southern torrent salamanders. However, tailed frog density was significantly greater in riffles compared to pools (Table 2b). In the unimpacted streams tailed frog larvae showed strong habitat specialization and were significantly more abundant in both riffles and step runs compared with other mesohabitat types (Table 2b). The torrent salamander also occurred more often in riffle than pool habitat in the unimpacted streams (Table 2b), although there were no captures in pools, glides, or runs (Fig. 4). There were no differences in mesohabitat type for the Pacific giant salamander in the unimpacted streams (Fig. 4).

Effects of fine sediment attributes.—The Pacific giant salamander and tailed frog yielded significant covariate models (Table 3), indicating additional variation was explained beyond the ANOVA. For these same dependent variables, percentage embedded caused the greatest reduction in variability (Table 3). In the models that were adjusted for percentage embedded, there were no significant differences detected with respect to impact (the first hypothesis test) for the Pacific giant salamander, or the tailed frog, indicating that once the data were adjusted for this covariate no further differences could be explained (Table 3a, b).

With respect to percentage embedded, the Pacific

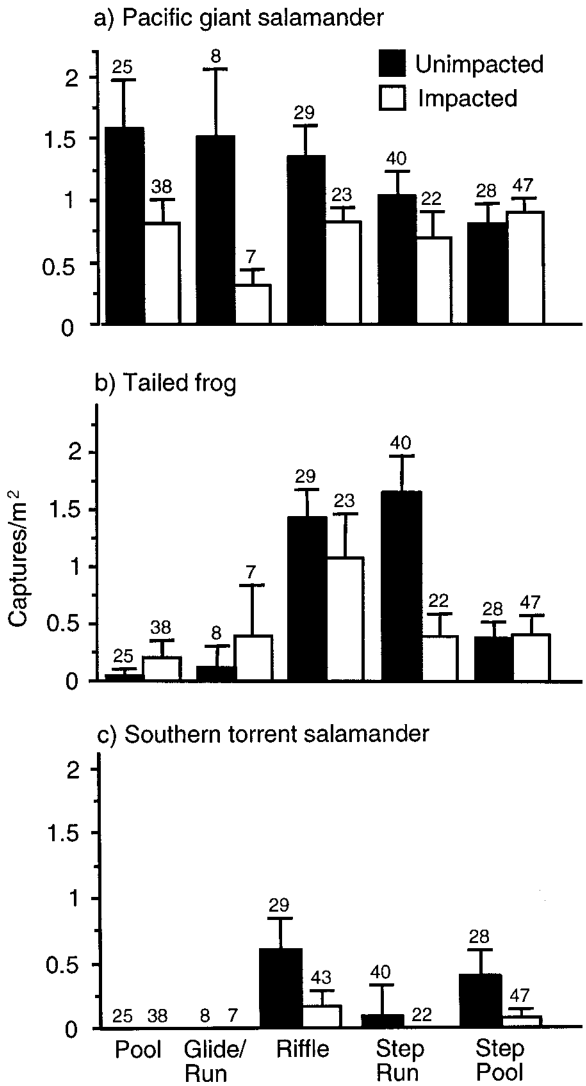

FIG. 4. Densities of three species of amphibians are shown with respect to impact and mesohabitat type. Bars represent means (and one standard error) for the stream sets (five streams in each). Numbers over bars are belts sampled.

giant salamander showed no differences for mesohabitat type or the interaction (Table 3a). In the model adjusted for percentage embedded, the tailed frog showed significant results in the tests for mesohabitat type and its interaction with impact, indicating that additional sediment effects were influencing the system beyond those explained by the ANOVA (Table 2) and the adjustment for percentage embedded (Table 3b).

The Pacific giant salamander had one additional significant covariate, percentage fines. As with percentage embedded above, no significant differences were found in the tests (Table 3a). There were no other significant covariate models for any of the three species (Table 3).

Correlation analyses of microhabitat attributes.—Of the 28 microhabitat parameters we examined, 14 were significantly correlated with amphibian density (Table 4). Nine attributes were correlated with Pacific giant salamander density, two attributes were correlated with tailed frog density, and five attributes were correlated with southern torrent salamander density (Table 4). The two salamander species responded differently to flow rates within belts. The Pacific giant salamander densities were lower in areas of high flow, while southern torrent salamander densities increased with flow rate (Table 4). Pacific giant salamander density increased in belts with larger amounts of woody debris cover, while southern torrent salamander density declined in association with both wood cover and substrates (Table 4).

DISCUSSION

Our study indicated that the stream amphibian community was negatively impacted by the erosion event caused by the bypass construction and the October 1989 storm (Table 3, Fig. 4). Our analysis indicated that this response differed considerably by species (Table 3). For example, the ANCOVA model for the Pacific giant salamander suggests that it is less sensitive than the other species to fine sediments (Table 3), but it was negatively associated with sand (Table 4). Our ANCOVA results for the tailed frog (Table 3) suggested that the impact of the erosion event acted at the level of microhabitat within streams and consisted primarily of fine particles restricting access to the streambed matrix (i.e., percentage embedded) (cf. Lisle 1989, Lisle and Lewis 1992). However, the significant results for both mesohabitat type and the interaction effects (Table 3) indicated that additional factors may be affecting the tailed frog. For the Pacific giant salamander and the tailed frog, we found significant positive associations with relatively coarse substrates (e.g., cobble; Table 4), where matrix interstices can be reduced or eliminated by fine sediments (i.e., percentage embedded).

Pacific giant salamander

The Pacific giant salamander was the least habitat specific, showing no clear association with any particular stream mesohabitat type (Fig. 4, Table 3). As a habitat generalist, this species is most likely affected by sedimentation across all stream mesohabitat types, but probably more so in pools where fine sediment accumulation is greatest (Lisle and Hilton 1992).

Analysis of substrate associations indicated that higher relative amounts of gravel and cobble were the best predictors of Pacific giant salamander abundance (Table 4; H. H. Welsh and A. J. Lind, *unpublished manuscript*). This outcome underscores the relative importance of coarse, rocky substrates, which have a high relative amount of interstitial space (see also Welsh 1993). Parker (1991) experimentally demonstrated the importance of cobble-size substrates as cover for larval Pacific giant salamanders in pool habitats in a stream similar to ours in northwestern California. Concomitantly, we found fewer salamanders in areas with great-

TABLE 2. (a) Partial hierarchical analysis of variance (ANOVA) of three amphibian species by impact (presence or absence of fine sediment infusion), stream number, and mesohabitat type, and (b) Tukey pairwise comparisons of mesohabitat types.

a) ANOVA results

Factor	df	MSE	F	P	Result
Dependent: Pacific giant salamander					
Overall model	46, 132	0.2799	0.96	0.5588	
Tests					
Impact	1, 8	1.5010	3.95	0.0820	U > I†
Mesohabitat type	4, 29	0.2932	1.58	0.2050	NS
Impact × Mesohabitat type	4, 29	0.3058	1.65	0.1881	NS
Dependent: Tailed frog					
Overall model	46, 132	0.1803	2.72	0.0001	
Tests					
Impact	1, 8	0.9252	2.06	0.1888	NS
Mesohabitat type	4, 29	2.2925	11.38	0.0001	
Impact × Mesohabitat type	4, 29	0.7507	3.73	0.0145	
Dependent: Southern torrent salamander					
Overall model	46, 132	0.0568	2.78	0.0001	
Tests					
Impact	1, 8	0.7982	4.93	0.0572	U > I†
Mesohabitat type	4, 29	0.3144	2.67	0.0519	
Impact × Mesohabitat type	4, 29	0.1258	1.07	0.3896	NS

b) Tukey pairwise comparison results‡

Pacific giant salamander	Comparison: Mesohabitat type (with respect to Impact)				
Impacted streams	Glide/run	Step Pool	Pool	Step Run	Riffle
Unimpacted streams	Step Pool	Riffle	Glide/Run	Pool	Step Run
Tailed frog	Comparison: Impact × Mesohabitat type				
Impacted streams	Pool	Glide/Run	Step Run	Step Pool	Riffle
Unimpacted streams	Pool	Glide/Run	Step Pool	Riffle	Step Run
Southern torrent salamander	Comparison: Mesohabitat type (with respect to Impact)				
Impacted streams	Pool	Step Run	Glide/Run	Step Pool	Riffle
Unimpacted streams	Pool	Glide/Run	Step Run	Step Pool	Riffle

† U = unimpacted streams, I = impacted streams.
‡ Amphibian mean density increases from left to right; lines indicate nonrejecting subsets.

er volumes of sand (Table 4), a condition that limits available interstitial spaces (see also Hall et al. 1978, Murphy and Hall 1981, Murphy et al. 1981, Hawkins et al. 1983, Corn and Bury 1989). However, none of the fine sediment variables alone could explain the significant differences we saw in giant salamander abundances with respect to impact (Table 2, Table 3). Because giant salamanders use more available stream mesohabitat types (Fig. 4), it is possible they are better able to compensate for habitat loss resulting from sedimentation (Table 2). Such adjustments might involve changing habitat use patterns or even modifying preferred sites by excavating sediments as has been seen with an ambystomatid salamander (e.g., Jennings 1996), but these hypotheses are currently untested.

Tailed frog larvae

Tailed frog larvae were the most specific in habitat use, showing a strong association with step runs and riffles vs. step pools and all other stream mesohabitat types (Fig. 4). They also demonstrated a strong association with coarse substrates (cobble) (Table 4; see also Nussbaum et al. 1983, Welsh 1993; H. H. Welsh and A. J. Lind, *unpublished manuscript*). Coarse substrates provide the interstitial space important for cover from both predation and high winter stream flows (e.g., Metter 1963, 1968), as well as providing abundant surface area for diatom production, an important food source. Fast-water habitats are less prone to trapping sediment due to the higher, more uniform velocity of water (Lisle and Hilton 1992). However, results for the tailed frog showed a significant interaction between sediment impact and mesohabitat type (Table 3). This indicated that tailed frog larvae were adversely impacted even in those high velocity habitats that are likely to have lower sediment loads (Fig. 4). This result suggests that something other than sediment filling the interstices was affecting tailed frog abundances in impacted streams. Sediment may impact critical food resources, both in adjacent lower gradient areas and in those mesohabitats occupied by tailed frog larvae. When we examined data from across all streams, we

TABLE 3. Partial hierarchical analysis of covariance of three species by impact (presence or absence of sediment), stream number, and mesohabitat type. The covariates were sediment variables taken in association with animal sampling.

Factor	df	MSE	F	P
a) Dependent: Pacific giant salamander				
i) Overall model	47, 130	0.268	1.11	0.3127
Covariate: Percentage embedded	1, 8	1.812	6.75	0.0105†
Tests				
Impact	1, 8	0.071	0.19	0.6718
Mesohabitat type	4, 29	0.346	1.84	0.1484
Impact × Mesohabitat type	4, 29	0.313	1.66	0.1860
ii) Overall model	47, 131	0.269	1.11	0.3199
Covariate: Percentage fines	1, 8	1.711	6.36	0.0129†
Tests				
Impact	1, 8	0.474	1.27	0.2928
Mesohabitat type	4, 29	0.311	1.61	0.1984
Impact × Mesohabitat type	4, 29	0.353	1.83	0.1510
iii) Overall model	47, 130	0.280	0.96	0.5512
Covariate: Sand volume	1, 8	0.248	0.89	0.3476
b) Dependent: Tailed frog				
i) Overall model	47, 130	0.166	3.17	0.0001
Covariate: Percentage embedded	1, 8	2.255	13.61	0.0003†
Tests				
Impact	1, 8	0.519	1.34	0.2808
Mesohabitat type	4, 29	0.933	3.82	0.0129‡
Impact × Mesohabitat type	4, 29	0.720	2.95	0.0367‡
ii) Overall model	47, 131	0.178	2.75	0.0001
Covariate: Percentage fines	1, 8	0.470	2.64	0.1065
iii) Overall model	47, 130	0.179	2.72	0.0001
Covariate: Sand volume	1, 8	0.435	2.42	0.1219
c) Dependent: Southern torrent salamander				
i) Overall model	47, 130	0.058	2.69	0.0001
Covariate: Percentage embedded	1, 8	0.024	0.41	0.5212
ii) Overall model	47, 131	0.057	2.71	0.0001
Covariate: Percentage fines	1, 8	0.009	0.16	0.6881
iii) Overall model	47, 130	0.057	2.82	0.0001
Covariate: Sand volume	1, 8	0.008	0.14	0.7119

Note: Test results are not reported for those models lacking a significant covariate.
† Covariate models with a significant model F using Type III SS and reduction in the MSE in the overall model over that of the ANOVA.
‡ Hypothesis tests with a significant effect detected using Type I SS after the covariate has been incorporated into the model.

found highly significant negative correlations between percentage of nonfilamentous algae and the three fine sediment variables used in our ANCOVA (percentage embedded, $r = -0.572$, $P = 0.0001$; percentage fines, $r = -0.476$, $P = 0.0001$; sand volume, $r = -.393$, $P = 0.0001$). Welsh (1993) reported that the amount of nonfilamentous algae (diatoms or periphyton) was a significant predictor of the presence and abundance of tailed frog larvae. Diatoms are the primary food for larval tailed frogs (Metter 1964, Nussbaum et al. 1983), so it follows that they would occur in greater abundance where periphyton is plentiful and avoid areas where it is sparse or absent. Even a thin layer of fine sediment can block sufficient light and inhibit the growth of algae (Newcombe and MacDonald 1991). During high flows greater amounts of sediment might scour algae off streambed substrates and thereby reduce periphyton biomass (Alabaster and Lloyd 1982).

Southern torrent salamander

The southern torrent salamander demonstrated intermediate mesohabitat specificity compared with the other two species examined. Southern torrent salamanders were absent from pools, and glides and runs. They occurred predominately in riffles, step runs, and step pools (Fig. 4). Thus, all of the mesohabitat types where they did occur were comprised primarily of moving and mixing waters. Even in these mesohabitats, southern torrent salamanders were found in higher abundance in the thalweg (main flow) and appeared to avoid mesohabitats composed primarily of margin (Table 4). This meso- and microhabitat specificity may be related to physiological constraints resulting from their specialized, reduced gill-arch system that restricts them to habitats that are characterized by cold, highly oxygenated water (i.e., mountain brooks, Valentine and Dennis 1964). The specific meso- and microhabitat associations of the southern torrent salamander could reflect a response to lower sediment loads in these habitats, but the lack of an interaction (Table 2) suggests that this habitat specificity is an ecological or evolutionary adaptation (Holt 1987) rather than a temporary response to adverse conditions. This species also ap-

TABLE 4. Significant results of Pearson product-moment correlations are reported for 14 microhabitat variables (Appendix). Correlations were performed using stream belts with captures in unimpacted streams.

Variable	Pacific giant salamander†	Tailed frog‡	Southern torrent salamander§
Aquatic conditions			
Water temperature	−0.335	···	···
Proportion margin	···	···	−0.448
Flow thalweg	−0.282	···	0.311
Cover types			
Woody debris cover	0.318	···	−0.421
Riparian vegetation	0.197	···	···
Large rock cover	−0.383	···	···
Without cover	···	···	0.502
Coarse aquatic substrates			
Cobble	0.245	0.298	···
Large rock substrates	−0.437	···	···
Fine gravel volume	0.234	···	···
Woody debris substrates	0.273	···	−0.460
Fine aquatic substrates			
Embedded	···	−0.461	···
Fines	···	···	−0.654
Sand volume	−0.272	···	···

† Correlations with salamander density are based on 78 belts with salamander captures; correlations > 0.188 are significant at $P = 0.10$.

‡ Correlations with tadpole density using 49 stream belts in the four primary mesohabitat types that had tailed frog captures (step runs, step pools, runs/glides, riffles); correlations > 0.238 are significant at $P = 0.10$.

§ Correlations with salamander density using 19 stream belts in the three primary mesohabitat types that had southern torrent salamander captures (step runs, step pools, and riffles); correlations > 0.389 are significant at $P = 0.10$.

peared to use areas lacking large cover objects (Table 4). We suspect that their avoidance of wood cover and substrates could be a means to elude predatory Pacific giant salamanders, which were often found associated with this cover type (Table 4). Stebbins (1953) and Nussbaum (1969) also speculated that Pacific giant salamander presence may restrict southern torrent salamander distribution.

The lack of a significant covariate model for the southern torrent salamander indicated that no further effects were detected over what was indicated by the ANOVA. However, the correlation analysis for this salamander showed a strong negative relationship with percentage fines (Table 4). Previous research also concluded that torrent salamanders are sensitive to fine sediments in, and substrate embeddedness of, the streambed matrix (Welsh 1993, Welsh and Lind 1996). Nonetheless, the southern torrent salamander may be able to compensate to some degree for the negative effects of sedimentation by favoring shallow stream microhabitats with steady flow where they occur in close association with cobble substrates (Welsh 1993, Welsh and Lind 1996). However, we cannot discount the possibility that the lack of an interaction effect may have resulted from the low number of belts with captures (10%) or high variability, which may have partially compromised our ability to detect differences.

In summary, our study indicated that sediment deposits from the October 1989 storm event had a negative effect on amphibian populations, with a pronounced effect on two out of three species examined. Furthermore, our ANCOVA results add new insight into the explanation for reduced abundances of tailed frog larvae based on sedimentation of interstices offered by Corn and Bury (1989). It appears that tailed frog larval abundances were reduced by some factor other than the direct impact of embeddedness, possibly as a result of the inhibition of periphyton growth, the scouring of that growth from streambed substrates, or both. Our results also documented differential use of stream mesohabitats by two of these species, and demonstrate how fine sediments can differentially affect stream amphibians in accordance with their particular meso- and microhabitat associations.

Amphibians as bioindicators

Results of our analyses are consistent with other studies that examined the habitat associations of these species at finer spatial scales and in ecosystems other than the redwoods (Murphy et al. 1981, Hawkins et al. 1983, Corn and Bury 1989, Bury et al. 1991, Parker 1991, Welsh 1993; H. H. Welsh and A. J. Lind, *unpublished manuscript*). Bury and Corn (1988) discussed the potential negative impacts of erosion events on stream amphibians of the Pacific Northwest. Such impacts have been documented for other stream systems in connection with timber harvesting activities and associated road building (Burns 1972, Beschta 1978, Rice et al. 1979, Reid and Dunne 1984, Chamberlin et al. 1991, Furniss et al. 1991). Corn and Bury (1989) documented differences in amphibian species richness and in the density and biomass of southern torrent salamanders, tailed frog larvae, and Pacific giant salamanders in logged vs. unlogged streams in southern Oregon. They attributed these declines to loss of critical microhabitat due to infusions of fine sediments. Populations of stream amphibians can be particularly sensitive to increased siltation because they frequent interstitial spaces among the loose, coarse substrates that comprise the matrix of most natural streambeds of the Pacific Northwest (Bury and Corn 1988, Corn and Bury 1989). Sedimentation fills these spaces, reducing available cover and foraging area and, undoubtedly, has similar impacts on other substrate-dwelling biota (cf. Lisle 1989, Lisle and Lewis 1992; see also Waters 1995).

As to the question of their applicability as bioindicators of environmental stress, we conclude that measuring and monitoring stream amphibian densities can provide a highly suitable and extremely sensitive barometer of ecological stress resulting from fine sediment inputs, arguably one of the most pervasive stressors of lotic systems worldwide (Waters 1995). Other studies have indicated that the tailed frog and torrent salamander also show a marked sensitivity to another stressor in lotic systems, increased water temperature (Brattstrom 1963, deVlaming and Bury 1970, Claussen 1973, Welsh 1990, Welsh and Lind 1996). We believe that stream amphibians demonstrate strong potential as ''sensitive species'' (cf. Odum 1992), whose numbers can change relatively quickly in response to a range of environmental perturbations. Furthermore, use of streambed interstices by amphibians is a characteristic shared with early life stages of both resident and anadromous fishes, as well as many stream invertebrates. These other taxa, however, are either short-lived, explosive breeders, or subject to seasonal movements, all of which can complicate their use as bioindicators. Many species of stream-dwelling amphibians are highly philopatric, long-lived, and occur in relatively stable populations in undisturbed ecosystems. These attributes can make their relative numbers a useful and reliable indicator of environmental perturbations, both from known causes (Corn and Bury 1989, Blaustein et al. 1994*b*) and also possibly from causes that have yet to be identified (e.g., Corn and Fogleman 1984, Weygoldt 1989, Drost and Fellers 1996, Laurance 1996, Laurance et al. 1996, Pounds et al. 1997, Woolbright 1997, Lips 1998).

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APPENDIX

Definitions of primary mesohabitat types, pool sediment measures, and microhabitat attributes measured or estimated in association with belt samples.

Term	Definition
a) Mesohabitat attributes	
i) Primary mesohabitat types†	
All pools	Reaches with water depths from shallow to deep with evidence of scour. Cause of scour may be an obstruction, blockage, merging of flows, or constriction. This type includes main channel, lateral, backwater, and secondary channel pools. Flow velocities range from very low to swift. Substrate size is highly variable.
Run/glide	Wide shallow reaches flowing smoothly, with little surface agitation and no major flow obstructions. Velocities are low to moderate. These often appear as flood riffles. Typical substrates are gravel, cobble, and boulders.
Riffle	Shallow to moderately deep, swift, turbulent water. Amount of exposed substrate will vary. Substrates are usually cobble or boulder dominated.
Step run	A sequence of runs separated by short riffle steps. Substrates are usually cobble and boulder dominated.
Step pools	A sequence of pools separated by short riffle steps. Substrates are usually cobble and boulder dominated.
ii) Pool sediment measures	
Pool tail embedded	Visual estimate (percentage) of vertical surfaces of large substrates buried in fines and/or sand in pool tail.
Pool bowl sediment depth	Depth of sediment to the nearest tenth of a centimeter is taken at three points along the midline of the pool bowl. These measures are then averaged.
b) Microhabitat attributes	Measures and estimates of microhabitat attributes taken in association with amphibian sampling.
i) Aquatic conditions	
Proportion margin‡	Visual estimate (percentage) of channel composed of margin flow (percentage).
Proportion intermediate	Visual estimate (percentage) of channel composed of intermediate flow.
Proportion thalweg	Visual estimate (percentage) of channel flow composed of thalweg flow.
Flow margin	Flow rate in channel margin measured with a flowmeter in centimeters per second.
Flow intermediate	Flow rate in intermediate channel flow measured with a flowmeter in centimeters per second.
Flow thalweg	Flow rate in channel thalweg measured with a flowmeter in centimeters per second.
Canopy open‡	Measured by densiometer at center of the belt (percentage).
Water temperature	Measured by thermometer (°C).
Density of other amphibians§	Density (captures per square meter) of the two other species of amphibians present in the belt.
ii) Cover estimates	Visual estimate of instream cover (percentage) in a series of categories.
Undercut banks‡	Overhang of stream banks, within 30 cm of water surface.
Woody debris‡	Woody debris of any size, including leaf litter overhanging water surface or underwater.
Riparian vegetation‡	Vegetation growing on the banks or in the stream. Must overhang within 30 cm of the water surface.
Large rock‡	Comprised of boulders and bedrock ledges. Only those portions that provide an overhang capable of hiding an amphibian are counted in this estimate.
Without cover‡	Portion of the belt lacking any of the above cover types.
iii) Coarse aquatic substrates‖	Visual estimate of belt surface area comprised of coarse substrates (percentage) in the following categories.
Gravel	2.0–32.0 mm in diameter
Pebble	32.0–64.0 mm in diameter
Cobble	64.0–256.0 mm in diameter
Large rock	>256.0 mm in diameter and bedrock
Woody debris‡	Woody debris of any size and leaf litter. Must be in or surrounded by water.
Fine gravel volume	Proportion of mass of sediment sample taken at each belt (2.0–16.0 mm diameter).
Coarse gravel volume	Proportion of mass of sediment sample taken at each belt (16.0–32.0 mm diameter).
iv) Fine aquatic substrates‖	
Embedded	Visual estimate (percentage) of vertical surfaces of large substrates buried in fines and/or sand in the belt.
Fines‡	Visual estimate (percentage) of belt surface area comprised of substrates <0.06 mm diameter.
Sand‡	Visual estimate (percentage) of belt surface area comprised of substrates 0.06–2.0 mm diameter.
Silt volume‡	Proportion of mass of sediment sample taken at each belt (samples are dried before sifting and weighing; <0.063 mm diam).
Sand volume‡	Proportion of mass of sediment sample taken at each belt (0.063–2.0 mm diameter).
Nonfilamentous algae	Visual estimate (percentage) of belt substrates covered by nonfilamentous algae growth.

† Modified from Hawkins et al. (1993).
‡ Variable is transformed using arcsine to meet assumptions of normality.
§ Variable is transformed using natural log to meet assumptions of normality.
‖ Particle size based on Platts et al. 1983.

REVISED RECOVERY PLAN

FOR THE

PAIUTE CUTTHROAT TROUT

(*Oncorhynchus clarki seleniris*)

Original Approved: January 25, 1985

Region 1
U. S. Fish and Wildlife Service
Portland, Oregon

Approved: ____
Manager, California/Nevada Operations Office

Date: ____

DISCLAIMER

Recovery plans delineate reasonable actions that are believed to be required to recover and/or protect listed species. We, the U.S. Fish and Wildlife Service, publish recovery plans, sometimes preparing them with the assistance of recovery teams, contractors, State agencies, and other affected and interested parties. Objectives of the recovery plan will be attained and any necessary funds made available subject to budgetary and other constraints affecting the parties involved, as well as the need to address other priorities. Recovery plans do not obligate other parties to undertake specific tasks and may not represent the views nor the official positions or approval of any individuals or agencies involved in recovery plan formulation, other than our own. They represent our official position *only* after they have been signed by the California/Nevada Operations Manager, Regional Director, or Director as *approved*. Recovery plans are reviewed by the public and submitted to additional peer review before we adopt them as approved final documents. Approved recovery plans are subject to modification as dictated by new findings, changes in species status, and the completion of recovery actions.

LITERATURE CITATION SHOULD READ AS FOLLOWS:

U.S. Fish and Wildlife Service. 2004. Revised Recovery Plan for the Paiute cutthroat trout (*Oncorhynchus clarki seleniris*). Portland, Oregon. ix + 105 pp.

An electronic version of this recovery plan will also be made available at http:/pacific.fws.gov/ecoservices/endangered/recovery/plans.html and http://endangered.fws.gov/recovery/index.html.

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EXECUTIVE SUMMARY

<u>Background</u>: The Silver King Creek drainage is located on the eastern slope of the Sierra Nevada Range, in Alpine County, California. It is a major tributary to the East Fork of the Carson River, which drains into the Lahontan Basin. It provides habitat for one fish species, Paiute cutthroat trout (*Oncorhynchus clarki seleniris*), that is listed as threatened under the Endangered Species Act of 1973, as amended. It also provides known or potential habitat for two amphibian candidate species, the Sierra Nevada population of the mountain yellow-legged frog (*Rana muscosa*) and the Yosemite toad (*Bufo canorus*). All Paiute cutthroat trout recovery actions were evaluated to minimize adverse impacts to the frog and toad.

<u>Current Species Status</u>: The Paiute cutthroat trout was originally listed as endangered on March 11, 1967 (U.S. Fish and Wildlife Service 1967) under the Endangered Species Preservation Act of 1966. On July 16, 1975, Paiute cutthroat was reclassified as threatened under the Endangered Species Act of 1973 (U.S. Fish and Wildlife Service 1975) to facilitate management and allow regulated angling. It currently occupies approximately 18.6 kilometers (11.5 miles) of historically fishless stream habitat in the Silver King drainage above Llewellyn Falls and above a barrier in Corral and Coyote Creeks (Figures 1 and 2). Four self-sustaining, genetically pure populations of Paiute cutthroat trout are known to occur out-of-basin in the North Fork of Cottonwood Creek, Stairway Creek, Sharktooth Creek, and Cabin Creek (Figures 1, 3, and 4).

<u>Recovery Priority</u>: The Paiute cutthroat trout has a recovery priority number of 9, per criteria published by a Federal Register notice in 1983 (U.S. Fish and Wildlife Service 1983). This priority number indicates a subspecies with moderate degree of threat and a high potential for recovery.

<u>Habitat requirements</u>: The life history and habitat requirements for Paiute cutthroat trout are similar to those reported for other western stream-dwelling salmonids. All life stages require cool, well-oxygenated waters. Adult fish prefer stream pool habitat in low gradient meadows with undercut or overhanging banks and abundant riparian vegetation. Paiute cutthroat trout can survive in lakes, but there is no evidence that they ever occurred naturally in any of the lakes within the

Silver King basin. To spawn successfully, they must have access to flowing waters with clean gravel substrates.

Recovery Goal: Recovery of Paiute cutthroat trout sufficient to allow delisting of the species.

Recovery Objectives: Improve the status and habitat of Paiute cutthroat trout and eliminate competition from nonnative salmonid species.

Recovery Criteria: Paiute cutthroat trout will be considered for delisting when the following objectives are met:

1) All nonnative salmonids are removed from Silver King Creek and its tributaries downstream of Llewellyn Falls to fish barriers in Silver King Canyon;

2) A viable population occupies all historic habitat in Silver King Creek and its tributaries downstream of Llewellyn Falls to fish barriers in Silver King Canyon;

3) Paiute cutthroat trout habitat is maintained in all occupied streams;

4) The refuge populations in Corral and Coyote Creeks, Silver King Creek, and tributaries above Llewellyn Falls as well as out-of-basin populations are maintained as refugia and are secured from the introduction of other salmonid species; and

5) A long-term conservation plan and conservation agreement are developed, which will be the guiding management documents once Paiute cutthroat trout are delisted.

Recovery Actions:

1. Remove nonnative trout from historic Paiute cutthroat trout habitat.
2. Reintroduce Paiute cutthroat trout into historic habitat.
3. Protect and enhance all occupied Paiute cutthroat trout habitat.
4. Continue to monitor and manage existing and reintroduced populations.
5 Develop a long-term conservation plan and conservation agreement.
6. Provide public information.

Implementation Participants: The California Department of Fish and Game and the U.S. Forest Service will assist the U.S. Fish and Wildlife Service in implementing recovery tasks.

Total Estimated Cost of Recovery ($1,000's):

Year	Action 1	Action 2	Action 3	Action 4	Action 5	Action 6
2004	38	--	2	19.73	--	2.9
2005	31	--	49.5	31.23	--	2.9
2006	31	--	51.1	38.31	--	2.9
2007	--	8	37	20.73	--	2.9
2008	--	8	4.08	20.73	--	0.4
2009	--	8	3.6	23.81	--	--
2010	--	8	2	20.73	6	--
2011	--	8	2	20.73	6	--
2012	--	--	3.6	20.81	--	--
2013	--	--	4.08	18.73	--	--
TOTAL	100	40	158.95	235.5	12	12

The total estimated cost of recovering Paiute cutthroat trout is $558,450, plus additional costs that cannot be estimated at this time.

Date of Recovery: Delisting of the Paiute cutthroat trout could be initiated in 2013, if tasks are implemented as recommended and recovery criteria are met.

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I. INTRODUCTION

A. Brief Overview

The Paiute cutthroat trout (*Oncorhynchus clarki seleniris*) is native to Silver King Creek in the East Fork Carson River drainage on the Humboldt-Toiyabe National Forest, Alpine County, California. This basin also provides known or potential habitat for two amphibian candidate species, the Sierra Nevada population of the mountain yellow legged frog (*Rana muscosa*) and the Yosemite toad (*Bufo canorus*). Paiute cutthroat trout evolved in isolation from other fish species in this headwater tributary of the Lahontan Basin.

The Paiute cutthroat trout was originally listed as endangered on March 11, 1967 (U.S. Fish and Wildlife Service 1967) under the Endangered Species Preservation Act of 1966. On July 16, 1975, the Paiute cutthroat trout was reclassified as threatened under the Endangered Species Act of 1973 (U.S. Fish and Wildlife Service 1975) to facilitate management and allow regulated angling. Critical habitat has not been designated for this species. The historical distribution of Paiute cutthroat trout is thought to have been limited to Silver King Creek and its tributaries below an impassable barrier (Llewellyn Falls) to downstream barriers located in Silver King Canyon. In the early part of the twentieth century they were eliminated from their presumed historic habitat through hybridization with introduced rainbow trout (*Oncorhynchus mykiss*), golden trout (*Oncorhynchus mykiss aguabonita*), and Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*). Their range was extended into the upper reaches of Silver King Creek and its tributaries by one or more unofficial transplants of fish above Llewellyn Falls starting in 1912.

The current distribution of Paiute cutthroat trout within the Silver King Creek drainage is the upper reaches of Silver King Creek and its tributaries above Llewellyn Falls, and Corral Valley and Coyote Valley Creeks below Llewellyn Falls. The progeny of these early day transplants have been introduced into several other lakes and streams in California and at least four self-sustaining populations have become established outside the historic drainage (Figure 1). The four out-of-basin populations occur in the North Fork of Cottonwood Creek

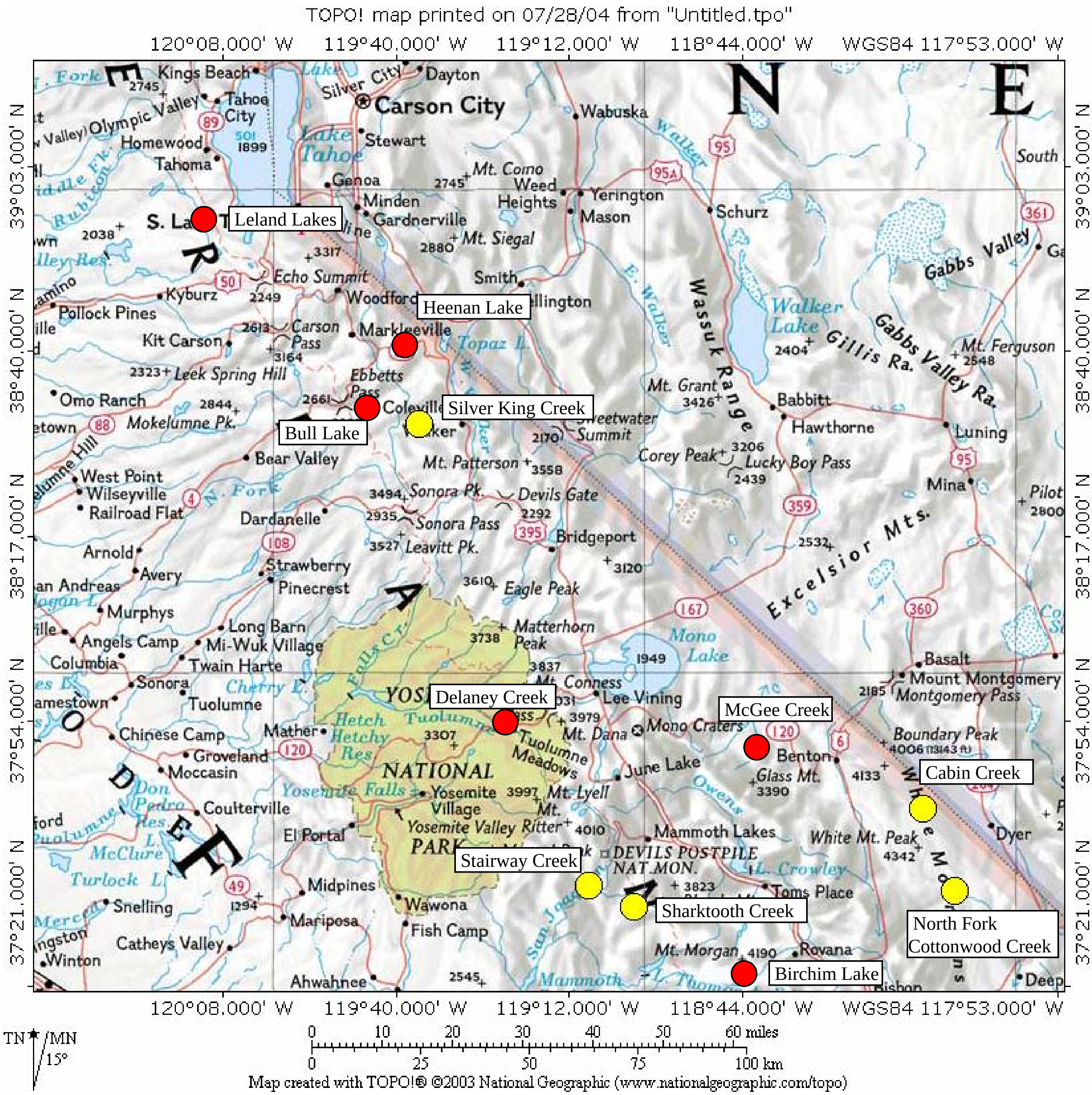

Figure 1. Distribution of Paiute cutthroat trout in east-central California, showing locations of currently occupied streams (yellow circles) and introductions that have failed or introgressed with other trout species (red circles).

and Cabin Creek (Inyo National Forest, Mono County, California), Sharktooth Creek (Sierra National Forest, Fresno County, California), and Stairway Creek (Sierra National Forest, Madera County, California). To prevent the extinction of this fish and to attain its recovery, all viable extant populations must be maintained and secured, nonnative fish must be removed from historic habitat, and Paiute cutthroat trout must be successfully reintroduced into Silver King Creek from Llewellyn Falls downstream to Silver King Canyon.

A recovery plan for the Paiute cutthroat trout was prepared in 1985 (U.S. Fish and Wildlife Service 1985). The objectives of the 1985 recovery plan were to reestablish a pure population of Paiute cutthroat trout in Silver King Creek above Llewellyn Falls, and secure and maintain the integrity of the occupied habitats in Silver King Creek, North Fork Cottonwood Creek, and Stairway Creek, all which occur outside of the presumed historic habitat. The 1985 recovery plan did not address recovering Paiute cutthroat in its historic habitat because it was not known that natural barriers existed which would prevent upstream migration of non-native salmonids into historic habitat. This revised recovery plan will incorporate recent research data and address the species' current status, threats, distribution, and recovery needs. It also addresses the effects of recovery actions on the mountain yellow-legged frog and Yosemite toad, both of which occur within the Silver King Creek drainage and at the sites of the out-of-basin populations. All Paiute cutthroat trout recovery actions have been evaluated to minimize adverse impacts to the frog and toad. In keeping with our current policy (U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration 1994), this recovery plan identifies tasks to maintain ecosystem integrity as well as recover the listed species.

Based on new information and completed tasks, we have determined it is necessary to revise recovery criteria and tasks within the 1985 Paiute cutthroat trout recovery plan. The new information and completed tasks include: 1) the discovery of fish barriers downstream of Llewellyn Falls that would enable the expansion of Paiute cutthroat trout into historic habitat, 2) elimination and reduction of threats to existing populations, 3) increased knowledge about Paiute cutthroat trout population dynamics based on long-term trend data, and

4) information about the current status of out-of-basin populations based on recent population estimates.

The extremely limited native range of the Paiute cutthroat trout, approximately 14.7 kilometers (9.1 miles) of stream habitat within a single watershed (Figure 2), is the primary factor in identifying recovery tasks. Potential recovery activities within the native range include the reintroduction of Paiute cutthroat trout downstream from Llewellyn Falls to Silver King Canyon once nonnative fish have been removed, and the protection of stream habitat in the Silver King Creek watershed. If the Paiute cutthroat trout occurred only in its currently occupied habitat, it would be highly vulnerable to extinction because: 1) genetic diversity could be dramatically reduced by a catastrophic event within any of the five drainages it currently occupies, 2) populations could become quickly introgressed (lose their distinctiveness due to introduction of genes from another population into the gene pool) as the result of an unauthorized introduction of other salmonids, and 3) genetic diversity could be subjected to additional severe bottlenecks due to inadequate population size. However, reintroduction of Paiute cutthroat trout to historical habitat, in combination with populations existing upstream of Llewellyn Falls and out-of-basin, will substantially reduce these extinction threats.

B. Species Description

The Paiute cutthroat trout is a distinctive member of the cutthroat trout complex, distinguishable from other cutthroat trouts by body coloration and the absence, or near absence, of body spots. Snyder (1933, 1934) described these fish as a new species, (*Salmo seleniris*), based on: 1) absence of body spots, 2) slender body form, 3) relatively small scales, and 4) vivid coloration. Subsequent comparisons of the type specimens with other cutthroat subspecies (Ryan and Nicola 1976, Behnke 1980) revealed that the meristic (relating to number and relation of body parts) and morphometric (relating to measurement of external form) characters for Paiute cutthroat trout are also typical of those characterizing Lahontan cutthroat trout. In recognition of the similarity of Paiute cutthroat trout

Figure 2. Historic (blue) and currently occupied (red) habitat for Paiute cutthroat trout in Silver King Creek, Humboldt-Toiyabe National Forest, Alpine County, California.

and other cutthroat subspecies, Vestal (1947) relegated the Paiute cutthroat trout to a subspecies of *Salmo clarki*. Miller (1950) and Shapovalov and Dill (1950) accepted this reclassification and it was recognized as *Salmo clarki seleniris*. All western North American trout have been reclassified from the genus *Salmo* to the genus *Oncorhynchus*, as summarized by Smith and Stearly (1989) and adopted by the American Fisheries Society's Committee on Names of Fishes (Robins *et al.* 1991).

Behnke and Zarn (1976) concluded, on the basis of gillraker comparisons, that the separation of Paiute cutthroat from Lahontan cutthroat occurred relatively recently (no more than 5,000 to 8,000 years ago), following the desiccation of Lake Lahontan. Paiute cutthroat trout and Lahontan cutthroat trout both typically possess 150 to 180 lateral series scales, 60 to 63 total vertebrae, 50 to 70 pyloric caeca (finger-like projections of the small intestine), and 21 to 27 gill rakers (bony projections from the gill arches). In the past, it was not possible to distinguish between the two subspecies on the basis of electrophoretic analytical techniques (Busack and Gall 1981). However, development of diagnostic DNA microsatellite markers may provide discrimination in the future (B. May, University of California, Davis, California, pers. comm. 2001).

Body spotting is the primary diagnostic character distinguishing the Paiute cutthroat trout from the Lahontan cutthroat trout. Paiute cutthroat trout have been known to have up to 9 body spots, but rarely more than 5, whereas Lahontan cutthroat trout typically possess 50 to 100 body spots and may have more. A secondary, but unquantifiable, distinguishing character is body coloration. Paiute cutthroat trout are typically coppery to purplish-pink, whereas Lahontan cutthroat trout from comparable stream environments are normally silver-yellow to light green.

C. Associated Candidate Species

In addition to Paiute cutthroat trout, two amphibian species that are candidates for listing, the mountain yellow-legged frog (*Rana muscosa*) and Yosemite toad (*Bufo canorus*), are known to occur in the Silver King Creek drainage.

1. Sierra Nevada Population of Mountain Yellow-legged Frog

On October 12, 2000, we published a 90-day finding for a petition to list the Sierra Nevada population of the mountain yellow-legged frog under the Endangered Species Act (U.S. Fish and Wildlife Service 2000a). We found the petition to have substantial evidence that listing the species as endangered may be warranted. We subsequently prepared a 12-month finding on the petition to list the Sierra Nevada population of the mountain yellow-legged frog. This finding was published in the Federal Register on January 16, 2003 (U.S. Fish and Wildlife Service 2003). We found that proposing to list this population was warranted but precluded by higher priority listing actions, and the population is now considered a candidate for listing. The southern California population of the mountain yellow-legged frog, which is currently listed as endangered, does not occur within the range of the Paiute cutthroat trout.

The mountain yellow-legged frog is a member of the family Ranidae (true frogs). It is a medium-sized frog with adults reaching 50 to 80 millimeters (2.0 to 3.1 inches) in length. The species attains lengths of 67 millimeters (2.6 inches) in males and 80 millimeters (3.1 inches) in females (Zweifel 1955, 1968). Their undersides range from a cream color to brilliant yellow. Dorsal coloration varies from drab olive to dark brown, with patterns ranging from discrete dark spots that can be few and large, to smaller and more numerous spots with a mixture of size and shapes. Tadpoles reach up to 76 millimeters (3.0 inches) in size and take from 2 to 4 years to metamorphose. Male frogs can smell strongly of garlic during the breeding season. The call of the male frogs is rarely heard because they vocalize while underwater.

Within the Silver King Creek drainage, mountain yellow-legged frogs have been observed along the mainstem in Upper Fish Valley, the artificial channel in Upper Fish Valley, the lower portion of Fly Valley Creek, and at Whitecliff Lake. As recently as 1993, several thousand mountain yellow-legged frogs were observed in the Silver King Creek drainage along the shores of Whitecliff Lake (P. Shanley, U.S. Forest Service, pers. comm. 2000). Prior to 2001, mountain yellow-legged frog occurrence information was primarily gathered during fish survey or management activities. In the summer of 2001, the

California Department of Fish and Game conducted a drainage-wide survey for amphibians. No adult mountain yellow-legged frogs were observed at Whitecliff Lake or other areas within the Silver King Creek drainage. However, two mountain yellow-legged frog tadpoles were observed in an artificial channel created as rearing habitat for Paiute cutthroat trout in Upper Fish Valley. In 2002, three adult mountain yellow-legged frogs were observed above Llewellyn Falls in the course of Paiute cutthroat trout surveys.

Chango and Wolf Creek Lakes, south of the Silver King Creek drainage in the West Walker River drainage, historically supported mountain yellow-legged frogs. Chango Lake is approximately 4.0 kilometers (2.5 miles) from upper Silver King Creek. Wolf Creek Lake is approximately 4.8 kilometers (3.0 miles) from upper Silver King Creek. In 1999, approximately 200 adult and 300 larval frogs were seen at Chango Lake (P. Shanley, pers. comm. 2000). An early survey in 2001 at Chango Lake yielded no mountain yellow-legged frogs. However, in a follow-up late-season survey, a total of 3 adults and 95 tadpoles were observed (D. Becker, California Department of Fish and Game, pers. comm. 2001). The population in Wolf Creek Lake is believed to be extirpated.

A conservation assessment and strategy program has been initiated for the mountain yellow-legged frog. A draft assessment has been prepared by the U.S. Forest Service, in cooperation with State and Federal agencies, universities, and research scientists, but has not yet been finalized. This conservation assessment will synthesize the best available information, including life history, habitat association, and risk factors and identify occupied and unoccupied habitats essential for the conservation of the species (U.S. Forest Service 2001).

2. Yosemite Toad

On October 12, 2000, we published a 90-day finding for the petition to list the Yosemite toad (U.S. Fish and Wildlife Service 2000b). We found the petition to have substantial evidence that listing the species as endangered may be warranted. Our 12-month finding on the petition to list the Yosemite toad was published in the Federal Register on December 10, 2002 (U.S. Fish and Wildlife Service 2002). We found that proposing to list the Yosemite toad was warranted, but precluded by higher priority listing actions; the species is now considered a candidate for listing.

The Yosemite toad is a high elevation species that occurs in the central Sierra Nevada Range (Stebbins 1966). Within the Silver King Creek drainage, the range of the Yosemite toad and western toad (*Bufo boreas*) overlap, and some degree of hybridization is suspected to occur. The Yosemite toad is a close relative of three toad species, the western toad, black toad (*B. exsul*), and Amargosa toad *(B. nelsoni*) (Blair 1972, Stebbins 1966). Yosemite/western toad hybridization occurs in the northern portion of the Yosemite toad's range in the Blue Lake region of the Carson-Iceberg Wilderness, just southeast of Carson Pass in Alpine County (Karlstrom 1962, Stebbins 1966). The Yosemite toad is a small to medium-sized toad with no head crests and large, flat circular parotoid glands (warty poison glands on the head) that are slightly separated (Karlstrom 1962). Yosemite toads show a high degree of sexual dimorphism (differing appearance of males and females). Females are larger and darker colored, with irregular dark blotches bordered with white, and males are smaller and speckled with black spots on a dull yellow to olive-greenish background and without distinct dark patches on their back (Karlstrom 1962).

A California Department of Fish and Game summer amphibian survey in 2001 documented occurrence of Yosemite toads, western toads, and hybrid Yosemite/western toads in the Silver King Creek drainage. Yosemite toads have also been observed in Silver Creek Meadows, which is situated below Chango Lake, in the West Walker River drainage. No quantitative surveys have been conducted to assess population size in the Silver King drainage. Additionally, the Sierra National Forest has been conducting surveys for Yosemite toads for the past decade. Yosemite toads have been noted in the Stairway Creek drainage in 1996, 2000, and 2001, and at Sharktooth Lake in 1999 (P. Strand, Sierra National Forest, pers. comm. 2002). A conservation assessment that is similar to efforts by the U.S. Forest Service for the mountain yellow-legged frog will also be undertaken for the Yosemite toad.

Other than recent surveys, no specific conservation actions directed towards the mountain yellow-legged frog and Yosemite toad in the Silver King

Creek drainage have been completed. However, several measures including livestock grazing closures and other habitat improvement projects have likely benefitted the mountain yellow-legged frog and Yosemite toad. Habitat improvements to the artificial channel in Upper Fish Valley have been a benefit to both amphibians. The chemical treatment of Bull Canyon Creek above the falls to Whitecliff Lake and the cessation of stocking in Tamarack Lake have reduced the impacts associated with introduced trout.

Prior to treatment to remove introgressed fish below Llewellyn Falls, amphibian surveys will be conducted on lower Silver King Creek, Tamarack Lake, Tamarack Creek, and other tributaries entering into the mainstem in that reach. All amphibians captured in surveys will be relocated during the treatments. There may be some negative impacts on amphibians if they are not captured during the relocation process or through stress of handling. However, the long-term effects of removal of nonnative and hybrid fish will be beneficial to native amphibians.

Whitecliff Lake, Tamarack Lake, and their outflows will be maintained as fishless waters. Amphibian populations will be monitored annually and biological and ecological data will be gathered. An evaluation is expected to be completed annually following the treatment to determine whether recolonization is occurring naturally or if the reintroduction from adjacent amphibian populations is necessary.

Recommendations from the range-wide conservation assessment and strategy efforts will be incorporated into management activities within the Silver King Creek drainage. These two amphibian species also co-occur with the four out-of-basin populations of Paiute cutthroat trout (North Fork Cottonwood, Stairway, Sharktooth, and Cabin Creeks), and conservation efforts will also be undertaken at these locations.

D. Life History and Habitat Requirements

Few studies have been completed on the biology of the Paiute cutthroat trout. Most of what is known is based on studies conducted by Wong (1975) and

Diana (1975) on the introduced population in the North Fork of Cottonwood Creek, Mono County, California. Its life history and habitat requirements appear to be similar to those reported for other western stream-dwelling salmonids. All life stages require cool, well-oxygenated waters. Adult fish prefer stream pool habitat in low gradient meadows with undercut or overhanging banks and abundant riparian vegetation (Behnke and Zarn 1976). Pools are important rearing habitat for juveniles and act as refuge areas during winter (Raleigh *et al.* 1984; Swales *et al.* 1986; Berg 1994). During the winter months, trout move into pools to avoid physical damage from ice scouring (Hartman 1965; Scrimgeour *et al.* 1994) and to conserve energy (Everest and Chapman 1972; Cunjak 1996). As with other salmonids, suitable winter habitat may be more restrictive than summer habitat (Jakober *et al.* 1998). Paiute cutthroat trout survive in lakes, but there is no evidence that they ever occurred naturally in any lakes within the Silver King basin. Paiute cutthroat trout demonstrate fluvial spawning behavior and must have access to flowing waters with clean gravel substrates.

Paiute cutthroat trout reach sexual maturity at the age of 2 years. Peak spawning activity occurs in June and July (Wong 1975). The eggs hatch in 6 to 8 weeks and the fry emerge from the gravel in another 2 to 3 weeks. Young-of-the-year fish rear in mainstem shoals or backwaters, and often move into intermittent tributary streams until they reach about 50 millimeters (2.0 inches) in length (Diana and Lane 1978; W. Somer, California Department of Fish and Game, pers. comm. 2001).

Paiute cutthroat trout are opportunistic feeders, utilizing whatever aquatic and terrestrial invertebrates occur in the drift. They set up dominance hierarchies and defend these positions (Wong 1975). The largest fish typically occupy pools, while the smaller fish utilize runs and riffles and whatever other unoccupied habitats are available. Growth rates vary with water temperature and the abundance of food organisms. In stream environments Paiute cutthroat trout seldom reach sizes in excess of 250 millimeters (10 inches) total length (Moyle 1976). They attain a maximum size of 342 millimeters (13.5 inches) in Silver King Creek (W. Somer, pers. comm. 2002). In lakes they may grow to 450 millimeters (18 inches) or more (Ryan and Nicola 1976).

Paiute cutthroat trout eggs and fry have several natural predators -- water shrews (*Sorex palustris*), dippers (*Cinclus mexicanus)*, and trichopteron larvae -- but adult fish have few predators. Disease is apparently a significant cause of adult mortality, particularly in the post-spawning period. Wong (1975) observed extensive fungal infections on the dorsal and caudal fins of several spawned-out fish in the North Fork of Cottonwood Creek. Many of these fish were so weakened by spawning they were unable to recover. This fungal infection has never been observed outside of North Fork of Cottonwood Creek. Few Paiute cutthroat trout apparently live beyond the age of 3 years in a wild stream environment (Wong 1975).

Paiute cutthroat trout are less wary than other trouts, presumably because they evolved in a high mountain environment where terrestrial and avian predators are not frequently encountered (Moyle 1976). Their unwariness makes them highly vulnerable to angling. Significant population declines have been noted in waters that are exposed to moderate or even light fishing pressure (MacPhee 1966; Behnke 1980).

E. Distribution

The presumed historic distribution of the Paiute cutthroat trout is limited to 14.7 kilometers (9.1 miles) of habitat, in Silver King Creek (from Llewellyn Falls downstream to Silver King Canyon) as well as the accessible reaches of three small named tributaries: Tamarack Creek, Tamarack Lake Creek, and the lower reaches of Coyote Valley Creek downstream of barrier falls (Figure 2). This watershed is entirely within the boundaries of the Humboldt-Toiyabe National Forest. The issue of what constitutes the native range is complicated by the paucity of early collection records and the conflicting recollections of early observers. The situation is further complicated by one or more unofficial transplants, and by natural events that may have altered the course of Silver King Creek. The account presented here is based on the conclusions of Ryan and Nicola (1976) and supported by Behnke (1980).

A barrier or series of barriers that developed in the Silver King Canyon during the last 10,000 years led to the isolation of Paiute cutthroat trout from

Lahontan cutthroat trout. Connell and others reported that a high falls exists on lower Silver King Creek a short distance upstream from its confluence with Snodgrass Creek (Ashley 1970). A 1994 California Department of Fish and Game survey identified six potential fish barriers in the Silver King Canyon, the two highest being 2.44 meters (8 feet) and 3.05 meters (10 feet) in two separate channels.

Steep barrier falls exist at several locations on the mainstem and tributaries of Silver King Creek. The locations of all known fish barriers in the Silver King Creek drainage are shown in Figure 2. Llewellyn Falls is assumed to have been a historic barrier to upstream fish movements in Silver King Creek on the basis of Virgil Connell's observations and recollections. Connell, an early grazing permittee in the basin, reported that there were no fish above Llewellyn Falls in the early 1890's (V. Connell, letter in Ryan and Nicola 1976). In 1912, Joe Jaunsaras, a herdsman employed by Connell, caught some fish below Llewellyn Falls and transplanted them into Silver King Creek above the falls (V. Connell, letter in Ryan and Nicola 1976). According to Connell these (unspotted) fish increased in numbers above the falls ". . . until in 1924 the stream was so well stocked, that fishing above the falls was better than below." Connell also noticed that sometime during this period the fish below the falls became " . . . mixed with other kinds, probably due to the stocking on the lower stream of different varieties."

An alternative scenario for the introduction of Paiute cutthroat trout into upper Silver King Creek is presented by Ashley (1970). He concluded, on the basis of conversations with a herdsman, that the 1912 transplant was a failure and that the population above Llewellyn Falls became established as the result of an introduction in 1924. John Jaunsaras, the brother of the herdsman who made the 1912 transplant, reported that he and another man carried 75 5-gallon buckets of trout upstream around the falls. The fish reportedly originated from a small tributary of Silver King Creek that entered the mainstem just below Llewellyn Falls. Ryan and Nicola (1976) rejected this explanation because large numbers of fish were reported to be present above Llewellyn Falls by Connell in 1924, and because the purported donor population below Llewellyn Falls may already have become introgressed by 1924. There is no evidence to suggest that the population above Llewellyn Falls became introgressed anytime before 1949.

The means by which rainbow trout and Lahontan cutthroat trout gained access to historic Paiute cutthroat trout habitat, and the date on which it first occurred, are not known. It may have happened in the mid-1920's as the result of a flood that changed the course of Silver King Creek. Ashley (1970) accepted Connell's account of a severe cloudburst in the Silver King Creek drainage in 1927, and concluded that the resultant flood altered the course of Silver King Creek near its confluence with Snodgrass Creek and eliminated a historic waterfall. Alternatively, rainbow trout and Lahontan cutthroat trout may have been introduced by early ranchers or anglers.

By 1933 when Snyder made his collections in Silver King Creek, the population below Llewellyn Falls consisted of heavily spotted fish, and the population above Llewellyn Falls was made up of fish without any, or with only a small number of, body spots. Of the 79 specimens of Paiute cutthroat trout collected by Snyder from above Llewellyn Falls in 1933, 47 had no body spots and the remaining 32 had from 1 to 9 body spots (S. Nicola, pers. comm. in U.S. Fish and Wildlife Service 1985).

It is not known if Paiute cutthroat trout are native to Corral Valley Creek and its tributary Coyote Valley Creek (Figure 2). Falls near the mouth of Corral Valley Creek are assumed to have been a historic fish barrier. However, there are no records to confirm that this tributary was originally barren of fish. Ashley (1970) reported that both Corral Valley and Coyote Valley Creeks contained Paiute cutthroat trout when Connell first visited the area in 1889. Connell believed their presence was due to the activities of French-Canadian loggers who were working in the area in the 1860's (Ashley 1970). Vestal (1947) made the first documented collections from these two streams in 1946, and believed that the streams were ". . . formerly barren of fish life." He attributed their presence to the activities of sheepmen who ". . . reportedly planted Piute (sic) trout a few at a time in buckets from Upper Fish Valley."

Sometime after 1950, Paiute cutthroat trout in Silver King Creek above Llewellyn Falls became introgressed as the result of introductions of rainbow and Lahontan cutthroat trout into the upper watershed by the California Department of Fish and Game. Planting records indicate that 5,040 rainbow trout fry were stocked above Llewellyn Falls during September 1949. It is unclear when or where Lahontan cutthroat trout were stocked above Llewellyn Falls. The populations in Corral Valley and Coyote Valley Creeks also became introgressed sometime during the 1950's from an unknown source.

Efforts to restore pure populations of Paiute cutthroat trout above Llewellyn Falls appear to have been successful following multiple chemical treatments, combined with removal of hybridized trout using electrofishing. A 3-year chemical treatment project conducted during 1991 through 1993 successfully removed hybrid trout from Silver King Creek in Upper Fish Valley upstream from Llewellyn Falls. The population of Paiute cutthroat trout in Fly Valley Creek has remained isolated by a barrier falls. Hybridized trout have been removed from Four Mile Canyon Creek by electrofishing and chemical treatment during 1991 through 1993. Corral Valley Creek was chemically treated during 1964, and retreated during 1977 to remove hybridized trout. Electrofishing surveys following the 1977 treatment eliminated surviving hybridized trout. The chemical treatments of Coyote Valley Creek during 1964 and 1977 failed, however, retreatment during 1987 and 1988 appears successful because no hybrid trout have been observed during subsequent electrofishing surveys. These results have been reconfirmed by allozyme and nuclear DNA analysis of tissue samples from all populations (Israel *et al.* 2002).

In summary, available evidence suggests that the native range of the Paiute cutthroat trout is limited to the reach of Silver King Creek between Llewellyn Falls and a presumed historic barrier in Silver King Canyon, and all accessible tributaries within this reach. This range constitutes about 14.7 kilometers (9.1 miles) of stream habitat. It is also possible that Paiute cutthroat trout are native to Corral Valley and Coyote Valley Creeks, but that will probably remain a matter of conjecture because there are no collection records available from these streams to document their faunal composition before they were influenced by man. For this reason, there is also a slight possibility that Connell's account of the situation is incorrect and that the true native range of the Paiute cutthroat trout is Silver King Creek above Llewellyn Falls.

Following Snyder's discovery and description, the California Department of Fish and Game made several attempts to transplant Paiute cutthroat trout into other waters. The first documented introduction was made in 1937 into upper and lower Leland Lakes. That transplant failed, but another effort was made in 1946 when they were introduced into the North Fork of Cottonwood Creek. Progeny of that transplant survive to the present. A list of known transplant attempts is shown in Table 1. The present distribution of Paiute cutthroat trout consists of a population in Silver King Creek above Llewellyn Falls and tributary populations in Fly Valley, Four Mile Canyon Creek, Coyote Valley, and Corral Valley Creeks (Figure 2), and four self-sustaining, pure populations outside the native drainage in the North Fork of Cottonwood and Cabin Creeks (Figure 3), and Stairway and Sharktooth Creeks (Figure 4). The introduced population in Delaney Creek, Yosemite National Park, Tuolumne County, introduced in 1968, is suspected to be extirpated due to the presence of brook trout (*Salvelinus fontinalis*). The only known self-sustaining lake population in Birchim Lake (Inyo National Forest, Inyo County) was confirmed to be introgresssed with rainbow trout in 1984 (D. Wong, California Department of Fish and Game, pers. comm. 2000).

F. Abundance

1. Silver King Creek Drainage

Paiute cutthroat trout now occupy a minimum of 33.2 kilometers (20.6 miles) of stream habitat in five widely separated drainages. Populations in the Silver King Creek drainage occupy about 18.6 kilometers (11.5 miles) of stream habitat, including 12.9 kilometers (8 miles) of good quality habitat that supports on average 1,020 adult fish (> 150 millimeters [6 inches]) in 6 stream populations. Paiute cutthroat trout occupy approximately 4.3 kilometers (2.7 miles) in Silver King Creek above Llewellyn Falls. Results from the 2001 population survey in Upper Fish Valley were within the range of its historical population abundance,

Table 1. Recorded transplants of Paiute cutthroat trout.

Water Location	Year	Source	Number	Status
Lower and Upper Leland Lakes (El Dorado Co., CA)	1937	Silver King Cr.	400	Disappeared by 1941.
North Fork of Cottonwood Cr. (Mono Co., CA)	1946	Silver King Cr. Coyote Valley Cr. Corral Valley Cr.	125 249 27	Reproducing population established.
McGee Cr. (Mono Co., CA)	1956	North Fork of Cottonwood Cr.	?	Unsuccessful.
Bull Lake (Alpine Co., CA)	1957	Silver King Cr.	46	Unsuccessful.
Birchim Lake (Inyo Co., CA)	1957	North Fork of Cottonwood Cr.	70	Highly Introgressed.
Delaney Cr. (Tuolumne Co., CA)	1966	Four Mile Canyon Cr. Fly Valley Cr.	40 3	Displaced by brook trout.
Sharktooth Lake (Fresno Co., CA)	1968	North Fork of Cottonwood Cr. Delaney Cr.	23 6	Population established in outflow (Sharktooth Creek).
Cabin Cr. (Mono Co., CA)	1968	North Fork of Cottonwood Cr.	60	Small reproducing population established.
Stairway Creek (Madera Co., CA)	1972	Delaney Cr.	77	Reproducing population established.
Heenan Lake (Alpine Co.. CA)	1983	Coyote Valley Cr.	170	Unsuccessful.

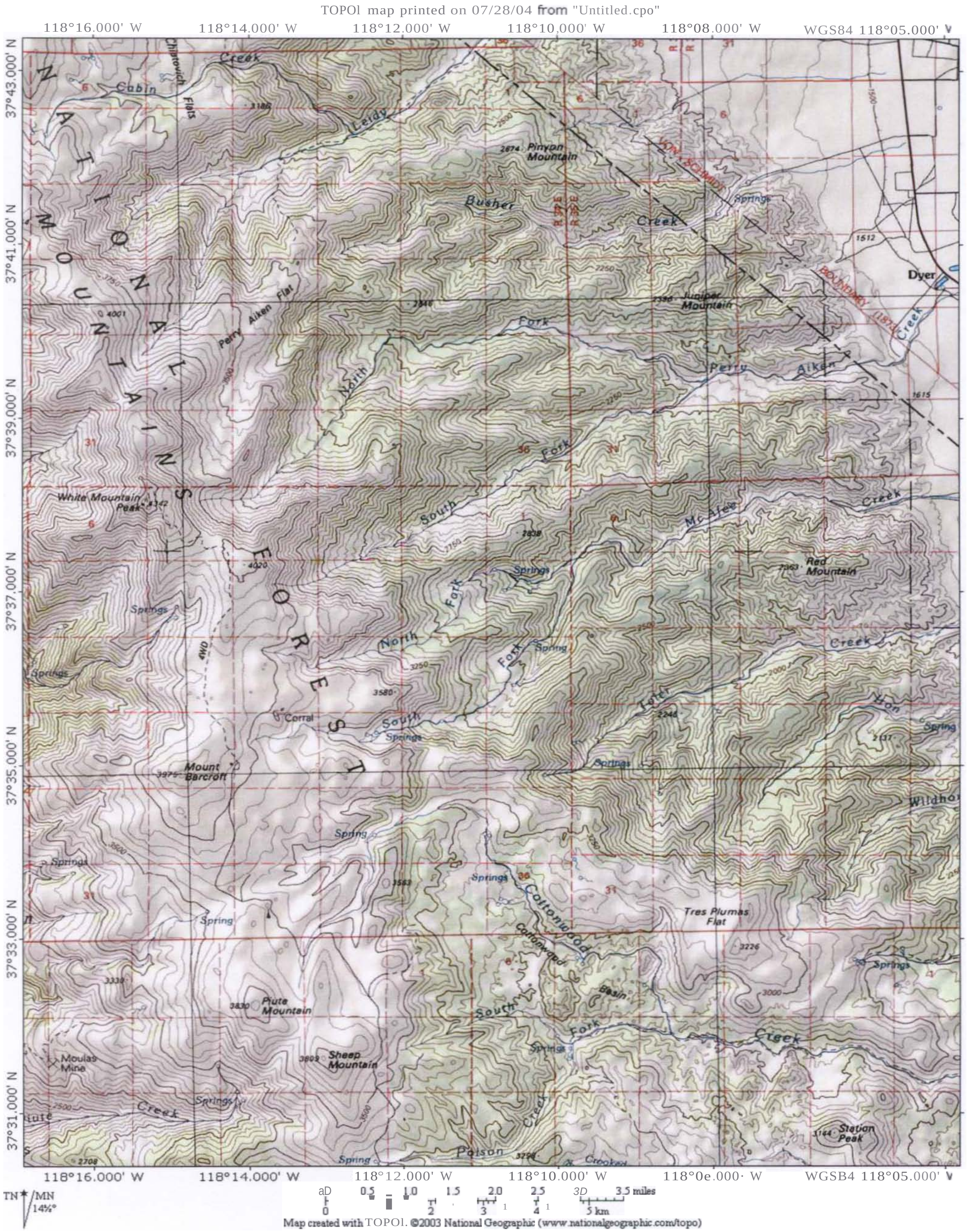

Figure 3. Refugial populations of Paiute cutthroat trout in North Fork Cottonwood Creek and Cabin Creek, Inyo National Forest, Mono County, California.

Figure 4. Refugial populations of Paiute cutthroat trout in the Sierra National Forest, in Stairway Creek, Madera County, and Sharktooth Creek, Fresno County, California.

suggesting that the population may still be expanding (Figure 5). A total of 217 adult trout were observed during the snorkel and electrofishing surveys in 2001. Based on population estimates that compare multiple-pass electrofished test sections, the population could consist of as many as 424 adult fish, which is the average number of adults for this 1,900-meter (1.2-mile) reach. Figures 4 through 8 show how variable these populations can be as well as how quickly Paiute cutthroat trout rebound from chemical treatments and natural disturbance.

Twenty population estimate surveys have been conducted on Four Mile Canyon Creek. The first was in 1968, and they have been conducted nearly every year since 1984. Figure 6 shows the results from those surveys. In 2000, California Department of Fish and Game surveyed 250 meters (820 feet) of stream and estimated 78 adult fish per kilometer (126 per mile), which is lower than the average of 133 adult fish per kilometer (215 per mile). Adult numbers have stayed relatively constant while juvenile numbers have fluctuated widely. Paiute cutthroat trout occupy approximately 3 kilometers (1.9 miles) of habitat in Four Mile Canyon Creek.

Seven population estimate surveys have been conducted on Fly Valley Creek. The first survey was in 1984 and the last was in 2000 (Figure 7). In 2000, California Department of Fish and Game surveyed 150 meters (492 feet) of stream and estimated 118 adult fish per kilometer (190 per mile), which is lower than the average of 221 adult fish per kilometer (356 per mile). While juvenile numbers have historically fluctuated, adult numbers have stayed relatively constant. Paiute cutthroat trout occupy approximately 1.8 kilometers (1.1 miles) of habitat in Fly Valley Creek.

Eight population estimate surveys have been conducted on Corral Valley Creek. The first survey was in 1974 and the last was in 2000 (Figure 8). In 2000, California Department of Fish and Game surveyed a 150-meter (492-foot) section and estimated 59 adult fish per kilometer (95 per mile), which is lower than the average of 148 adult fish per kilometer (238 per mile). It is unclear why the population decreased in 2000, but this decrease is most likely due to natural fluctuations in the population. Paiute cutthroat trout occupy approximately 3.6 kilometers (2.2 miles) of habitat in Corral Valley Creek.

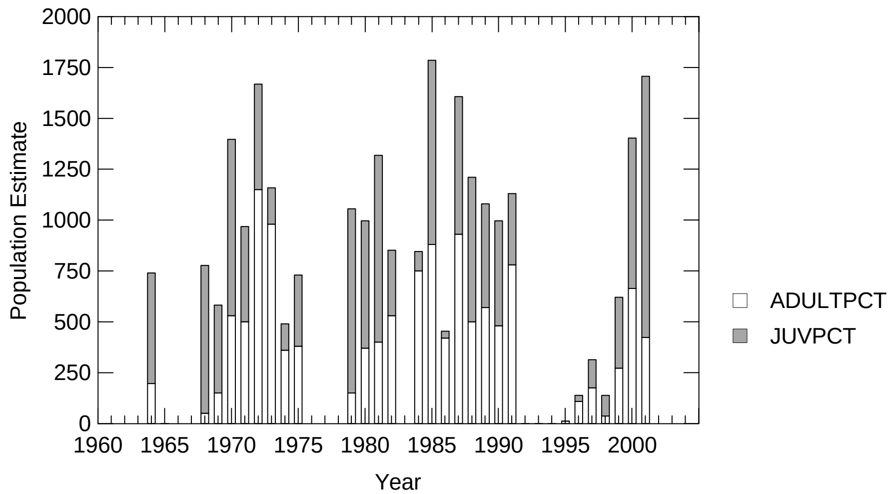

Figure 5. Historical population estimates (1964 to 2001) from the Upper Fish Valley reach of Silver King Creek. The white bars represent adult Paiute cutthroat trout (over 150 millimeters [6 inches]) and the dark bars represent juvenile Paiute cutthroat trout (under 150 millimeters [6 inches]). Upper Fish Valley was treated with rotenone in 1964, 1976, and 1991 to 1993. The Silver King Creek drainage experienced heavy runoff in 1982, 1986, and 1998. (W. Somer, California Department of Fish and Game, unpubl. data).

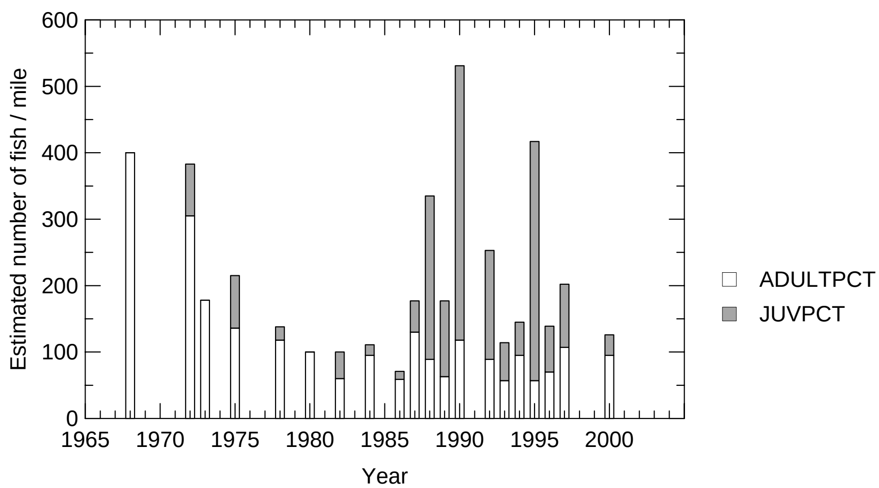

Figure 6. Historical population estimates (1968 to 2000) from Four Mile Canyon Creek in the Silver King Creek drainage. The white bars represent adult Paiute cutthroat trout (over 150 millimeters [6 inches]) and the dark bars represent juvenile Paiute cutthroat trout (under 150 millimeters [6 inches]). In 1968, 1973, and 1980 population estimates represent both adult and juvenile fish. Four Mile Canyon Creek was treated with rotenone from 1991 to 1993. The Silver King Creek drainage experienced heavy runoff in 1982, 1986, and 1998. (W. Somer, California Department of Fish and Game, unpubl. data).

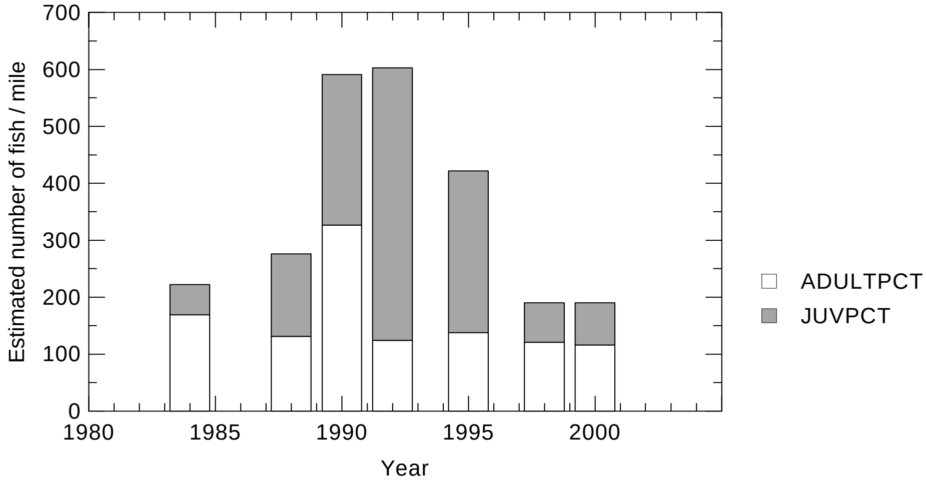

Figure 7. Historical population estimates (1984 to 2000) from Fly Valley Creek in the Silver King Creek drainage. The white bars represent adult Paiute cutthroat trout (over 150 millimeters [6 inches]) and the dark bars represent juvenile Paiute cutthroat trout (under 150 millimeters)[6 inches]. Fly Valley Creek has never been treated with rotenone. The Silver King Creek drainage experienced heavy runoff in 1982, 1986, and 1998. (W. Somer, California Department of Fish and Game, unpubl. data).

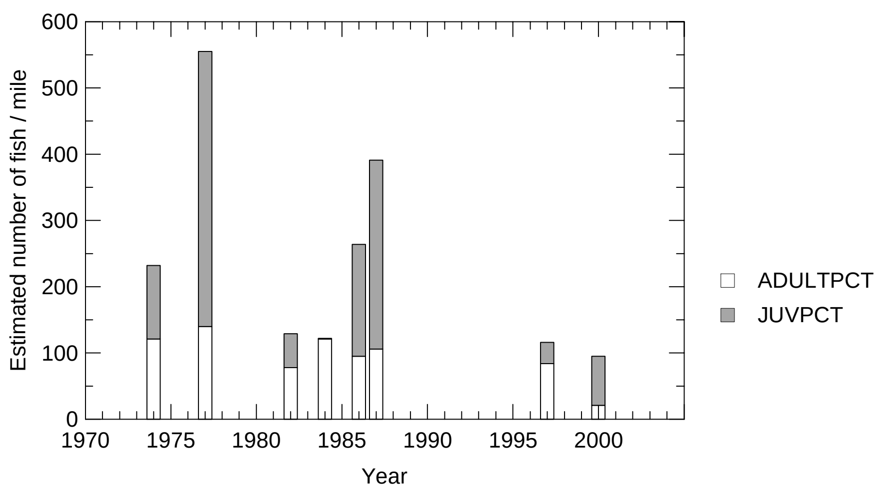

Figure 8. Historical population estimates (1974 to 2000) from Corral Valley Creek in the Silver King Creek drainage. The white bars represent adult Paiute cutthroat trout (over 150 millimeters [6 inches]) and the dark bars represent juvenile Paiute cutthroat trout (under 150 millimeters [6 inches]). Corral Valley Creek was treated with rotenone in 1964 and 1977. The Silver King Creek drainage experienced heavy runoff in 1982, 1986, and 1998. (W. Somer, California Department of Fish and Game, unpubl. data).

Population estimates on Coyote Valley Creek were sporadically conducted from 1964 to 2000 (Figure 9). Two separate 150-meter (492-foot) sections, Upper Meadow and Lower Meadow, were surveyed. In 2000, California Department of Fish and Game estimated 508 adult fish per kilometer (819 per mile) for the Upper Meadow section, which is slightly lower than the average of 528 adult fish per kilometer (852 per mile). The Lower Meadow section had an estimated 589 adult fish per kilometer (950 per mile), which is higher than the average of 444 adult fish per kilometer (716 per mile). Paiute cutthroat trout occupy approximately 4.9 kilometers (3 miles) of habitat in Coyote Valley Creek.

2. North Fork of Cottonwood Creek

Occupied habitat for Paiute cutthroat trout in the North Fork of Cottonwood Creek is limited to the uppermost 5.5 kilometers (3.4 miles) of stream above the Tres Plumas barrier. In 1946, 401 Paiute cutthroat trout from the Silver King Creek drainage (Table 1) were stocked. A standard section of stream, from Granite Meadow downstream to a standard point just above the Tres Plumas barrier, has been surveyed visually since 1989 by the California Department of Fish and Game (Figure 10). The exclusion of grazing since 1993 and spawning enhancement projects in 1995 and 1996, which created 51 spawning sites, appear to have increased Paiute cutthroat trout numbers (D. Becker, unpubl. data).

3. Cabin Creek

Cabin Creek was originally stocked in 1968 with 60 individuals from the North Fork of Cottonwood Creek. Occupied habitat for Paiute cutthroat trout in Cabin Creek is approximately 2.4 kilometers (1.5 miles). Visual surveys were conducted on Cabin Creek in 1995 and 2000 (D. Becker, California Department of Fish and Game, unpubl. data). In 1995, 139 fish were observed and were broken down into size classes. Thirty-eight fish were between 100 and 200 millimeters (4 and 8 inches). The remaining 101 fish were between 200 to 254 millimeters (8 to 10 inches). In 2000, 186 fish were observed. This survey did not break down individual sizes, although multiple size classes were present.

A

B

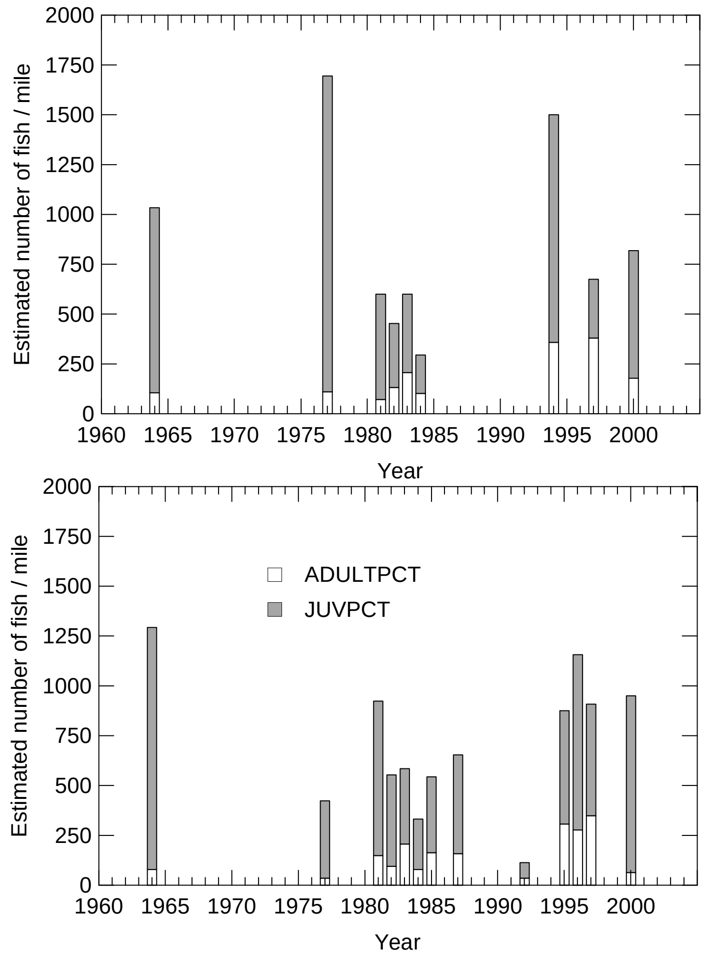

Figure 9. Historical population estimates (1984 to 2000) from Coyote Valley Creek in the Silver King Creek drainage. Figure A represents the Upper Meadow section and figure B represents the Lower Meadow section. The white bars represent adult Paiute cutthroat trout (over 150 millimeters [6 inches]) and the dark bars represent juvenile Paiute cutthroat trout (under 150 millimeters [6 inches]). Coyote Valley Creek was treated with rotenone in 1964, 1977, and 1987 to1988. The Silver King Creek drainage experienced heavy runoff in 1982, 1986, and 1998. (W. Somer, California Department of Fish and Game, unpubl. data).

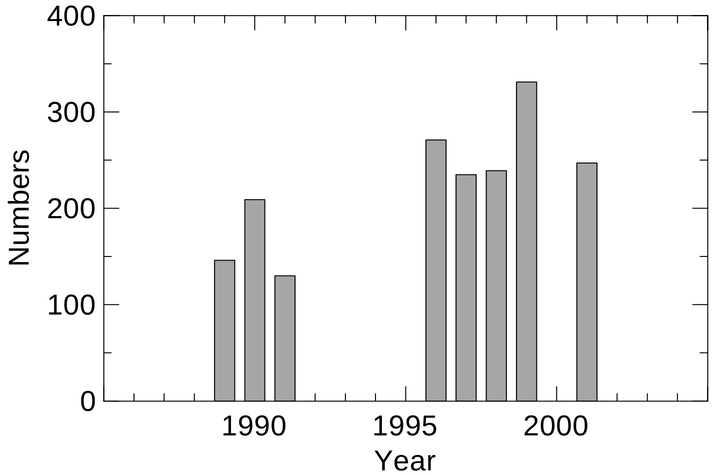

Figure 10. Visual observations from the North Fork of Cottonwood Creek, Inyo National Forest, since 1989. The numbers include all size classes observed (D. Becker, California Department of Fish and Game, unpubl. data).

4. Stairway Creek

The population in Stairway Creek occupies approximately 3.5 kilometers (2 miles) of stream habitat. Strand and Eddinger (1999) provide a summary of historic population estimates in Stairway Creek. In 1972, 77 individuals from Delaney Creek were stocked into Stairway Creek. Population surveys in Stairway Creek using electrofishing methods occurred in 1974 through 1977 and 1981. In 1974, surveys located 5 adults and in 1975, 12 individuals were located (6 adults and 6 juveniles). Surveys conducted in 1976 and 1977 showed a large increase in numbers found with 150 and 118 individuals respectively. In 1981, a more thorough survey was conducted, which estimated the population at 36.6 individuals per 100 meters (590 per mile) (excluding young of year) with 76 percent of the population estimated as adults (greater than 127 millimeters [5 inches]). In 1996, the Sierra National Forest conducted visual observations of Paiute cutthroat trout in each habitat by life stage on 2.5 kilometers (1.5 miles) of stream. Strand and Eddinger (1999) reported seeing 22.7 individuals per 100 meters (366 per mile) with an estimated 70 percent of the population being adults (greater than 127 millimeters [5 inches]). Comparison of population estimates between years is not statistically reliable since different methods were used and different lengths of stream were surveyed. A rain on snow event that occurred in 1997 resulted in down-cutting of the stream channel, reduced habitat complexity, and fewer fish during the 2000 survey (P. Strand, pers. comm 2002). However, the fish that were observed appeared more robust. Because of past mortality rates from electrofishing salmonids on the Sierra National Forest, fly rod depletion (Stephens and Christenson 1980) was selected as a means to estimate the number of fish per pool during the 2000 survey. Thirty pools were sampled with an average of 4.3 individuals per pool (P. Strand, unpubl. data). The fly rod depletion method is not intended to be statistically reliable and is biased towards larger fish; however, it can be used to determine the minimum number of fish per pool.

5. Sharktooth Creek

Strand and Eddinger (1999) also provided a summary of historic population estimates in Sharktooth Creek. In 1968, 29 individuals, 6 from Delaney Creek and 23 from North Fork of Cottonwood Creek, were stocked into Sharktooth Lake. In 1970, a 4-hour angling survey conducted in the lake resulted in no fish taken. In 1973, visual surveys of the lake and outlet stream (Sharktooth Creek) resulted in no observations. In 1975, personnel of the California Department of Fish and Game noted several Paiute cutthroat trout in the outlet stream. The next survey was conducted in 1999 by Sierra National Forest personnel. Fish from Sharktooth Lake evidently moved downstream into Sharktooth Creek and now occupy approximately 3.2 kilometers (2 miles) of stream from the outlet of Sharktooth Lake to the confluence with Lost Keys Lake outlet stream. Fly rod depletion and visual observation were selected as a means to estimate the number of fish per pool (Stephens and Christenson 1980). Twenty-five pools were sampled in the only low gradient section of occupied habitat. Fifty-eight individuals were caught or observed in the pools for an average of 2.32 fish per pool. The fly rod depletion method is not intended to be statistically reliable and is biased towards larger fish; however, it can be used to determine the minimum number of fish per pool.

G. Habitat Description

1. Silver King Creek Drainage

As part of the California Wilderness Act, 65,000 hectares (160,000 acres) were set aside in 1984 as the Carson-Iceberg Wilderness. This area is managed both by the Humboldt-Toiyabe and Stanislaus National Forests. The entire portion of the Silver King Creek drainage occurs within the Humboldt-Toiyabe National Forest. This description of habitat is based on the account presented by Ryan and Nicola (1976).

Silver King Creek is a tributary of the East Fork Carson River, which drains into the Lahontan Basin. The creek originates at 2,926 meters (9,600 feet) elevation in the southernmost portion of the drainage, and flows north through three distinct valleys for approximately 22.5 kilometers (14 miles) where it meets the East Fork Carson River. Between the headwaters and the confluence of Silver King Creek with the East Fork Carson River, eight tributaries, three above and five below Llewellyn Falls, join Silver King Creek. Llewellyn Falls, at an elevation of 2,438 meters (8,000 feet), is located at the head of Lower Fish Valley, some 16.2 kilometers (10 miles) above the confluence with the East Fork Carson River. The physical characteristics of Silver King Creek and its tributaries are described in Table 2.

From its source, Silver King Creek flows precipitously for 3.2 kilometers (2.0 miles) before beginning a gradual descent to Upper Fish Valley in an area of washed-out beaver ponds just above the mouth of Fly Valley Creek. For 2.4 kilometers (1.5 miles), through Upper Fish Valley, it is a typical meandering meadow creek, averaging 3.7 meters (12 feet) wide and 0.3 meter (1 foot) deep in the summer. Several soda springs occur in the valley, with some seeping directly into the stream. From the southeast, Four Mile Canyon Creek enters 2.0 kilometers (1.2 miles) above Llewellyn Falls, while Bull Canyon Creek joins the mainstem from the west 0.8 kilometer (0.5 mile) above Llewellyn Falls. In 1984, an abandoned stream channel was reconnected with the mainstem, providing 0.46 kilometers (0.3 miles) of spawning and juvenile rearing habitat. The upstream portion of the channel begins approximately 0.2 kilometer (0.1 mile) below the

Table 2. Physical characteristics of Silver King Creek and its principal tributaries. Modified from Ryan and Nicola (1976).[1]

Stream	Length (kilometers)	Occupied habitat (kilometers	Historic habitat (kilometers)	Drainage area (hectares)	Elevation (meters) max	min	Average gradient (percent)
Fly Valley	2	1.8	0	414.4	2,682	2,512	8.5
Four Mile Canyon	4.5	3.0	0	880.6	3,048	2,487	12.5
Bull Canyon	4	1.0	0	673.4	2,902	2,463	11.0
Tamarack Lake	2	0	0.3	181.3	2,835	2,423	20.6
Unnamed tributaries	2.3	0	0.9	51.8	2,877	2,414	23.3
Tamarack	4.8	0	3.4	932.4	2,804	2,365	9.1
Coyote Valley	8	4.9	0.5	1,217.3	3,048	2,377	8.4
Corral Valley	5.6	3.6	0	1,346.8	3,347	2,743	7.1
Snodgrass	3.6	0	0	854.7	2,438	2,088	9.7
Silver King (exclusive of tributaries)	22.5	4.3	9.6	5,335.4	2,865	1,951	4.1
Total	59.3	18.6	14.7	11,914			

[1] Distances, areas, and elevations measured from USGS topographic maps.

confluence of Silver King Creek and Four Mile Canyon Creek. The lower portion of the channel rejoins the mainstem immediately above the confluence of Silver King Creek and Bull Canyon Creek.

At the lower end of Upper Fish Valley, the stream gradient increases through a sparsely forested section before reaching Llewellyn Falls. The vertical drop of Llewellyn Falls is approximately 6.1 meters (20 feet). Within the 2.8-kilometer (1.7-mile) length of Lower Fish Valley, two small tributaries enter the mainstem from the west: Tamarack Lake Creek, located 1.2 kilometers (0.7 mile) below Llewellyn Falls, and a short, unnamed tributary downstream another 1.2 kilometers (0.7 mile). Long Valley, only 1.5 kilometers (0.9 mile) long, is the shortest of the three valleys. No tributaries enter this section of Silver King Creek. Between Lower Fish Valley and Long Valley the gradient increases, but no barriers similar to Llewellyn Falls are known to exist in this section. Below Long Valley, Tamarack Creek enters Silver King Creek from the west 0.6 kilometer (0.4 mile) below Long Valley, and Coyote Valley Creek enters from the east 1 kilometer (0.6 mile) farther downstream.

Approximately 2.8 kilometers (1.7 miles) below the mouth of Coyote Valley Creek, Silver King Creek descends through Silver King Canyon and emerges from the canyon in the vicinity of Snodgrass Creek. Upstream from Snodgrass Creek, in the canyon, a series of falls present a fish barrier to nonnative trout and nonsalmonid native fish species that occur downstream. No tributary of significance enters Silver King Creek from Snodgrass Creek downstream for 5.4 kilometers (3.4 miles) until its confluence with the East Fork Carson River. Three small lakes occur in the drainage: 1) Tamarack Lake, 2) Whitecliff Lake, and 3) an unnamed lake in the headwaters of Four Mile Canyon Creek. The average gradient of Silver King Creek is 4.1 percent, which is less than any of its tributaries. However, the portion of Silver King Creek between Fly Valley and Corral Valley Creeks, has an average gradient of 1.6 percent.

In 1984, 1987, and 1990, personnel from the California Department of Fish and Game, U. S. Fish and Wildlife Service, and the U.S. Forest Service along with volunteers from Trout Unlimited participated in interdisciplinary functional assistance trips to the Silver King Creek drainage to conduct physical habitat and biological field surveys (see Appendix A). The objectives of this effort were to provide the National Forest with a general assessment of habitat and to provide recommendations for future management. Habitat surveys were performed using the General Aquatic Wildlife System procedures (Duff *et al*. 1989). A Habitat Condition Index is obtained using the General Aquatic Wildlife System methodology which can then be used to provide habitat trend data. Nine stations were monitored on Silver King Creek above Llewellyn Falls, two stations on Bull Canyon Creek, one station on Fly Valley Creek, two stations on Four Mile Canyon Creek, four stations on Coyote Valley Creek, and two stations on Corral Valley Creek (Appendix A, Table A1 and Figures A1 and A2). The Habitat Condition Index over this 6 year period improved in nearly all of the stations monitored, which was primarily due to a change in grazing management (Table 3). However, even though most stations increased their Habitat Condition Index rating, 12 of the 21 stations still rated as fair to poor. No habitat monitoring has been done since 1990, nor has any habitat monitoring been done throughout the historic range of Paiute cutthroat trout from Llewellyn Falls downstream to Silver King Canyon.

Sediment samples were taken using a hollow core sampler during the functional assistance trips in 1984 and 1990. Five samples were taken in riffle areas at each station to determine how much fine sediment (particle sizes less than 6.35 millimeters [0.2 inches]) was present. Excess fine sediment is known to increase mortality of salmonid embryos (Chapman 1988; Bjornn and Reiser 1991) and could be a limiting factor in recruitment. Duff (1991) recommended that the minimum amount of fine sediment should not exceed 30 percent and that natural fine sediment amounts in Silver King Creek fluctuated between 20 and 30 percent. Results from this sampling effort revealed that the amount of fine sediment stayed constant between 1984 and 1990 (39.3 and 39.4 percent respectively) (Table A2). No sediment sampling has been done since grazing was stopped in 1994. The basin was logged in the 1860's, used as pasture for sheep in the early 1900's through the late 1930's, and used as pasture for cattle from the 1940's through 1994 (Overton *et al*. 1993; P. Shanley, pers. comm. 2000).

Table 3. Summary of Habitat Condition Index (HCI) ratings from 1984, 1987, and 1990. (Modified from Duff 1991).

Stream	Station	Channel Type	HCI 1984	HCI Rating 1984	HCI 1987	HCI Rating 1987	HCI 1990	HCI Rating 1990
Silver King	S1:610	C3	51.5	Poor	54.9	Poor	58.6	Poor
Silver King	S2:640	C3	65	Fair	55.3	Poor	84.2	Good
Silver King	S3:641	C3	64.8	Fair	54.6	Poor	78.8	Good
Silver King	S4:700	C3	38.5	Poor	37.9	Poor	68.4	Fair
Silver King	S5:725	C3	28.8	Poor	35.4	Poor	65.9	Fair
Silver King	S6:738	C3	48.3	Poor	54.6	Poor	69.7	Fair
Silver King	S6A:745	C3	58.5	Poor	66.7	Fair	70.4	Fair
Silver King	S7:775	B2/B3	63	Fair	63	Fair	69.7	Fair
Silver King	S8:813	C3	41.7	Poor	46.9	Poor	51	Poor
Bull Canyon	S1:040	C3	82.4	Good	83.7	Good	88.2	Excel.
Bull Canyon	S2:100	B2	54.3	Poor	57.8	Poor	69.4	Fair
Fly Valley	S1:500	B2/C2	84.4	Good	82.6	Good	83.4	Good
Four Mile	S1:250	C3	53	Poor	63.3	Fair	76.3	Good
Four Mile	S2:267	C3	---	---	77.7	Good	77.7	Good
Coyote	S1:400	C6	53	Poor	72	Good	75.2	Good
Coyote	S2:467	C3	58	Poor	61	Fair	77.4	Good
Coyote	S3:500	C6	40	Poor	68	Fair	69.1	Fair
Coyote	S4:542	C3	54.5	Poor	56.4	Poor	67.1	Fair
Corral	S1:571	C3	56	Poor	65.1	Fair	49	Poor
Corral	S2:574	C3	46.5	Poor	60.2	Fair	57.5	Poor

HCI Scale by Stream Type

HCI Rating	C3	C6	B2
Excellent	> 85	> 80	> 85
Good	75-84.9	70-79.9	75-84.9
Fair	60-74.9	60-69.9	60-74.9
Poor	< 60	< 60	< 60

Channel typesfollow Rosgen (1996):

B: Moderate gradient, riffle-dominated stream.

C: Low gradient,, meandering, riffle-pool stream.

Numbers denote streambed composition: boulders (2), cobble (3), or silt/clay (6)

Macroinvertebrate sampling also occurred during the functional assistance trips in 1984, 1987, and 1990. Samples were collected at most of the General Aquatic Wildlife System stations using a Winget-modified surber net. Three types of indices were reported: (1) a diversity index (DAT), which combines a measure of dominance and number of taxa (Table A3); (2) standing crop, which is the community dry weight biomass per sample (Table A4); and (3) a biotic condition index (BCI), which indicates, as a percentage, how close an aquatic ecosystem is to its own potential (Table A5). No trends were observed during these functional assistance trips, however, both the diversity and biotic condition indices were rated good to excellent while the standing crop data ranged from poor to excellent.

In the late 1940's and early 1950's, beaver (*Castor canadensis*) were introduced into Silver King Creek and the upper East Fork of the Carson River drainages (Hensley 1946; Ingles 1965). By 1964, they had established active colonies in lower and upper Four Mile Canyon Creek, and in Fly Valley at the confluence of Fly Valley and Silver King Creeks. Beaver have since been trapped out or have abandoned their colonies, so as of 2002, there are no active beaver colonies in the drainage.

In the nonmeadow portions of the watershed, Jeffrey pine (*Pinus jeffreyi*), lodgepole pine (*Pinus contorta*), and red fir (*Abies magnifica*) are the dominant conifers, while dense stands of aspen (*Populus tremuloides*) are common throughout the drainage. Sagebrush (*Artemisia tridentata*) is common near the outer periphery of the mainstem meadows. Six species of willow and sedges are the dominant riparian species present in the Silver King Creek drainage (Table 4).

2. North Fork of Cottonwood Creek

The North Fork of Cottonwood Creek is a small, spring-fed brook that originates on the east slope of Paiute Mountain, in the White Mountains of east-central California. All occupied portions of the stream occur within the Inyo National Forest in Mono County (Figure 3). The stream flows southeasterly for approximately 7.2 kilometers (4.5 miles) before merging with the South Fork to

Table 4. Common and scientific names of the riparian plant communities in the Silver King Creek drainage (Modified from Winward 1984).

Common Name	**Scientific Name**
Geyer willow	*Salix geyeriana*
Lemmons willow	*Salix lemmonii*
Blueberry willow	*Salix boothii*
Eastwoods willow	*Salix eastwoodiae*
Sierra willow	*Salix orestera*
Little willow	*Salix planifolia*
Rocky Mountain sedge	*Carex scopulorum*
Nebraska sedge	*Carex nebrascensis*
Water sedge	*Carex aquatilis*
Rusty sedge	*Carex subfusca*
Winged sedge	*Carex microptera*
Beaked sedge	*Carex rostrata*
Kentucky bluegrass	*Poa pratensis*
Tufted hairgrass	*Deschampsia caespitosa*
Red fescue	*Festuca rubra*
Western needlegrass	*Achnatherum occidentalis*

form Cottonwood Creek, which then flows eastward into Fish Lake Valley, Nevada. Only one major tributary, Tres Plumas Creek, enters the North Fork of Cottonwood Creek approximately 1.6 kilometers (1.0 mile) above its mouth. From its headwaters at 3,096 meters (10,155 feet) to the mouth of Tres Plumas Creek at 2,784 meters (9,141 feet), the North Fork of Cottonwood Creek descends 312 meters (1,023 feet) in 5.6 kilometers (3.5 miles) (Wong 1975). The average gradient is 5.6 percent, greater than that of Silver King Creek (Ryan and Nicola 1976). Despite the high gradient, the streambed is composed predominantly of fine sediments. The relatively stable, spring-fed flows, together with a low frequency of flooding, are believed to be responsible for the high amount of fine sediments (Wong 1975). Mean stream width is 2.3 meters (7.5 feet) with a 1:1 ratio of pools and riffles (Wong 1975). Pool depths range between 0.3 and 2.0 meters (1 and 7 feet) (Wong 1975).

Wong (1975) described the stream in three sections. The upper section flows through relatively flat stringer meadows with sections of heavy willow (*Salix* sp.) growth. The second section flows through a narrow canyon that increases the gradient, creating a series of cascades that form barrier falls 3 to 4 meters (10 to 13 feet) high. The stream is characterized by large boulders that create plunge pools and it is heavily overgrown with a tree canopy of aspen and understory of willow. The third section again flows through more meadows with low gradient, and willow dominates as stream cover. A 2.3-meter (7-foot) barrier is located 100 meters (330 feet) above the confluence with Tres Plumas Creek.

The climate of the Cottonwood Creek basin is cool and dry, as it is throughout the higher elevations of the White Mountains (Ryan and Nicola 1976). Studies in 1973 by Wong (1975) and in 1974 by Diana (1975) determined that the summer stream discharge ranges from 0.6 to 1.8 cubic feet per second, with daily maximum water temperatures ranging from 12 to 15.8 degrees Celsius (53.6 to 60.4 degrees Fahrenheit). Despite the abundance of spring-fed water sources, diurnal water temperatures varied as much as 10.5 degrees Celsius (18.9 degrees Fahrenheit). Limber pine (*Pinus flexilis*), aspen, and mountain mahogany (*Cercocarpus ledifolius*) are found in the drainage in addition to bristlecone pine (*Pinus longaeva*), but on the whole, there are considerably fewer species of trees than in the Silver King Creek drainage.

As in Silver King Creek, beavers were introduced into the Cottonwood Creek drainage. A colony became established in the North Fork of Cottonwood Creek, primarily between the Granite Meadow tributary and the mouth of Tres Plumas Creek (Ryan and Nicola 1976). Efforts to eliminate beaver from the North Fork of Cottonwood Creek have been successful. Grazing has occurred since the surrounding area was first settled. Originally, sheep were grazed, then beginning in 1923 only cattle were grazed. However, grazing within the North Fork Cottonwood drainage has been vacant and inactive since 2000.

3. Cabin Creek

Cabin Creek is a high elevation stream (3,200 meters [10,500 feet]) located 16 kilometers (10 miles) north of Cottonwood Creek in the White Mountains, Inyo National Forest, Mono County, California (Figure 3). Like Cottonwood Creek, Cabin Creek is small, flowing most of the year at less than 1 cubic foot per second. It flows south into Leidy Creek, which then flows eastward across the California-Nevada border into Fish Lake Valley. Dawne Becker (unpubl. data) characterizes Cabin Creek as a high gradient stream with many riffles, a few small pools, little spawning habitat, and poor winter habitat. The average gradient for the entire stream is 14.4 percent. The lower section of stream, from the confluence with Leidy Creek to about 3,000 meters (9,840 feet) elevation, has a gradient of 20.2 percent. The upper section of stream has an average gradient of 9.2 percent. Most of the stream is heavily vegetated with dense willows of all age classes with grasses, sedges (*Carex* sp.), and paintbrush (*Castilleja sp.*). Upland vegetation includes sagebrush, lupine (*Lupinus* sp.), and a few pine trees. Cabin Creek is within an active livestock grazing allotment. Some degradation of the riparian zone and stream is occurring from overutilization. Sloughing banks and trampling of tributary spring channels are causing increased sediment input.

4. Stairway Creek

Stairway Creek, Madera County, California, originates in two forks at 2,743 meters (9,000 feet) elevation and flows south into the Middle Fork San Joaquin River. The creek is located within the Ansel Adams Wilderness Area on the Sierra National Forest (Figure 4). Strand and Eddinger (1999) described

Stairway Creek based on a survey conducted in 1996. The survey focused on a 2.5-kilometer (1.6-mile) low gradient section of stream, just upstream of a 500-meter (1,640-foot) long section of stream with a greater than 40 percent gradient, above the confluence with the Middle Fork of the San Joaquin River. This section serves as a natural barrier to fish from downstream. A combination of A2 (greater than 4 percent gradient, confined channel, boulder substrate) and B3 (1 to 4 percent gradient, moderately confined channel, boulder/cobble substrate) Rosgen types (Rosgen 1996) describe this 2.5-kilometer (1.6-mile) reach. Using U.S. Forest Service Region 5 habitat typing methods (U.S. Forest Service 1996), 6 percent of the stream length was characterized as fast water while 94 percent was slow water. A breakdown of these data are summarized in Table 5. Channel stability (Pfankuch 1975) was rated "good" for all reaches sampled. Canopy cover was approximately 40 percent in the riparian zone, accounting for the low quantity of large woody debris, 3.3 pieces per 100 meters (328 feet), found in the stream.

A 650-meter (2,132-foot) Stream Condition Inventory (U.S. Forest Service 1996) reach was established in 2000 by Sierra National Forest personnel (P. Strand, unpubl. data), in order to monitor long-term habitat trends within Stairway Creek. This Stream Condition Inventory reach was within the 2.5-kilometer (1.6-mile) reach originally surveyed in 1996, and consisted of 41 percent (linear length) slow water habitats and 59 percent fast water habitats. Other information collected is summarized in Tables 6 and 7.

5. Sharktooth Creek

Sharktooth Creek exits Sharktooth Lake at 2,999 meters (9,836 feet). It is a headwater tributary to Fish Creek that flows northwest into the Middle Fork San Joaquin River, Fresno County, California. The creek is located within the John Muir Wilderness Area in the Sierra National Forest (Figure 4). Sharktooth Creek

Table 5. Summary of habitat surveys on Stairway Creek conducted in 1996. All habitats were reduced to pool, riffle, or run based on data output from FISHHAB program (U.S. Forest Service R5 Version 2) (Strand and Eddinger 1999). See Rosgen (1996) for description of stream type.

Reach #	Stream Type	Length (meters)	Percent Pools	Percent Riffles	Percent Runs
1	A2	314	92	1	7
2	B3	269	32	11	57
3	A2a+	257	70	30	0
4	B3	747	26	6	68
5	A2a+	377	71	7	22
6	B2	121	31	0	69
7	A2	418	56	5	39
Mean			54	8	38

Table 6. Cross sectional data from 2000 survey of Stairway Creek (P. Strand, U.S. Forest Service, unpubl. data).

Cross Section	Gradient (percent)	Entrenchment (meters)	Width/Depth Ratio
1	2.4	2.0	23.0
2	0.53	8.8	25.0
Mean	1.46	5.4	24.0

Table 7. Transect data from 2000 survey of Stairway Creek (P. Strand, U.S. Forest Service, unpubl. data).

Transect Number	Bankfull Width (meters)	Depth (meters)	Width at 2X Bankfull Depth (meters)	Width/Depth Ratio	Entrenchment (meters)
5	6.35	0.35	12.0	18	1.9
10	5.05	0.40	50.5	13	10.0
15	5.35	0.35	8.5	15	1.6
20	9.05	0.43	74.2	21	8.0
25	9.95	0.28	11.5	36	1.16
30	6.7	0.31	9.0	22	1.3
35	7.75	0.37	62.0	21	8.0
40	6.0	0.34	7.0	18	1.17
45	11.55	0.33	14.0	35	1.21
50	7.05	0.37	14.0	19	2.0
Mean	7.48	0.35	26.27	21.8	3.63

is described by Strand and Eddinger (1999) as having high gradient sections that provide natural migration barriers. A 250-meter (820-foot) section of stream near the confluence with Lost Keys Lake outflow is described as a cascade/falls that has a gradient of 35 percent with large cobble and boulders as substrate. Upstream of this point a step-pool sequence develops as the gradient reduces to less than 15 percent. Then comes a low gradient (less than 2 percent) section, approximately 1,565 meters (5,133 feet) in length, that is described in greater detail below. Above this section, the stream again increases in gradient to the outflow of Sharktooth Lake.

A Stream Condition Inventory (U.S. Forest Service 1996) reach was established in 1999 by Sierra National Forest personnel on the lower gradient middle section (approximately 1,565 meters [5,133 feet]) of Sharktooth Creek (Strand and Eddinger 1999), in order to monitor long-term habitat trends within Sharktooth Creek. This section of stream was a Rosgen type C3 (less than 2 percent gradient, well developed floodplain, mostly cobble with lesser amounts of gravel and sand, Table 8), and included 205 pieces of large woody debris with 7 aggregations, and stream shading was 71 percent, which indicates that the riparian area is dominated by large woody species of trees. Sixty percent of the stream length was characterized as fast water (riffles, cascades, and step-pools) while 40 percent was slow water (pools, glides and runs). Bank stability was 75 percent or greater for all 50 transect points, which are considered good ratings (U.S. Forest Service 1996). The mean temperature was 12 degrees Celsius (54 degrees Fahrenheit) with a pH of 7 and a dissolved oxygen reading of 10.4 milligrams per liter. These water quality data indicate that Sharktooth Creek does not have any water quality deficiencies for Paiute cutthroat trout, which require cool, well oxygenated water for all life stages. Table 8 shows cross section data while Table 9 provides transect data.

Table 8. Cross sectional data from 1999 survey of Sharktooth Creek (Strand and Eddinger 1999).

Cross Section	Gradient (percent)	Entrenchment (meters)	Width/Depth Ratio
1	1.64	2.15	11.12
2	1.37	3.17	19.25
3	1.19	6.52	14.37
Mean	1.4	3.95	14.91

Table 9. Transect data from 1999 survey of Sharktooth Creek (Strand and Eddinger 1999).

Transect Number	Bankfull Width (meters)	Depth (meters)	Width at 2X Bankfull Depth (meters)	Width/Depth Ratio	Entrenchment (meters)
5	8.6	0.32	> 30	26.87	3.49
10	3.1	0.41	8.1	7.52	2.61
15	3.3	0.45	14.65	7.33	4.44
20	3.21	0.12	7.79	26.75	2.43
25	4.25	0.19	9.1	22.37	2.14
30	4.2	0.3	9.8	14	2.33
35	3.08	0.24	7.05	12.83	2.29
40	2.7	0.12	4.1	22.5	1.52
45	5.95	0.37	8.05	16.08	1.35
50	3.1	0.64	8.85	4.84	2.85
Mean	4.15	0.32	10.75	16.11	2.55

H. Reasons for Listing and Current Threats

Species are placed on the endangered species list based on one or more of the five listing factors for Federal listing of a species in section 4(a)(1) of the Endangered Species Act. The five listing factors are: (1) The present or threatened destruction, modification, or curtailment of habitat or range; (2) Overutilization for commercial, recreational, scientific, or educational purposes; (3) Disease or predation; (4) Inadequacy of existing regulatory mechanisms; and (5) Other natural and manmade factors affecting the species' continued existence. The Paiute cutthroat trout was originally listed as endangered under the Endangered Species Preservation Act on March 11, 1967 (U.S. Fish and Wildlife Service 1967), and was reclassified to threatened on July 16, 1975 to facilitate management and allow regulated angling (U.S. Fish and Wildlife Service 1975). Threats at the time of reclassification included livestock grazing, recreational development, and hybridization from rainbow trout introduction. Appendix B delineates the relationships between threats, recovery actions that address them, and recovery criteria. Existing threats are as follows:

(1) The present or threatened destruction, modification, or curtailment of habitat or range.

Valuable cover for stream populations of cutthroat trout is provided by undercut banks, which are dependent on extensive vegetative cover for their stability (Behnke and Zarn 1976). Streambank sloughing occurs as the result of normal erosive forces (floods, channel realignment, etc.) but can be accelerated by human-caused activities (off-highway vehicle use, grazing, logging, etc.). Heavy recreation, such as use by anglers and backpackers, can also result in streambank degradation. Streambank sloughing results in the loss of instream cover, increased water temperatures, streambed sedimentation, elimination of spawning habitat, and reduced food supplies, and can retard the growth of willows and aspen along the stream bank (Armour *et al.* 1994; Bohn and Buckhouse 1985; Duff 1977; Kauffman *et al.* 1983a, 1983b; Marlow and Pogacnik 1985; and Meehan and Platts 1978).

Cattle last grazed the Silver King Basin during the summer of 1994. On March 31, 1995, all authorized grazing on the Silver King Allotment was placed under administrative rest and the allotment is currently vacant. It will remain vacant unless appealed and upheld under the administrative appeal process. The Cottonwood Creek and Tres Plumas allotments in the North Fork of Cottonwood Creek also have the potential to affect Paiute cutthroat trout habitat (Kondolf 1994). Grazing was suspended for both these allotments in 2000, and will be in non-use status for at least 10 years in the Cottonwood Basin (D. Hubbs, U.S. Forest Service, pers. comm. 2002). Cabin Creek is within an active grazing allotment and some degradation of habitat is occurring due to bank failure and increased sediment input. Grazing does not currently affect occupied habitat in Stairway and Sharktooth Creeks due to the inaccessibility of the area to livestock (P. Strand, pers. comm. 2003).

Beavers have been a past threat to Paiute cutthroat trout because they degrade spawning substrates and water quality. Beavers were introduced to the east slope of the Sierra Nevada (Hensley 1946; Ingles 1965). Willow and aspen growth along Silver King Creek and its tributaries, and the North Fork of Cottonwood Creek is not adequate to support a permanent beaver colony. When beavers colonize an area, as they did in upper Silver King Creek, they remove the aspen faster than it can be regenerated. Consequently after a short period, the beavers are forced to move on to other areas in search of food. After the beavers move out, the abandoned dams and lodges wash out, and the fine silt and sand from the dams is eroded and deposited in the streambed. The collapse of old beaver dams, and associated down-cutting in Four Mile Canyon Creek has caused degradation of that stream habitat. This series of events led to a 10-fold decline in the population (Ryan *in litt.* 1982).

(2) Overutilization for commercial, recreational, scientific, or educational purposes.

Paiute cutthroat trout are susceptible to unregulated angling. Connell (letter in Ryan and Nicola 1976) reported that in 1890 he and a companion took 1,500 fish from Silver King Creek in only 3 days of fishing. He noted that "...they fished only a very small part of the time" and that their angling success was enhanced when his fishing companion " ...conceived the idea of putting two hooks on his line and succeeded in bringing out two fish in the majority of his casts”. From 1952 to 1965, Silver King Creek was open to angling to reduce the number of hybrid fish and the population above Llewellyn Falls was severely depleted. Angling has been closed in Silver King Creek above Llewellyn Falls since 1965. In the early 1970's, the population above the Falls was again significantly reduced following a brief period of unauthorized angling by military personnel (Ryan and Nicola 1976). Currently, overutilization for commercial, scientific, or educational purposes is not occurring.

(3) Disease or predation

There are several natural predators (water shrews [*Sorex palustris*], dippers [*Cinclus mexicanus*], and trichopteron larvae) on Paiute cutthroat trout eggs and fry, but few on adult fish. Predation does not seem to be a significant threat at this time.

Disease is apparently a significant cause of adult mortality in the North Fork of Cottonwood Creek, particularly in the post-spawning period. Wong (1975) observed extensive fungal infections on the dorsal and caudal fins of several spawned-out fish in the North Fork of Cottonwood Creek. Many of these fish were so weakened by spawning that they were unable to recover. Few Paiute cutthroat trout apparently live beyond the age of 3 in a wild stream environment (Wong 1975). This disease has not been observed outside of the North Fork of Cottonwood Creek.

(4) Inadequacy of existing regulatory mechanisms.

Existing regulatory mechanisms appear to be adequate at this time. However, agency commitments to recovery actions may be limited due to budgetary constraints.

(5) Other natural and manmade factors affecting the species' continued existence.

In the early part of the twentieth century, Paiute cutthroat trout were eliminated from their presumed historic habitat through hybridization with introduced rainbow trout, golden trout, and Lahontan cutthroat trout. Stocking records from 1930 to 1953 document the plantings of thousands of nonnative salmonids within the Silver King Basin. Nonnative salmonids continue to occupy all of the historic habitat of the Paiute cutthroat trout.

Effective fish barriers are needed to keep other trout from invading Paiute cutthroat trout waters. Even with effective barriers, there is an ever-present risk that other trout will be introduced above the barriers by humans. Due to the proximity of nonnative fish below Llewellyn Falls, the threat of an unauthorized introduction of fish from below this area will remain until nonnative fish are removed and Paiute cutthroat trout are reestablished below the falls. This action will isolate Paiute cutthroat trout within the Silver King Basin because the Silver King Canyon contains several barriers that will prevent salmonids from migrating upstream. The Silver King Canyon is also difficult to access, which should discourage humans from moving other trout above the barriers into historical Paiute cutthroat trout habitat. The pre-1973 contamination of a portion of the North Fork of Cottonwood Creek population was apparently the result of an unauthorized trout introduction.

Paiute cutthroat trout have a distinctive evolutionary history that complicates management efforts to recover this fish. Paiute cutthroat trout evolved in isolation from other fish species, and accordingly faced substantially different selection pressures than most other North American salmonids. As a consequence, this subspecies has developed behavioral traits that render its prospects for coexisting with potential competitors highly unlikely. In those situations where other salmonids have invaded Paiute cutthroat trout habitats, the Paiute cutthroat trout have eventually been displaced. When associated with Lahontan cutthroat trout or rainbow trout, the Paiute cutthroat trout tend to lose their distinctiveness through introgressive hybridization. When associated with brook trout, Paiute cutthroat trout tend to be displaced by competition (Schroeter 1998).

The Paiute cutthroat trout faces several threats to its existence because of its limited distribution and its susceptibility to displacement by other salmonids. Several events have occurred in the past to imperil its existence, including: 1) the early introduction of rainbow trout and Lahontan cutthroat trout into the Silver King Creek drainage and subsequent introgression, 2) the introduction of beavers into the Silver King Creek drainage, 3) the occurrence of a flood in Silver King

Creek that may have eliminated a natural barrier and allowed nonnative salmonids to enter the drainage, 4) the degradation of habitat caused by livestock grazing and off-highway vehicle use in the North Fork of Cottonwood Creek, and (5) excessive angling. Its extremely limited distribution makes it vulnerable to extinction in the event of a large disturbance (Dunham *et al*. 2003; Miller *et al*. 2003). Dunham *et al*. (2003) reported that the degree to which fish are affected by a disturbance, such as fire, is related to the quality of the habitat before the disturbance, the quantity and distribution of habitat (habitat fragmentation), and the habitat requirements of the species impacted by the disturbance. The Paiute cutthroat trout population in Silver King Creek, once it becomes re-established throughout its native range, will be less susceptible than the out-of-basin populations due to the size of the drainage, the size of the population, and the quality and distribution of habitat in which it evolved.

I. Conservation Efforts

All Paiute cutthroat trout habitat is publicly owned. Silver King Creek and its tributaries are situated within the Humboldt-Toiyabe National Forest, the North Fork of Cottonwood Creek and Cabin Creek are located within the Inyo National Forest, and Stairway and Sharktooth Creeks lie within the Sierra National Forest. Silver King Creek and its tributaries are within the Carson-Iceberg Wilderness, Stairway Creek is within the Ansel Adams Wilderness, and Sharktooth Creek is within the John Muir Wilderness. The California Department of Fish and Game, with cooperation from us and the Humboldt-Toiyabe National Forest, has proposed activities intended to extend the range of Paiute cutthroat trout in Silver King Creek downstream of Llewellyn Falls to the Silver King Canyon during the fall of 2004.

Previous management efforts to protect and restore the Paiute cutthroat trout have primarily involved: 1) mechanical and chemical treatments to remove competing or introgressed fish, 2) transplants to restore fish populations in fishless waters, 3) land exchanges to secure essential habitat, 4) fishing closures, and 5) fish habitat restoration projects.

Paiute cutthroat trout have been introduced into several lakes and streams within and outside their native range (Table 1). Self-sustaining populations have been established in Silver King Creek above Llewellyn Falls, Fly Valley Creek, Corral Valley Creek, Coyote Valley Creek, and Four Mile Canyon Creek in the Silver King Creek drainage. Self-sustaining populations have also been established in the North Fork of Cottonwood Creek, Stairway Creek, Sharktooth Creek, and Cabin Creek. The introduced population in Delaney Creek is suspected to be extirpated due to the presence of brook trout; however, no recent surveys have been conducted. The 1983 introduction of Paiute cutthroat trout into Heenan Reservoir was made to establish a broodstock for artificial propagation. This population no longer exists.

Corral Valley and Coyote Valley Creeks were treated in 1964 and 1977 respectively, to remove nonnative and hybrid trout. Electrofishing efforts eliminated surviving hybrid trout and genetic analysis indicates that Corral Valley Creek now contains pure Paiute cutthroat trout (Israel *et al.* 2002). The single year treatment failed in Coyote Valley Creek because fish that survived above the treatment area repopulated downstream meadow reaches. Coyote Valley Creek was retreated during 1987 and 1988. Both Corral and Coyote Valley Creeks were restocked from Fly Valley Creek following treatments. Surveys and genetic analysis following the most recent treatments have not detected the presence of introgressed fish in either stream.

Silver King Creek was restocked from Coyote Valley and Fly Valley Creeks from 1994 through 1998, in various locations between the downstream end of Upper Fish Valley, upstream to the confluence of Fly Valley Creek. Additionally, Paiute cutthroat trout likely dispersed downstream from Fly Valley and Four Mile Canyon Creeks, which contributed to the population reestablishing. Annual snorkel surveys of Silver King Creek have revealed that substantial recruitment and multiple age classes had developed in the Paiute cutthroat trout population by 1997, and total numbers exceeded 400 fish during 1999.

Beaver control and habitat restoration were accomplished during the early to mid- 1980's in the Silver King Creek drainage above Llewellyn Falls and in the North Fork of Cottonwood Creek. Beavers have been extirpated in the vicinity of the confluence of Fly Valley Creek with Silver King Creek and also in Four Mile Canyon Creek. Beaver dams were subsequently breached in both locations. Extensive stream habitat restoration work, including rerouting the stream channel, was accomplished in Four Mile Canyon Creek. Beaver were noted in past years to occur in Tamarack and Snodgrass Creeks. No recent beaver activity has been observed in Tamarack or Snodgrass Creeks, however, the potential for recolonization throughout the drainage remains a concern.

In 1971, the Humboldt-Toiyabe National Forest completed a land exchange with the Sierra Pacific Power Company to secure management protection for most of the upper Silver King Creek watershed. The California Department of Fish and Game acquired 290 hectares (720 acres) in the vicinity of Poison Flat during 1990 for protection of Lahontan cutthroat trout, which also provides watershed protection for Silver King Creek. In 1963, the U.S. Marine Corps agreed to discontinue use of the watershed for survival training. In 1984, the Toiyabe National Forest and the California Department of Fish and Game rerouted lower Fly Valley Creek back into a historic channel to reduce sedimentation from a large headcut that was moving through a series of old beaver dams. Four Mile Canyon Creek was similarly rerouted from old beaver dams, and various habitat projects were performed to stabilize the streambanks and provide fish habitat during 1988 and 1989. Fish habitat improvement structures and bank protection projects were constructed in Silver King Creek during 1988. Cattle exclosure electric fences were constructed and maintained during 1985 through 1994 in both Silver King and Coyote Valley Creeks. These fenced exclosure areas protected stream reaches from grazing, and provided reference stream reaches to evaluate grazing impacts in the unfenced reaches.

Paiute cutthroat trout are managed by the State of California under the 4(d) rule published in 1975, which states that Paiute cutthroat trout can be taken in accordance with applicable State law and that violation of State law will also be a violation of the Endangered Species Act (Code of Federal Regulations Title 50, Section 17.44). Silver King Creek and its tributaries above Llewellyn Falls are closed to angling. Angling closures have also been established to protect the populations in Coyote Valley Creek, Corral Valley Creek, and the North Fork of Cottonwood Creek. Stairway Creek, Cabin Creek, and Sharktooth Creek are all relatively inaccessible and lightly used, and therefore are managed as wild trout fisheries without special protective regulations. The California Department of Fish and Game and the U.S. Forest Service have periodically maintained a stream guard in upper Silver King Creek to enforce the angling closure above Llewellyn

Falls. The Inyo National Forest prepared a habitat management plan for Paiute cutthroat trout in 1991. That plan includes several projects to improve habitat quality in the Cottonwood Creek basin. The actions proposed in the habitat management plan are compatible with the objectives of this recovery plan.

II. RECOVERY

A. Objective and Criteria

The objective of this recovery plan is to recovery the Paiute cutthroat trout by improving its status and habitat and eliminating nonnative salmonids so it can be delisted. Criteria for accomplishing the goal of delisting are:

1. All nonnative salmonids are removed from Silver King Creek and its tributaries downstream of Llewellyn Falls to fish barriers in Silver King Canyon;

2. A viable population occupies all historic habitat in Silver King Creek and its tributaries downstream of Llewellyn Falls to fish barriers in Silver King Canyon;

3. Paiute cutthroat trout habitat is maintained in all occupied streams;

4. The refuge populations in Corral and Coyote Creeks, Silver King Creek, and tributaries above Llewellyn Falls as well as out-of-basin populations are maintained as refugia and are secured from the introduction of other salmonid species; and

5. A long-term conservation plan and conservation agreement are developed, which will be the guiding management documents once Paiute cutthroat trout are delisted.

Specifications for these recovery criteria are discussed in greater detail below (section II.B).

Because this recovery plan is partially focused on habitat improvements, it also provides conservation benefits for two candidate species, the Sierra Nevada population of the mountain yellow-legged frog and the Yosemite toad.

B. Recovery Strategy

The primary threat to the Paiute cutthroat trout is hybridization with nonnative trout, compounded by its extremely limited distribution (making it vulnerable to catastrophic events). Consequently, it is critical to remove nonnative trout from the historic range downstream of Llewellyn Falls and reestablish Paiute cutthroat trout populations there, monitoring population abundance and genetics to evaluate success. Reinvasion of Paiute cutthroat trout habitat by nonnative trout should be prevented by monitoring or establishing instream barriers and discouraging deliberate introductions. Because the Paiute cutthroat trout is vulnerable to angling pressure, appropriate fishing regulations and closures should be maintained and enforced by a stream guard and signage. Potential habitat degradation should be addressed by appropriate fish habitat improvement actions, including management of recreational access and grazing, and control of beaver populations as necessary. The recovery criteria above should be met by addressing these threats, as detailed below.

Meeting the first and second recovery criteria will secure long-term protection and population viability of Paiute cutthroat trout by their expansion within their native range. This range expansion will be accomplished by removing nonnative trout from the Silver King Creek drainage from Llewellyn Falls downstream to the Silver King Canyon, including tributaries, followed by reintroduction with Paiute cutthroat trout from donor tributaries best suited as determined by genetic testing (Israel *et al*. 2002). A viable population will be achieved when the population is secure and comprises three or more age classes for 5 years, and consists of a minimum of 2,500 fish greater than 75 millimeters (3 inches) (Hilderbrand and Kershner 2000). This figure is a preliminary estimate and may need to be revised as additional information becomes available. Population estimates will be made during the non-native fish eradication. This estimate will be used as a surrogate to help us understand the population size of Paiute cutthroat trout that will be expected within the historic range and aid in validating the minimum number needed for recovery. Once this estimate is made, population data from above Llewellyn Falls will be used to estimate a range in the population size that can be expected due to inherent natural fluctuation as seen in Figure 5.

The third recovery criterion is to maintain suitable habitat for Paiute cutthroat trout. Historic and occupied Paiute cutthroat trout stream and riparian habitat should have no degradation from existing conditions due to anthropogenic effects. The condition of existing habitat will be identified using established stream habitat monitoring protocols which use measurable and repeatable methods (see section I.G above and Appendix A). Beaver control will need to be conducted in the event that they repopulate the drainage. To secure the protection of the North Fork Cottonwood population, a second barrier will be needed to protect the population from the introduction of nonnative trout species from downstream. Cabin Creek is within an active grazing allotment where continued management will be necessary to ensure degradation of Paiute cutthroat habitat does not occur. Stairway and Sharktooth Creeks are subject to limited human disturbance since they are in designated wilderness areas, are inaccessible to livestock, and get limited recreational use. Therefore, habitat monitoring should be done periodically to document stochastic events such as a rain on snow event which occurred in 1997 (P. Strand, pers. comm. 2002).

The fourth recovery criterion is to protect and enhance Paiute cutthroat trout that do not occupy historic habitat. To protect against a catastrophic event that could affect the entire Silver King Creek gene pool, populations in Corral Valley and Coyote Valley Creeks, Silver King Creek and tributaries (Four Mile Canyon, Fly Valley, and Bull Canyon Creeks) above Llewellyn Falls, and the four out-of-basin populations must be maintained as Paiute cutthroat trout refugia. Monitoring these populations will aid in management decisions aimed to maintain and improve the abundance of Paiute cutthroat trout and collection of long-term trend data. Continued genetic monitoring of all populations of Paiute cutthroat trout will be used to: 1) monitor population genetic diversity, 2) evaluate effective population size and reproductive isolation, 3) examine populations for evidence of hybridization, and 4) identify appropriate donor sources.

The fifth and final criterion is to develop a long-term conservation plan and conservation agreement that will guide the agencies responsible for the management of Paiute cutthroat trout after it is delisted. The purpose of the conservation plan is to ensure that adequate regulatory mechanisms and management programs remain in existence after delisting to ensure that all populations of Paiute cutthroat trout and their habitat are maintained. The conservation plan will be consistent with other existing cutthroat trout subspecies conservation plans. The purpose of the conservation agreement is to define the role of the management agencies and to document their commitment to implementing the conservation plan. The conservation plan and conservation agreement will need to be approved and signed by all responsible agencies before delisting occurs.

Prior to implementation of any task in this plan, the lead Federal agency must comply with all applicable provisions of the National Environmental Policy Act and the Endangered Species Act. All necessary Federal, State, and local permits or authorizations must be obtained. These recovery criteria were designed to provide a basis for consideration of delisting, but not for automatic delisting. Before delisting occurs, we must determine that the five listing factors (as discussed previously) no longer are present or continue to adversely affect the listed species. The final decision regarding delisting will be made only after a thorough review of all relevant information. It is our goal to achieve recovery as quickly as possible while minimizing social and economic impacts.

C. Narrative Outline of Recovery Actions

1. Remove nonnative fish from Silver King Creek downstream from Llewellyn Falls to barriers in Silver King Canyon. Hybridization, which has occurred within and outside the native drainage, continues to be a threat. Chemically treat Silver King Creek to remove all introgressed fish downstream from Llewellyn Falls to barriers in Silver King Canyon, including all tributaries that enter the mainstem in this reach. In addition, Tamarack Lake, which was formerly stocked with trout in 1991, must be treated to remove any remaining fish. Tamarack Lake will remain fishless for the benefit of amphibian species.

2. Reintroduce Paiute cutthroat trout into renovated stream reaches in historic habitat. The most effective means of insuring that the Silver King population remains above critical minimum levels is by expanding the population downstream into historical habitat. Franklin (1980) recommended an effective population size of at least 500 individuals to maintain adequate long-term genetic variation. Hilderbrand and Kershner

(2000) suggested that 2,500 individuals may be necessary to maintain cutthroat populations in small streams. This estimate is preliminary and may need to be revised as additional information becomes available.

Restock Silver King Creek below Llewellyn Falls with pure Paiute cutthroat trout within 1 year after the final chemical treatment. Restocking may need to be continued for several years to enhance recolonization. The fish used for restocking should be taken from populations based on results from genetic analysis and will be mixed with other populations, as necessary, to promote genetic heterozygosity (Israel *et al*. 2002). Expansion of Paiute cutthroat trout downstream from Llewellyn Falls will provide additional protection from the potential unauthorized introduction of non-native trout.

3. Protect and enhance all occupied Paiute cutthroat trout habitat. Habitats have been improved through livestock grazing closures and eradication of beavers. Historic and occupied Paiute cutthroat trout stream and riparian habitat should have no degradation from existing conditions due to anthropogenic effects. Existing habitat will be identified using established stream and riparian habitat monitoring protocols which use measurable and repeatable methods. Ongoing monitoring will be necessary to detect recolonization of beaver within Paiute cutthroat trout habitats. In addition, various types of habitat protection and restoration measures are needed to maintain populations at levels that are high enough to avoid the adverse effects associated with inbreeding depression. Several actions are needed to maintain or restore habitat conditions to the levels needed to support recovery.

3.1 Restore and maintain riparian habitat quality and stream channels in the Silver King Creek drainage. Recreation, livestock, and beaver have degraded habitat conditions in the Silver King basin. Paiute cutthroat trout evolved in an isolated headwater environment. They require good water quality and clean spawning gravel to survive. The most favorable habitat is provided by streams with undercut or overhanging banks and abundant riparian vegetation. Several management activities are needed to improve Silver King basin streams.

3.1.1 Institute a habitat monitoring program. Institute a stream and riparian habitat monitoring program which uses an established stream monitoring protocol with measurable and repeatable methods.

3.1.2 Monitor and manage the amount of recreational trail and campsite use adjacent to occupied habitats. Bank conditions must be monitored and managed to prevent physical damage to banks and associated riparian vegetation. Trails and campsites should be relocated away from streams in areas where stream-side degradation is occurring.

3.1.3 Protect Paiute cutthroat trout habitat from effects of grazing. Continue to exclude grazing in Silver King Creek drainage.

3.1.4 Conduct periodic surveys to detect reinvasion by beavers. Periodic surveys should be made to detect beavers that migrate to the Silver King Creek drainage from other areas before they construct dams that create barriers to fish migration and become sources of future streambed sedimentation.

3.1.5 Remove beavers from the watershed and dismantle dams and lodges if any are built. Beavers can severely degrade areas, such as Silver King Creek, that do not have adequate aspen or willow growth. Whenever they are discovered in the Silver King Creek drainage, they should be removed and the dams and lodges that have been built should be dismantled.

3.1.6 Develop and implement solutions for other identified habitat problems. If conflicting land uses are identified and problems develop, solutions to the problems should be developed and remedies implemented to provide habitat recovery.

3.2 Restore and maintain stream banks, riparian vegetation, and stream channels in the North Fork of Cottonwood Creek drainage. Habitat conditions in the North Fork of Cottonwood Creek drainage are generally good, but localized damage has occurred in some areas as the result of beaver use and past human activities.

Management activities will be required to maintain and/or improve habitat in portions of the North Fork of Cottonwood Creek. Stream reaches that support Paiute cutthroat trout should be periodically monitored to maintain existing habitat conditions using an established stream monitoring protocol with measurable and repeatable methods.

3.2.1 Conduct periodic habitat surveys. Conduct habitat surveys to determine if there are any potential sources of habitat degradation, including but not limited to stream sedimentation, stream bank stability, or riparian conditions.

3.2.2 Continue to enforce road closure barriers at existing and potential access points. Off-highway vehicles pose a threat to Paiute cutthroat trout by directly degrading habitat when crossing streams and creating new sources of erosion, and providing anglers with easier access to Paiute cutthroat trout streams. Existing road closures should be strictly enforced and new barriers constructed if they are needed to restrict access. If pioneer roads are created within the basin area that would allow access to Cottonwood Basin, establish barriers to eliminate unauthorized use.

3.2.3 Protect Paiute cutthroat trout habitat from effects of grazing. Continue to limit grazing in the North Fork Cottonwood drainage. If grazing is allowed, cattle should be excluded from all riparian areas and appropriate grazing strategies implemented.

3.2.4 Maintain recreation opportunities as primitive and semi-primitive. Directing large numbers of recreational users to North Fork of Cottonwood Creek would inevitably stimulate unauthorized angling for Paiute cutthroat trout. Because Paiute cutthroat trout are currently present in very low numbers and are extremely vulnerable to angling, recreational access to the basin should be maintained at appropriate levels.

3.2.5 Conduct periodic surveys to detect reinvasion by beavers. Periodic surveys should be made to detect beavers that migrate back to North Fork Cottonwood Creek from other areas so they can be removed before they construct dams that create barriers to fish migration and become sources of future streambed sedimentation.

3.2.6 Remove beavers from the watershed and dismantle dams and lodges. Whenever beavers are discovered in North Fork Cottonwood Creek, they should be removed and the dams and lodges they have built should be dismantled.

3.2.7 Construct a second barrier on North Fork Cottonwood Creek. The existing pure population in the North Fork of Cottonwood Creek is now restricted to the upper 5.5 kilometers (3.4 miles) above a natural barrier. A second barrier is necessary to secure the population from reinvasion of nonnative trout species.

3.2.8 Develop and implement solutions for other identified habitat problems. If conflicting land uses are identified and problems develop, solutions to the problems should be developed and remedies implemented to provide habitat recovery.

3.3 Maintain stream and riparian habitat quality in Stairway, Sharktooth, and Cabin Creeks. Habitat conditions in the Stairway and Sharktooth Creek drainages are generally very good, and future management needs will be limited to maintaining existing conditions. Cabin Creek is within an active grazing allotment where continued management will be necessary to ensure degradation of Paiute cutthroat trout habitat does not occur.

Stream reaches that support Paiute cutthroat trout should be periodically monitored to maintain existing habitat conditions using an established stream monitoring protocol with measurable and repeatable methods.

3.3.1 Conduct periodic habitat surveys. Conduct habitat surveys of each stream to determine if there are any potential sources of habitat degradation, including but not limited to stream sedimentation, stream bank stability, or riparian conditions.

3.3.2 Protect Paiute cutthroat trout habitat from effects of grazing in Cabin Creek. Implement a grazing strategy that will protect occupied habitat from the effects of grazing in the Cabin Creek drainage.

3.3.3 Develop and implement solutions for other identified habitat problems. If conflicting land uses are identified and problems develop, solutions to the problems should be developed and remedies implemented to provide habitat recovery.

4. Continue to monitor and manage existing and reintroduced populations. The number of fish in the existing populations must be stable or

increasing. Monitoring of Paiute cutthroat trout populations should track population abundance and composition, identify any hybridization, assess barrier integrity, and maintain genetic heterozygosity.

4.1 Enforce all laws and regulations protecting the Paiute cutthroat trout and its habitat, and periodically review their effectiveness. All laws and regulations that provide protection for Paiute cutthroat trout must be enforced. Enforcement personnel from all agencies should be given maps denoting the location of all populations within their area of responsibility. These personnel should also be advised of the types of activities most likely to be detrimental to the Paiute cutthroat trout.

4.1.1 Maintain a seasonal guard in upper Silver King Creek. Because of the extreme susceptibility of Paiute cutthroat trout to angling pressure, a seasonal guard should be hired to insure that the angling regulations above Llewellyn Falls in Silver King Creek are properly enforced.

4.1.2 Prevent exotic fish introductions into Paiute cutthroat trout waters. Paiute cutthroat trout have been displaced from several streams and lakes because of unauthorized introductions of nonnative trout. This threat will always exist, but several actions can be taken to minimize the risk. Packers and recreational users should be informed and educated on the distinctiveness of the Paiute cutthroat trout and advised of the consequences an unauthorized transplant would have on existing populations and on their opportunities to use the affected streams in the future.

4.2 Review existing laws and regulations and propose necessary changes. The Paiute cutthroat trout is unwary and therefore, highly vulnerable to angling. Consequently, restrictive regulations are necessary to maintain viable populations. The opportunity for a highly regulated and special designation fishery above Llewellyn Falls should be explored during the non-native fish eradication described under Recovery Action 1. No fishing should be allowed in the North Fork of Cottonwood Creek. Explore additional out-of-basin population locations.

4.3 Maintain viable, genetically pure populations in Silver King Creek. A variety of actions are needed to maintain the genetic integrity of the existing populations in Silver King Creek. Baseline and follow-up surveys are needed to ensure population levels are stable or increasing and that other trout species have not invaded Paiute cutthroat trout waters.

4.3.1 Monitor abundance and age class composition. Annually survey test sections to assess population size, determine age class composition, and monitor the condition of the different populations within the Silver King Creek drainage.

4.3.2 Evaluate the potential for occurrence of hybrid trout. Conduct annual surveys until population levels reach or exceed recovery plan objectives. Subsequent surveys should be conducted periodically to identify unauthorized introductions of other trout species. Surveys should include appropriate genetic analysis to detect hybrid individuals. If hybrids are discovered, appropriate action should take place.

4.3.3 Assess integrity of barriers. Periodically inspect all fish barriers in the Silver King Creek drainage to ascertain their effectiveness in preventing other fish species from invading Paiute cutthroat trout habitats.

4.3.4 Mix populations in the Silver King drainage as necessary to maintain genetic diversity. If it is determined that any of the populations in the Silver King drainage suffer from inbreeding depression or the long-term depletion of genetic variance, they may be mixed with other populations to promote genetic heterozygosity.

4.3.5 Develop and implement actions, as needed, to protect genetic integrity. Take action and develop solutions to protect the genetic integrity of these populations if threats are identified.

4.4 Maintain viable, genetically pure populations in the North Fork of Cottonwood Creek. The North Fork of Cottonwood Creek is a necessary refuge for Paiute cutthroat trout in the event of a catastrophic occurrence in the Silver King Creek drainage. It is also important because it will help secure the genetic diversity of other Paiute cutthroat populations. A variety of actions are needed to maintain the genetic integrity of the existing population. Baseline and follow-up surveys are needed to ascertain if population levels are stable or increasing, critical fish barriers are intact, and to ensure that other trout species have not invaded Paiute cutthroat trout waters.

4.4.1 Monitor abundance and age class composition. Periodically survey test sections to assess population size, determine age class composition, and monitor the condition of the different populations.

4.4.2 Evaluate the potential for occurrence of hybrid trout. Conduct periodic surveys to look for unauthorized introductions of other trout species. Surveys should include appropriate genetic analysis to detect hybrid individuals. If hybrids are discovered, appropriate action should take place.

4.4.3 Assess integrity of barriers. Periodically inspect all fish barriers in the drainage to ascertain their effectiveness in preventing other fish species from invading Paiute cutthroat trout habitats.

4.4.4 Mix populations in North Fork Cottonwood Creek as necessary to maintain genetic diversity. If it is determined that the population in North Fork Cottonwood Creek suffers from inbreeding depression or the long-term depletion of genetic variance, they may be mixed with other populations to promote genetic heterozygosity.

4.4.5 Develop and implement actions, as needed, to protect genetic integrity. Take action and develop solutions to protect the genetic integrity of this population if threats are identified.

4.5 Maintain viable, genetically pure populations in Stairway, Sharktooth and Cabin Creek drainages. The remote locations of Stairway, Sharktooth and Cabin Creeks make them excellent refuge habitats for the Paiute cutthroat trout. Maintaining the existing population should require only modest management efforts because of their remote locations.

4.5.1 Monitor abundance and age class composition. Periodically survey test sections to assess population size, determine age class composition, and monitor the condition of the different populations.

4.5.2 Evaluate the potential for occurrence of hybrid trout. Conduct periodic surveys to look for unauthorized introductions of other trout species. Surveys should include appropriate genetic analysis to detect hybrid individuals. If hybrids are discovered, appropriate action should take place.

4.5.3 Assess integrity of barriers. Periodically inspect fish barriers in each stream to ascertain their effectiveness in preventing other fish species from invading Paiute cutthroat trout habitats.

4.5.4 Mix populations in Stairway, Sharktooth, and Cabin Creeks as necessary to maintain genetic diversity. If it is determined that any of the populations in Stairway, Sharktooth, or Cabin Creeks suffer from inbreeding depression or the long-term depletion of genetic variance, they may be mixed with other populations to promote genetic heterozygosity.

4.5.5 Develop and implement actions, as needed, to protect genetic integrity. Take action and develop solutions to protect the genetic integrity of these populations if threats are identified.

4.6 Explore additional out-of-basin locations. Because Paiute cutthroat trout have a very limited range and refuge populations are in isolated drainages susceptible to stochastic and anthropogenic disturbances, it may be useful to increase the number of refuge populations.

5. Develop a long-term conservation plan and conservation agreement. A conservation plan for the long-term management of Paiute cutthroat trout and a conservation agreement between all involved agencies must be developed before the species can be delisted. The purpose of the conservation plan is to ensure that adequate regulatory mechanisms and management programs remain in existence after delisting to ensure that all populations of Paiute cutthroat trout and their habitat are maintained. The purpose of the conservation agreement is to define the role of the management agencies and to document their commitment to implementing the conservation plan.

5.1 Develop a long-term conservation plan. A conservation plan should be prepared that will incorporate all the information obtained through the completion of the recovery plan actions. All agencies will need to maintain records on their recovery activities and provide pertinent information in development of the conservation plan. The conservation plan will need to provide pertinent biological and management information on the Paiute cutthroat trout for use in maintaining Paiute cutthroat trout populations. It must identify how populations will be monitored to document the status and condition of populations and habitats, and will identify conditions that would warrant relisting the Paiute cutthroat trout. The conservation plan should be developed and approved through the conservation agreement by all agencies with management jurisdiction over Paiute cutthroat trout populations before the species is delisted.

5.2 <u>Develop a conservation agreement</u>. A conservation agreement should be approved and signed by all involved agencies with Paiute cutthroat trout populations on areas under their jurisdiction to document their approval and commitment to implementing the conservation plan.

6. <u>Inform the public of Paiute cutthroat trout recovery objectives and pertinent management activities</u>. Existing and prospective public users of the areas that support Paiute cutthroat trout populations should be informed about the Paiute cutthroat trout recovery effort, and should be notified of activities, such as chemical treatments, that may temporarily restrict their use of an area. Packers and other recreational users should be informed of the consequences that unauthorized angling or "coffee-can" transplants will have on the integrity of pure populations and on future recreational opportunities.

6.1 <u>Manufacture and post informational signs</u>. Informational signs should be installed at public access areas, and interested individuals and organizations should be notified of management activities that might affect their use of an area.

6.2 <u>Notify user groups of restoration goals, chemical treatments, and future management</u>. User groups should be notified of chemical treatment schedules and advised to use alternative recreational areas. Details of transplants should be made public by inclusion in California Department of Fish and Game archives and publication in California Fish and Game if deemed appropriate by the editors. User groups should also be informed regarding the long-term restoration goal of expanding Paiute cutthroat trout downstream to Silver King Canyon, as well as the opportunity for California Department of Fish and Game to establish a recreational fishery.

III. IMPLEMENTATION SCHEDULE

The implementation schedule that follows lists the actions and estimated costs for the recovery program for the Paiute cutthroat trout. It is a guide for meeting the recovery goals outlined in this plan. Parties with authority, responsibility, or expressed interest to implement a specific recovery action are identified in the Implementation Schedule. The listing of a party in the Implementation Schedule does not require, nor imply a requirement, that the identified party has agreed to implement the actions or to secure funding for the implementing the actions. However, parties willing to participate may benefit by being able to show in their own budgets that their funding request is for a recovery action identified in an approved recovery plan and is therefore considered a necessary action for the overall coordinated effort to recover Paiute cutthroat trout. Also, section 7(a)(1) of the Endangered Species Act directs all Federal agencies to utilize their authorities in furtherance of the purposes of the Endangered Species Act by carrying out programs for the conservation of threatened and endangered species.

In the implementation schedule, actions are arranged in priority order. The assigned priorities are defined as follows:

Priority 1 - An action that **must** be undertaken to prevent extinction or to prevent the species from declining irreversibly in the **foreseeable** future.

Priority 2 - An action that must be taken to prevent a significant decline in population or habitat quality, or some other significant negative impact short of extinction.

Priority 3 - All other actions necessary to meet the recovery objective.

Key to Acronyms used in the Implementation Schedule:

Agencies

CDFG	=	California Department of Fish and Game
FS	=	U.S. Forest Service
FWS	=	U.S. Fish and Wildlife Service
*	=	Primary responsible partner: a partner likely to take the lead, or have an especially large role in implementing a recovery action.

Streams

CC	=	Cabin Creek
NFCC	=	North Fork Cottonwood Creek
SHC	=	Sharktooth Creek
SKC	=	Silver King Creek
STC	=	Stairway Creek

† Continued implementation of action expected to be necessary after delisting.

‡ Task expected to be necessary until delisting of species.

Implementation Schedule for the Revised Recovery Plan for the Paiute Cutthroat Trout										
Priority Number	Action Number	Action Description	Action Duration (Years)	Responsible Parties	Total Cost ($1,000's) 2004-2013	Cost Estimates ($1,000's) by Fiscal Year				
						FY 04	FY 05	FY 06	FY 07	FY 08
1	1	Remove nonnative fish from SKC downstream from Llewellyn Falls to barriers in SKC Canyon	3	CDFG* FS FWS	80 10 10	30 4 4	25 3 3	25 3 3		
1	2	Reintroduce Paiute cutthroat trout into renovated stream reaches in historic habitat in lower SKC	5	CDFG* FS	38 2				7.6 0.4	7.6 0.4
1	3.2.7	Construct a second barrier on lower NFCC	3	FS*	105		35	35	35	
1	4.1.2	Prevent exotic fish introductions into Paiute cutthroat trout waters	Ongoing†	CDFG	20	2	2	2	2	2
Priority 1 actions subtotal					265	40	68	68	45	10
2	3.1.1	Institute a habitat monitoring program	Periodic‡	FS*	Unknown					
2	3.1.2	Monitor and manage amount of recreational trail and campsite use adjacent to occupied habitats in SKC watershed	Ongoing†	FS*	10	1	1	1	1	1
2	3.1.3	Protect Paiute cutthroat trout habitat from effects of grazing in SKC watershed	Periodic†	FS* FWS	Unknown					

Implementation Schedule for the Revised Recovery Plan for the Paiute Cutthroat Trout

Priority Number	Action Number	Action Description	Action Duration (Years)	Responsible Parties	Total Cost ($1,000's) 2004-2013	Cost Estimates ($1,000's) by Fiscal Year				
						FY 04	FY 05	FY 06	FY 07	FY 08
2	3.1.4	Conduct periodic surveys in SKC to detect reinvasion by beavers	Periodic‡	FS* CDFG	2 0.5					1 0.25
2	3.1.5	Remove beavers from SKC watershed if detected and dismantle dams and lodges if any are built	Periodic†	CDFG* FS	Unknown					
2	3.2.1	Conduct periodic habitat surveys at NFCC	Periodic‡	FS*	Unknown					
2	3.2.2	Continue to enforce road closure barriers in NFCC at existing and potential access points	Ongoing‡	FS*	10	1	1	1	1	1
2	3.2.3	Protect Paiute cutthroat trout habitat in NFCC from effects of grazing	Periodic†	FS* FWS	Unknown					
2	3.2.4	Maintain recreation opportunities as primitive and semi-primitice in NFCC	Ongoing†	FS*	Unknown					
2	3.2.5	Conduct periodic surveys in NFCC to detect reinvasion by beavers	Periodic‡	FS* CDFG	0.325 0.325					0.162 0.162
2	3.2.6	Remove beavers in NFCC if detected and dismantle dams and lodges if any are built.	Periodic†	CDFG* FS	Unknown					

Implementation Schedule for the Revised Recovery Plan for the Paiute Cutthroat Trout										
Priority Number	Action Number	Action Description	Action Duration (Years)	Responsible Parties	Total Cost ($1,000's) 2004-2013	Cost Estimates ($1,000's) by Fiscal Year				
						FY 04	FY 05	FY 06	FY 07	FY 08
2	3.2.8	Develop and implement solutions for other identified habitat problems in NFCC	Periodic	FS* FWS	Unknown					
2	3.3.2	Protect Paiute cutthroat trout habitat from effects of grazing in CC	Periodic†	FS* FWS	Unknown					
2	4.1.1	Maintain a seasonal guard in upper SKC	Ongoing†	CDFG* FS	10 10	1 1	1 1	1 1	1 1	1 1
2	4.3.3	Assess integrity of barriers in SKC	Ongoing‡	FS* CDFG	4 3.5	0.4 0.35	0.4 0.35	0.4 0.35	0.4 0.35	0.4 0.35
2	4.3.4	Mix populations in SKC as necessary to maintain genetic diversity	5	CDFG* FS FWS	8 1 1				1.6 0.2 0.2	1.6 0.2 0.2
2	4.4.2	Evaluate the potential for occurrence of hybrid trout in NFCC	Ongoing‡	CDFG* FS FWS	3 1 1	0.3 0.1 0.1	0.3 0.1 0.1	0.3 0.1 0.1	0.3 0.1 0.1	0.3 0.1 0.1
2	4.4.3	Assess integrity of barriers in NFCC	Ongoing‡	FS* CDFG	2 1.75	0.2 0.17	0.2 0.17	0.2 0.17	0.2 0.17	0.2 0.17
2	4.4.4	Mix populations in NFCC as necessary to maintain genetic diversity	1	CDFG* FS FWS	Unknown					
2	4.5.4	Mix populations in STC, SHC, and CC as necessary to maintain genetic diversity	Periodic	CDFG* FS FWS	Unknown					

Implementation Schedule for the Revised Recovery Plan for the Paiute Cutthroat Trout

Priority Number	Action Number	Action Description	Action Duration (Years)	Responsible Parties	Total Cost ($1,000's) 2004-2013	Cost Estimates ($1,000's) by Fiscal Year				
						FY 04	FY 05	FY 06	FY 07	FY 08
Priority 2 actions subtotal					69.4	5.63	5.63	5.63	7.63	9.2
3	3.1.6	Develop and implement solutions for other identified problems in SKC	Periodic	CDFG* FS* FWS	1.6 1.6 1.6			0.53 0.53 0.53		
3	3.3.1	Conduct periodic habitat surveys in STC, SHC, and CC	Periodic‡	FS*	25		12.5	12.5		
3	3.3.3	Develop and implement solutions for other identified habitat problems in STC, SHC, and CC	Periodic	CDFG* FS FWS	0.33 0.33 0.33					0.165 0.165 0.165
3	4.2	Review existing laws and regulations and propose necessary changes	Ongoing	CDFG*	2	1				
3	4.3.1	Monitor abundance and age class composition in SKC	Ongoing‡	CDFG* FS	70 10	7 1	7 1	7 1	7 1	7 1
3	4.3.2	Evaluate the potential for occurrence of hybrid trout in SKC	Ongoing‡	CDFG* FS FWS	9 1 1	0.9 0.1 0.1	0.9 0.1 0.1	0.9 0.1 0.1	0.9 0.1 0.1	0.9 0.1 0.1
3	4.3.5	Develop and implement actions, as needed, to protect genetic integrity in SKC	Periodic	CDFG* FWS FS	Unknown					
3	4.4.1	Monitor abundance and age class composition in NFCC	Ongoing‡	CDFG* FS	30 10	3 1	3 1	3 1	3 1	3 1

Implementation Schedule for the Revised Recovery Plan for the Paiute Cutthroat Trout										
Priority Number	Action Number	Action Description	Action Duration (Years)	Responsible Parties	Total Cost ($1,000's) 2004-2013	Cost Estimates ($1,000's) by Fiscal Year				
						FY 04	FY 05	FY 06	FY 07	FY 08
3	4.4.5	Develop and implement actions, as needed, to protect genetic integrity in NFCC	Periodic	CDFG* FWS FS	Unknown					
3	4.5.1	Monitor abundance and age class composition in STC, SHC, and CC	Periodic‡	CDFG* FS	15 25		12.5	5 12.5		
3	4.5.2	Evaluate the potential for occurrence of hybrid trout composition in STC, SHC, and CC	Periodic‡	CDFG* FS	4 1			1.33 0.33		
3	4.5.3	Assess integrity of barriers in STC, SHC, and CC	Periodic‡	CDFG* FS	0.75 0.5			0.25 0.166		
3	4.5.5	Develop and implement actions, as needed, to protect genetic integrity in STC, SHC, and CC	Periodic	CDFG* FWS FS	Unknown					
3	4.6	Explore additional out-of-basin locations	Ongoing	CDFG* FWS FS	Unknown					
3	5.1	Develop long-term conservation plan	2	CDFG* FWS FS	12					
3	5.2	Develop a conservation agreement	2	CDFG* FWS FS	Unknown					

Implementation Schedule for the Revised Recovery Plan for the Paiute Cutthroat Trout										
Priority Number	Action Number	Action Description	Action Duration (Years)	Responsible Parties	Total Cost ($1,000's) 2004-2013	Cost Estimates ($1,000's) by Fiscal Year				
						FY 04	FY 05	FY 06	FY 07	FY 08
3	6.1	Manufacture and post informational signs	4	FS*	4	1	1	1	1	
3	6.2	Notify user public of restoration goals, chemical treatments, and future management	5	CDFG* FS FWS	6 1 1	1.5 0.2 0.2	1.5 0.2 0.2	1.5 0.2 0.2	1.5 0.2 0.2	 0.2 0.2
Priority 3 actions subtotal					224.05	17	41	49.68	16	14

Total Estimated Cost of Recovery: $558,450 + additional costs that cannot be estimated at this time.

Total costs of recovery for ongoing and periodic tasks are calculated based on the projected 10-year period to delisting. Costs of certain tasks (i.e., those relating to developing and implementing additional actions to protect genetic integrity, developing solutions to future land use conflicts, protecting habitat from impacts due to potential future alteration of grazing management, exploring additional out-of-basin locations, removal of beavers that may colonize Paiute cutthroat trout habitat, and developing a conservation agreement) cannot be estimated because their scope and the need to implement them will be dependent on future events or obtaining additional information.

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APPENDIX A. Summary of General Aquatic Wildlife System (GAWS) Survey Locations, Sediment Sampling, and Macroinvertebrate Sampling

Table A1. Summary of GAWS station site characteristics (Modified from Duff 1985).

Stream	Station	Elevation (m)	Gradient	Channel Width (m)	Water Depth (m)	Riffle/Pool Ratio	Reach Length (m)
Silver King	S1:610	2457	2.0	7.92	0.26	40/60	150
Silver King	S2:640	2463	2.5	9.79	0.25	24/76	374
Silver King	S3:641	2465	2.0	8.10	0.22	70/30	296
Silver King	S4:700	2454	2.0	11.4	0.14	12/88	150
Silver King	S5:725	2484	1.0	7.4	0.17	20/80	150
Silver King	S6:738	2486	2.0	9.58	0.24	52/48	150
Silver King	S6A:745	2488	2.0	8.36	0.17	45/55	150
Silver King	S7:775	2499	2.0	5.72	0.16	31/69	150
Silver King	S8:813	2505	2.5	6.42	0.12	15/85	150
Bull Canyon	S1:040	2463	2.5	5.64	0.24	50/50	150
Bull Canyon	S2:100	2475	4.5	6.86	0.095	33/67	150
Fly Valley	S1:500	2646	3.0	2.76	0.82	20/80	150
Four Mile	S1:250	2560	2.5	3.27	0.15	17/83	300
Coyote	S1:400	2484	1.0	2.9	0.10	60/40	150
Coyote	S2:467	2489	1.5	3.8	0.84	60/40	150
Coyote	S3:500	2492	1.0	2.76	0.12	60/40	150
Coyote	S4:542	2498	2.0	2.56	0.11	55/45	150
Corral	S1:571	2525	2.5	2.5	0.14	55/45	150
Corral	S2:574	2532	3.0	2.46	0.33	40/60	240

Five transects were measured within each reach. Channel (bankfull) width is the average width of all five transects. Water depth is the average water depth taken over all 5 transects (15 to 20 depth measurements were taken at each of the 5 transects,75-100 measurements). Station locations are identified in Figures A-1 and A-2.

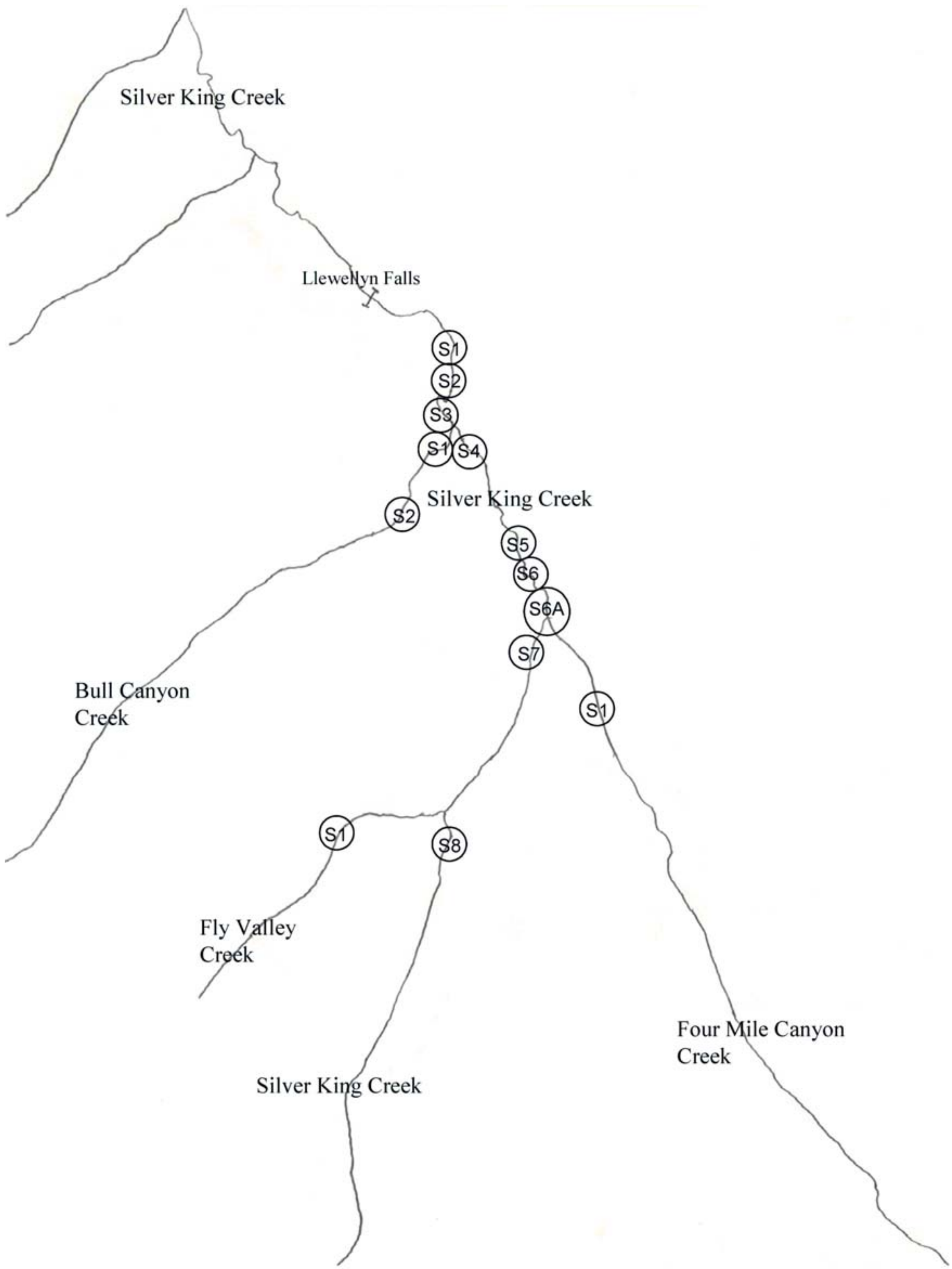

Figure A1. Location of GAWS stations on Silver King, Four Mile Canyon, Fly Valley, and Bull Canyon Creeks, Alpine County, California.

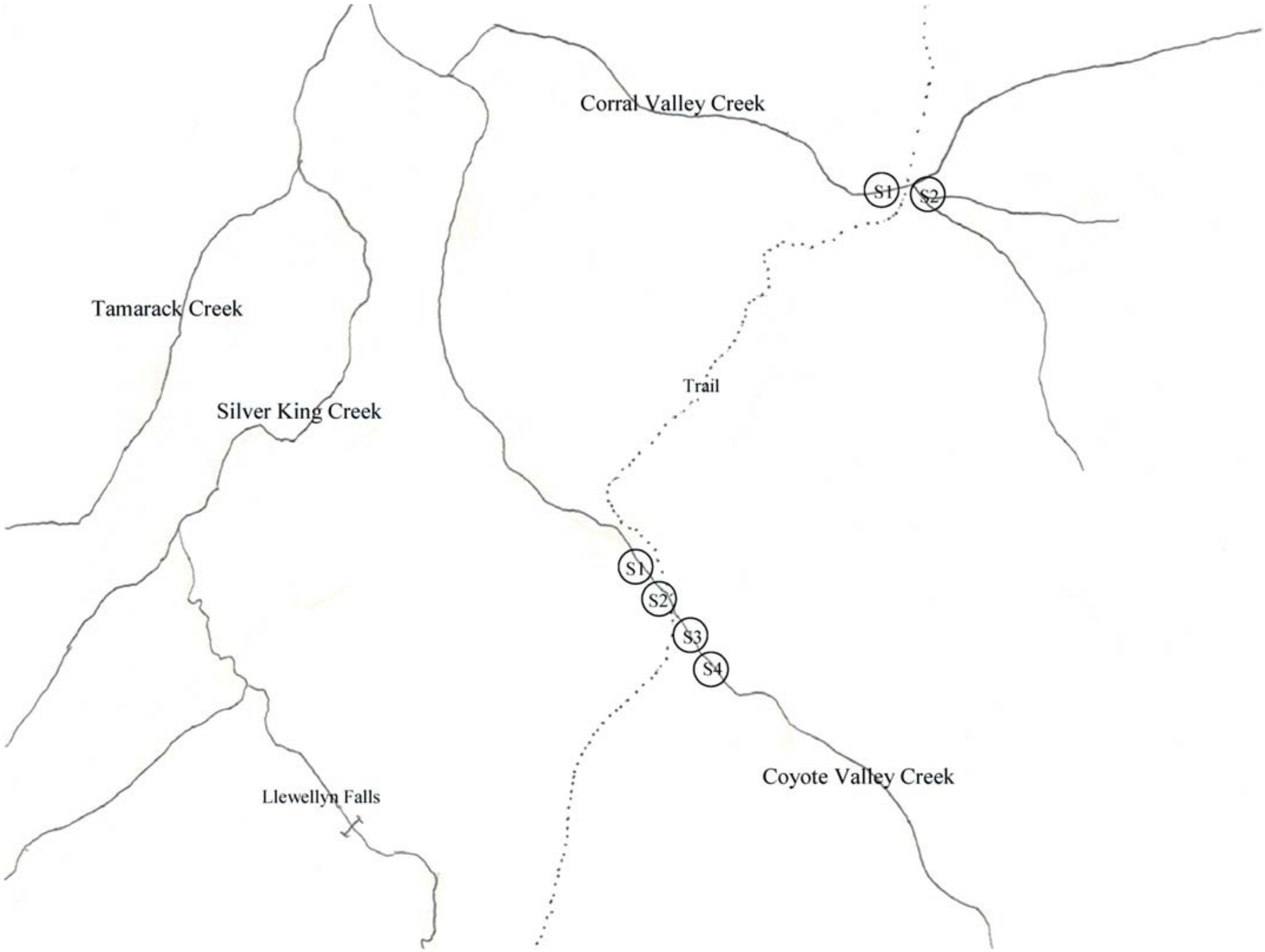

Figure A2. Location of GAWS stations on Corral Valley and Coyote Valley Creeks, Alpine County, California.

Table A2. Summary of sediment samples collected in 1984 and 1990 (Modified from Duff 1991).

Stream	Station	Percent Fines Passing Sieve			
		>6.35 mm	>0.84 mm	>0.21 mm	<6.35 mm
Silver King	S1:610	---	---	---	---
Silver King	S2:640	64.3 (60.3)	21.8 (20.4)	13.9 (19.3)	35.7 (39.7)
Silver King	S3:641	61.8 (53.4)	23.1 (30.8)	15.1 (15.8)	38.2 (46.6)
Silver King	S4:700	57.0 (61.5)	20.2 (21.9)	22.8 (16.6)	43.0 (38.5)
Silver King	S5:725	57.3 (59.0)	25.5 (24.7)	17.2 (16.3)	42.7 (41.0)
Silver King	S6:738	59.3 (68.6)	24.7 (15.7)	16.0 (15.7)	40.7 (31.4)
Silver King	S6A:745	---	---	---	---
Silver King	S7:775	64.8 (57.6)	25.6 (28.0)	9.6 (14.4)	35.2 (42.4)
Silver King	S8:813	60.0 (64.2)	29.0 (25.5)	11.0 (10.6)	40.0 (35.8)
MEAN		**60.6 (60.7)**	**24.3 (23.8)**	**15.1 (15.5)**	**39.4 (39.3)**
Bull Canyon	S1:040	62.9 (61.0)	20.9 (23.5)	16.2 (15.5)	37.1 (39.0)
Bull Canyon	S2:100	---	---	---	---
Fly Valley	S1:500	62.9 (67.9)	24.3 (26.7)	12.8 (5.4)	37.1 (32.1)
Four Mile	S1:250	69.6 (72.4)	20.4 (18.5)	10.0 (9.1)	30.4 (27.6)
Coyote	S1:400	---	---	---	---
Coyote	S2:467	32.2 (41.0)	38.3 (39.5)	29.5 (19.5)	67.8 (59.0)
Coyote	S3:500	44.5 (52.1)	36.5 (31.1)	19.0 (16.8)	55.5 (47.9)
Coyote	S4:542	---	---	---	---
MEAN		**38.4 (46.6)**	**37.4 (35.3)**	**24.3 (18.2)**	**61.7 (53.5)**
Corral	S1:571	51.8 (51.0)	30.4 (32.0)	17.8 (17.0)	48.2 (49.0)
Corral	S2:574	45.7 (46.9)	27.3 (36.0)	27.0 (17.1)	54.3 (53.1)
MEAN		**48.8 (49.0)**	**28.9 (34.0)**	**22.4 (17.1)**	**51.3 (51.1)**

Values outside parentheses represent 1990 data and values inside parentheses represent data from 1984.

Table A3. Summary of macroinvertebrate diversity index (DAT) ratings from 1984, 1987, and 1990. (Modified from Mangum 1991)

Stream	Station	Channel Type	DAT 1984	DAT 1987	DAT 1990
Silver King	S1:610	C3	---	---	---
Silver King	S2:640	C3	24.3	17.8	25.0
Silver King	S3:641	C3	21.7	12.3	20.2
Silver King	S4:700	C3	20.1	19.2	21.1
Silver King	S5:725	C3	17.4	20.5	20.9
Silver King	S6:738	C3	17.5	13.8	17.5
Silver King	S6A:745	C3	---	---	---
Silver King	S7:775	B2/B3	15.6	19.1	20.0
Silver King	S8:813	C3	11.2	18.3	20.3
Bull Canyon	S1:040	C3	---	---	---
Bull Canyon	S2:100	B2	17.8	21.2	---
Fly Valley	S1:500	B2/C2	20.8	17.5	---
Four Mile	S1:250	C3	19.8	21.1	16.4
Coyote	S1:400	C6	14.9	17.1	---
Coyote	S2:467	C3	---	---	---
Coyote	S3:500	C6	17.3	14.9	---
Coyote	S4:542	C3	---	---	---
Corral	S1:571	C3	17.9	18.8	---
Corral	S2:574	C3	---	---	---

Scale	Macroinvertebrate Diversity Index
Excellent	18 - 26
Good	11 - 17
Fair	6 - 10
Poor	0 - 5

Table A4. Summary of macroinvertebrate standing crop data from 1984, 1987, and 1990. (Modified from Mangum 1991)

Stream	Station	Channel Type	Standing Crop g/m^2 1984	Standing Crop g/m^2 1987	Standing Crop g/m^2 1990
Silver King	S1:610	C3	---	---	---
Silver King	S2:640	C3	5.6	1.3	1.1
Silver King	S3:641	C3	3.1	0.9	0.5
Silver King	S4:700	C3	1.0	0.7	1.1
Silver King	S5:725	C3	2.5	4.0	0.5
Silver King	S6:738	C3	2.1	0.8	0.6
Silver King	S6A:745	C3	---	---	---
Silver King	S7:775	B2/B3	1.6	1.1	2.6
Silver King	S8:813	C3	0.9	1.0	1.0
Bull Canyon	S1:040	C3	---	---	---
Bull Canyon	S2:100	B2	0.5	0.8	---
Fly Valley	S1:500	B2/C2	1.3	0.5	---
Four Mile	S1:250	C3	1.8	2.5	1.2
Coyote	S1:400	C6	1.4	1.8	---
Coyote	S2:467	C3	---	---	---
Coyote	S3:500	C6	1.1	1.6	---
Coyote	S4:542	C3	---	---	---
Corral	S1:571	C3	1.3	1.6	---
Corral	S2:574	C3	---	---	---

Scale	Macroinvertebrate Standing Crop
Excellent	4.0 - 12.0
Good	1.6 - 4.0
Fair	0.6 - 1.5
Poor	0.0 - 0.5

Table A5. Summary of macroinvertebrate Biotic Condition Index (BCI) ratings from 1984, 1987, and 1990. (Modified from Mangum 1991)

Stream	Station	Channel Type	BCI 1984	BCI 1987	BCI 1990	BCI Desired
Silver King	S1:610	C3	---	---	---	---
Silver King	S2:640	C3	96	100	100	110
Silver King	S3:641	C3	96	100	100	110
Silver King	S4:700	C3	93	96	100	110
Silver King	S5:725	C3	100	100	98	110
Silver King	S6:738	C3	91	100	89	110
Silver King	S6A:745	C3	---	---	---	---
Silver King	S7:775	B2/B3	88	100	100	110
Silver King	S8:813	C3	93	98	100	110
Bull Canyon	S1:040	C3	88	---	---	---
Bull Canyon	S2:100	B2	---	98	---	110
Fly Valley	S1:500	B2/C2	96	100	---	105
Four Mile	S1:250	C3	106	110	91	115
Coyote	S1:400	C6	93	96	---	110
Coyote	S2:467	C3	93	---	---	110
Coyote	S3:500	C6	98	88	---	110
Coyote	S4:542	C3	---	96	---	110
Corral	S1:571	C3	94	86	---	105
Corral	S2:574	C3	---	---	---	---

Scale	Macroinvertebrate Biotic Condition Index (BCI)
Excellent	> 90
Good	75 - 89
Fair	< 75
Poor	< 75

APPENDIX B. Summary of Threats and Recommended Recovery Actions for the Paiute Cutthroat Trout.

LISTING FACTOR	THREAT	RECOVERY CRITERIA	RECOVERY ACTION NUMBERS
A	Streambank degradation from recreational activities	3	3.1.1, 3.1.2, 3.1.6, 3.2.1, 3.2.2, 3.2.4, 3.3.3
A	Streambank degradation from cattle grazing	3	3.1.1, 3.1.3, 3.1.6, 3.2.1, 3.2.3, 3.2.8, 3.3.1, 3.3.2, 3.3.3
A	Degradation of water quality and spawning substrates by beavers	3	3.1.1, 3.1.4, 3.1.5, 3.1.6, 3.2.1, 3.2.5, 3.2.6, 3.2.8, 3.3.1, 3.3.3
B	Unregulated angling	2, 4	3.1.6, 3.2.4, 3.2.8, 3.3.1, 3.3.3, 4.1.1, 4.2, 6.1, 6.2
C	Natural predators [not currently significant]	Not Applicable	
C	Fungal infections	2	4.3.1, 4.4.1, 4.5.1
D	Potential budgetary constraints on agency commitment to recovery actions	1, 2, 3, 4, 5	4.2, 5.1, 5.2
E	Hybridization and competition with introduced trout	1, 2, 4	1, 2, 4.1.1, 4.1.2, 4.3.2, 4.3.5, 4.4.2, 4.4.5, 4.5.2, 4.5.5
E	Need for fish barriers to prevent upstream migration of introduced trout	1, 2, 4	1, 2, 3.2.7, 4.3.3, 4.3.5, 4.4.3, 4.4.5, 4.5.3, 4.5.5
E	Human introduction of trout	1, 2, 4	1, 2, 4.1.1, 4.1.2, 4.2, 4.3.5, 4.4.5, 4.5.5, 6.1, 6.2
E	Vulnerability to catastrophic events due to limited distribution	2, 3, 4	2, 4.3.1, 4.3.4, 4.4.1, 4.4.4, 4.5.1, 4.5.4, 4.6

Listing Factors:

A. The Present or Threatened Destruction, Modification, or Curtailment Of Its Habitat or Range

B. Overutilization for Commercial, Recreational, Scientific, Educational Purposes (not a factor)

C. Disease or Predation

D. The Inadequacy of Existing Regulatory Mechanisms

E. Other Natural or Manmade Factors Affecting Its Continued Existence

Recovery Criteria:

1. All nonnative salmonids are removed from Silver King Creek and its tributaries downstream of Llewellyn Falls to fish barriers in Silver King Canyon.

2. A viable population occupies all historic habitat in Silver King Creek and its tributaries downstream of Llewellyn Falls to fish barriers in Silver King Canyon.

3. Paiute cutthroat trout habitat is maintained in all occupied streams.

4. The refuge populations in Corral and Coyote Creeks, Silver King Creek, and tributaries above Llewellyn Falls as well as out-of-basin populations are maintained as refugia and are secured from the introduction of other salmonid species.

5. A long-term conservation plan and conservation agreement are developed, which will be the guiding management documents once Paiute cutthroat trout are delisted.

APPENDIX C. Summary of Comments on the Draft Revised Recovery Plan

On January 26, 2004, we released the Draft Revised Recovery Plan for the Paiute Cutthroat Trout for a 60 day public comment period that ended March 26, 2004. We received 14 comment letters from respondents including various governmental agencies, conservation organizations, and private individuals. These comments, where appropriate, have been incorporated into the final revised recovery plan. In addition, we offer the following discussion in the interest of providing a fuller explanation and response to certain specific comments.

Issues:

Historic habitat (Comments 1-6)

1985 Recovery Plan (Comment 7)

Fisheries Management (Comments 8-11)

Fish Barriers (Comments 12-14)

Tamarack Lake (Comments 15-16)

Non-native Fish Removal (Comments 17-27)

Non target Species (Comments 28-35)

General Comments (Comments 36-70)

Historic Habitat

1. **Comment:** Paiute cutthroat trout have been restored already; historic range is above Llewellyn Falls. **Response:** As stated in the Revised Recovery Plan, the best available information supports the location of historic habitat from Llewellyn Falls downstream to barriers in Silver King Canyon and all accessible tributaries within this stream reach.

2. **Comment:** Historic habitat is uncertain; barriers are present which prevent upstream migration into presumed historic habitat. **Response:** See response to Comment 1.

3. **Comment:** The 1985 Plan assumed that the native habitat for Paiute cutthroat trout was above Llewellyn Falls in Silver King Creek because that is the type locality for the subspecies. **Response:** The 1985 Recovery Plan states "The presumed historic distribution of the Paiute cutthroat trout is limited to a short

reach of Silver King Creek below Llewellyn Falls and the accessible reaches of three small tributaries: Tamarack Creek, Tamarack Lake Creek, and the lower 0.4 km of Corral Valley Creek." The Revised Recovery Plan also states this. Neither plan identifies the native habitat of Paiute cutthroat trout as occurring above Llewellyn Falls.

4. **Comment:** Fundamental to the Plan is a claim, now, that the historic habitat of Paiute cutthroat trout is Silver King Creek below Llewellyn Falls. The evidence for this highly speculative claim is based on hearsay and anecdote and is variable and contradictory in the original sources (Ryan and Nicola 1976, Vestal 1947). **Response:** Both the 1985 Recovery Plan and the 2004 Draft Revised Recovery Plan clearly state that the historic range of Paiute cutthroat trout is from Llewellyn Falls downstream to barriers in Silver King Canyon. This claim is documented in a letter from Virgil S. Connell (sheep herder) to Brian Curtis (California Department of Fish and Game) on August 8, 1944 who writes "Above the falls we found there were no fish." Later he writes "During the year 1912, a young Basque Joe Jaunsaras was herding for me, and while fishing just below the falls, and catching more than he wanted, he put some in a can and carried them above the falls." We have no reason to believe that Mr. Connell was not telling the truth. The only contradictory issue documented in Ryan and Nicola (1976) that we could find is the date Paiute cutthroat trout were stocked above Llewellyn Falls. In contrast to Ryan and Nicola, Ashley (1970) stated that the 1912 transplant was a failure and Paiute cutthroat trout were replanted in 1924. We agree with the evidence and conclusion presented in Ryan and Nicola (1976), that Paiute cutthroat trout were successfully transplanted above Llewellyn Falls in 1912 and that the historic range of Paiute cutthroat trout is from Llewellyn Falls downstream to barriers in Silver King Canyon. The historic habitat for Paiute cutthroat trout is also reported in Behnke (1979).

5. **Comment:** There is no reason and no new scientific information to alter the conclusion given in the 1985 Recovery Plan that stated "The issue of what constitutes the native range is complicated by the paucity of early collection records and the conflicting recollections of early observers." (1985 Recovery Plan, p. 7). Therefore, the type locality above Llewellyn Falls must be accepted as the historic range of Paiute cutthroat trout. **Response:** The 1985 Recovery Plan and this final Revised Recovery Plan conclude that the best available information supports the location of historic habitat from Llewellyn Falls downstream to barriers in Silver King Canyon and all accessible tributaries within this stream reach.

6. **Comment:** How could Paiute cutthroat trout have existed in such a limited length of stream for perhaps thousands of years; but now, occupying twice as much stream and in five times as many drainages, it is at risk from catastrophic events?

Response: All existing populations are isolated in headwater drainages which make them susceptible to catastrophic events (Dunham *et al*. 2003, Reiman *et al*. 2003). Paiute cutthroat trout will always be susceptible to stochastic events because of their limited range. When Paiute cutthroat trout are repatriated throughout their historic range, they will be less susceptible than the out-of-basin populations due to the size of the drainage, the size of the population, and the quality and distribution of habitat in which it evolved.

<u>1985 Recovery Plan</u>

7. **Comment:** The 1985 Plan states "At what point or condition can the species be considered recovered?" which is answered "When a pure population of Paiute cutthroat trout has been reestablished in Silver King Creek above Llewellyn Falls, and the integrity of the habitats in Silver King Creek, North Fork Cottonwood Creek, and Stairway Creek has been secured and maintained over a consecutive five-year period with stable or increasing overwintering populations of 500 or more adult fish in each of these streams". These conditions have been met.
   Response: Most of the objectives of the 1985 Recovery Plan have been accomplished. However, the 1985 Recovery Plan did not address recovery in terms of restoring Paiute cutthroat trout into its historic range because the barriers in Silver King Canyon had not been investigated. The 2004 Draft Revised Recovery Plan addressed new information concerning these natural barriers. California Department of Fish and Game, Forest Service, and U.S. Fish and Wildlife Service personnel have investigated the existence of natural fish barriers in Silver King Canyon and have concluded that these barriers prevent fish from migrating upstream into Paiute cutthroat trout habitat. These barriers are the likely downstream extent of historic habitat as they form a barrier that isolates Paiute cutthroat trout from historic Lahontan cutthroat trout habitat. Therefore, this final Revised Recovery Plan incorporates this new information to address recovery in terms of restoring Paiute cutthroat trout into their historic habitat, from Llewellyn Falls downstream to Silver King Canyon.

<u>Fisheries Management</u>

8. **Comment:** The Plan is a marketing scheme to permit fishing for threatened trout.
   Response: The purpose of the Revised Recovery Plan is to identify actions that are needed, and criteria that must be met, for the recovery and delisting of Paiute cutthroat trout as a threatened species under the Endangered Species Act.

9. **Comment:** The plan fails to address the root causes of risks to Paiute cutthroat trout, namely the stocking of nonnative trout in any part of the watershed in which

stocked fish can eventually migrate into existing and historic habitat. There can be no long-term restoration of native fish as long as fish stocking by fish and game agencies and U.S. Fish and Wildlife Service continues in the drainages of concern. **Response:** The California Department of Fish and Game has not stocked nonnative trout in the Silver King Canyon drainage since the early 1950's and there is no plan to stock in the future.

10. **Comment:** It is disturbing that the reason Paiute cutthroat trout were downlisted to facilitate management and allow for regulated angling, the Endangered Species Act was not established to promote fishing. **Response:** Paiute cutthroat trout, Lahontan cutthroat trout, and Apache trout were all downlisted to threatened status at the same time and for the same reasoning (U.S. Fish and Wildlife Service 1975). It was determined that regulated angling could be used as a management tool for these species if streams became overpopulated, and take was authorized in accordance with applicable State law. However, since downlisting occurred, fishing has not been allowed in upper Silver King Creek or North Fork Cottonwood Creek, which are the two most accessible Paiute cutthroat trout occupied streams.

11. **Comment:** Opening Silver King Creek above Llewellyn Falls to angling, now closed, will increase the risk to Paiute cutthroat trout of hybridization. Paiute cutthroat trout now exist in the stream section below Llewellyn Falls because some fish go over the falls and on the barrier on Coyote and Corral creeks and are available for anglers to catch in the lower section of Silver King Creek below Llewellyn Falls which is presently open to angling. The unique experience of catching Paiute cutthroat trout in their native drainage is provided currently. Does the U.S. Fish and Wildlife Service see no contradiction in recommending fishing above Llewellyn Falls where the population is claimed to be finally secured? If the stream reach below Llewellyn Falls is converted to a monospecific population of Paiute cutthroat trout, it will always be at risk of introductions of non-native fish into any tributaries above Llewellyn Falls or into Corral Valley or Coyote Valley Creeks anyway. There is no reason to assume that non-native fish could only be introduced into the most accessible area. **Response:** Very few fish go over the falls as supported by the fact that electrofishing efforts have not found Paiute cutthroat trout below the falls. A highly regulated fishery has been discussed above Llewellyn Falls during the non-native fish eradication efforts to offset any public fishing lost during this time. This would not occur unless the Paiute cutthroat trout population is large enough to support fishing and a stream guard (Action 4.1.1) is in place to monitor fishing activity. The California Department of Fish and Game is responsible for the management of Paiute cutthroat trout. Any regulations relating to angling would be subject to the 4(d) provisions of the Endangered Species Act.

Fish Barriers

12. **Comment:** Construction of barriers within wilderness areas contradicts the intent of the Wilderness Act. **Response:** No barrier construction or removal is planned in any wilderness area. Recovery Action 3.2.7 describes construction of a fish barrier in North Fork Cottonwood Creek which is not in a designated wilderness.

13. **Comment:** The success of the current plan-that is, restoring and maintaining genetically pure Paiute cutthroat trout in the lower portion of the Silver King Creek drainage-is highly dependent on whether or not these "potential barriers" will in fact prevent movement of other native and introduced trout from portions of the drainage below Silver King Canyon. The Plan assumes that reinvasion can be prevented by monitoring the identified natural barriers or establishing artificial instream barriers. It appears the assumption that reinvasion of lower Silver King Creek is unlikely needs to be qualified as "uncertain" (even setting aside the possibility of human reintroduction). If stream barriers have limited effectiveness in preventing fish migration, the plan fails to address effectiveness of barriers at all flow conditions. **Response:** California Department of Fish and Game, Forest Service, and U.S. Fish and Wildlife Service personnel have documented the existence of natural fish barriers in Silver King Canyon, and surveys support the conclusion that these barriers are effective in stopping fish from migrating upstream into historic Paiute cutthroat trout habitat regardless of the flow conditions.

14. **Comment:** Permanent barriers of some kind would have been necessary for the genetic isolation of the precursor of Paiute cutthroat trout. Once isolated, Paiute cutthroat trout evolved. The barriers in Silver King Canyon are apparently not large enough for this original isolation or the U.S. Fish and Wildlife Service would not be recommending that they be inspected and reinforced to prevent upstream migration of other non-native fish. **Response:** The Plan does not recommend reinforcing natural barriers in Silver King Canyon. Periodic monitoring of the natural barriers is needed to document effectiveness and any changes in the barriers due to stochastic events.

Tamarack Lake

15. **Comment:** Treatment of Tamarack Lake is not necessary for Paiute cutthroat trout recovery because any non-native fish in Tamarack Lake are isolated from the Silver King Creek drainage by barriers. The poisoning of Tamarack Lake is

uncalled for and should be dropped from the Plan. **Response:** Barriers are present and are effective in stopping fish from moving upstream into Tamarack Lake. However, it is necessary to treat Tamarack Lake because of the possible existence there of non-native trout, which have been stocked in the past and could migrate downstream into Silver King Creek during high flow events.

16. **Comment:** Alternatives such as gill netting in Tamarack Lake should be considered. **Response:** Alternatives, including gill netting, were analyzed through the National Environmental Policy Act process (Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion) and it was determined that treating Tamarack Lake was the most effective method for achieving recovery goals.

<u>Non-native Fish Removal</u>

17. **Comment:** The Plan calls for stream poisoning, which is unsound and will create ecological havoc on amphibians and macroinvertebrates. The Plan should address other alternatives to poisoning. **Response:** An Environmental Assessment has been completed for the rotenone treatment. Furthermore, the U.S. Fish and Wildlife Service, California Department of Fish and Game, and the Humboldt-Toiyabe National Forest have been planning for the reintroduction of Paiute cutthroat trout into its historic range for many years. All methods for the removal of non-native fish have been discussed at length and analyzed for all biological resources through the National Environmental Policy Act process in the Forest Services' Environmental Assessment and through section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion).

18. **Comment:** Plan fails to examine effects of poisoning on amphibians. **Response:** The effects of individual recovery actions, including use of rotenone, were analyzed through the National Environmental Policy Act process (Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion).

19. **Comment:** The recommended killing of endangered amphibians through poisoning is a far cry from the euphemistic statement "there may be some negative impacts on amphibians if they are not captured during the relocation process or through stress of handling". Since the plan recommends killing imperiled amphibians and disrupting of aquatic food webs via poisoning, it is incorrect to state "all Paiute cutthroat trout recovery actions were evaluated to minimize adverse impacts to mountain yellow-legged frog and Yosemite toad." **Response:** The Plan does not recommend killing amphibians through this conservation effort.

Surveys will be performed to relocate amphibians throughout the treatment area to outside of the treatment area. During amphibian surveys in 2003, no mountain yellow-legged frogs were found. Approximately 12 toad tadpoles were found and relocated outside the treatment area into suitable habitat. The lakes in Silver King Creek drainage will remain fishless for the ultimate benefit of amphibians.

20. **Comment:** Use of rotenone conflicts with wilderness values. **Response:** The Wilderness Act of 1964 allows for activities within wilderness boundaries when it involves the protection and propagation of threatened and endangered species. Section 4(b) of the Wilderness Act and House Report 98-40, which supplements the California Wilderness Act of 1984, establishing the Carson-Iceberg Wilderness, specifically states that "certain wildlife management activities, designed to enhance or restore fish populations, are permissible and often desirable in wilderness areas to aid in achieving the goal of preserving the wilderness character of the area". Guidelines for managing fish and wildlife in wilderness are found in Forest Service Manual 2323.3. This direction allows for the use of chemical treatments to prepare waters for reestablishment of indigenous, threatened or endangered, or native species, or to correct undesirable conditions caused by human influence.

21. **Comment:** The plan provides no evaluation or determination of success criteria or the purpose and need for subsequent chemical treatments. Success and re-treatment criteria need to be established based on feasibility and risk assessment. **Response:** The success criterion is described in action 1 "Chemically treat Silver King Creek to remove all introgressed fish downstream from Llewellyn Falls to barriers in Silver King Canyon." Two to three treatments are often necessary to remove all non-native fish from a certain stream segment.

22. **Comment:** Several issues relating to the adverse effects of rotenone use. **Response:** The Revised Recovery Plan provides actions that are needed for the recovery of Paiute cutthroat trout. The effects of individual recovery actions were analyzed through the National Environmental Policy Act process (Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion). This analysis concluded that the use of rotenone was the most effective method of achieving recovery goals. It also documented that the use of rotenone will be done in compliance with all applicable laws and regulations.

23. **Comment:** The Plan states that "Chemically treating Silver King Creek to remove all introgressed fish" jumps the gun by stating that chemicals would be used to eradicate nonnative fish when this has not been determined through a National Environmental Policy Act Environmental Assessment or Environmental Impact

Report/Environmental Impact Statement process. It also reveals the preconceived bias that poisons are the only effective means of achieving the goals of the project. **Response:** See response to Comment 17.

24. **Comment:** The Plan is an attempt to justify another large stream poisoning project in a Wilderness Area for the purpose of establishing a monospecific sportfishery for Paiute cutthroat trout that will be part of a California Department of Fish and Game angling contest for "heritage" trout. **Response:** See response to Comment 8.

25. **Comment:** The Plan fails to show why poisoning 11 miles of streams, springs, and a lake would benefit either Paiute cutthroat trout or the many other nontarget species that would be affected and endangered by this project. **Response:** The plan clearly states that recovery of Paiute cutthroat trout will require repatriation into historic habitat. To accomplish this task, nonnative salmonids, which are the most serious threat to Paiute cutthroat trout, must be eradicated from Llewellyn Falls downstream to Silver King Canyon. Amphibians will benefit by the removal of nonnative fish in Tamarack Lake within the Silver King Creek drainage.

26. **Comment:** The Draft Plan concludes that reintroduction to "native habitat" (below Llewellyn Falls) will somehow "substantially reduce these extinction threats." This reasoning is flawed and is constructed merely to justify another poisoning project in Silver King Creek for other purposes. **Response:** The purpose of this project is to reduce the threat of nonnative trout to Paiute cutthroat trout by removing nonnative trout from its historic habitat so Paiute cutthroat trout can then be reintroduced into Silver King Creek which will create a more secure population through connecting the existing isolated populations. These actions are necessary for the recovery and delisting of Paiute cutthroat trout as a threatened species under the Endangered Species Act.

27. **Comment:** There is no recognition in this Draft Plan that poisoning is a major habitat disturbance that can have long reaching and permanent effects on nontarget species and food supplies which are a component of habitat. **Response:** Effects to nontarget species are expected to be short-term due to the concentration of chemicals used, exposure time, and untreated adjacent and upstream habitat which will provide source populations for recolonization. The effects of individual recovery actions were analyzed through the National Environmental Policy Act process (Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion).

Non Target Species

28. **Comment:** Opposes any action that may contribute to the need to federally list the mountain yellow-legged frog and Yosemite toad. **Response:** As stated in the Revised Recovery Plan, once treated, Tamarack Lake will remain fishless for the benefit of amphibians. Whitecliff Lake, outside the treatment area, will also remain fishless. Additionally, amphibian surveys will be conducted in the project area. If any are found, they will be transported out of the project area prior to treatment. By ensuring that conservation measures are used, these projects should not lead to the need to federally list the mountain yellow-legged frog or Yosemite toad.

29. **Comment:** Macroinvertebrate and amphibian monitoring is inadequate. **Response:** The adequacy of macroinvertebrate and amphibian monitoring has been analyzed through both the National Environmental Policy Act process and section 7(a)(2) of the Endangered Species Act. Amphibians in the drainage have been specifically surveyed by the California Department of Fish and Game in 2001, as well as incidentally in the course of Paiute cutthroat trout surveys, and the treatment area will be surveyed for amphibians prior to rotenone application. Amphibian surveys will continue as part of post-treatment monitoring. A detailed macroinvertebrate sampling plan, including control sites and pre- and post-treatment surveys, was included as an appendix to the Environmental Assessment and was accepted by the Lahontan Regional Water Quality Control Board as adequate to address their concerns and meet Basin Plan Objectives.

30. **Comment:** Evidence clearly demonstrates that multiple rotenone treatments and livestock grazing have decimated amphibian populations in the project area and there is no evidence that native Paiute cutthroat trout have less impact on amphibians than nonnative fish. **Response:** The project area has never been treated with rotenone and we are unaware of any information that documents the decimation of amphibian populations in this area due to rotenone treatments. Livestock grazing has occurred in the area for over 100 years and probably has had an effect on amphibians, however, we have little information to support this. The only location where mountain yellow-legged frogs in Silver King Creek currently exist is above Llewellyn Falls, where they are coexisting with Paiute cutthroat trout.

31. **Comment:** Various issues relating to the adverse effects of recovery actions to macroinvertebrates. **Response:** The Revised Recovery Plan provides actions that are needed for the recovery of Paiute cutthroat trout. Macroinvertebrate sampling is a component of these actions. The effects of individual recovery actions have been analyzed through the National Environmental Policy Act process

(Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion).

32. **Comment:** The U.S. Fish and Wildlife Service, the agency responsible for endangered species, should have analyzed effects to nontarget species in this Draft Plan. Instead the Draft Plan is a myopic, single species approach to increasing numbers of one species for sport fishing. It was never the intent of the Endangered Species Act to conduct recovery projects to increase single species that would put other species at risk of extinction. **Response:** The Revised Recovery Plan provides actions that are needed for the recovery of Paiute cutthroat trout. Addressing the effects of these actions to nontarget species is not within the scope of this recovery plan. The effects of individual recovery actions, including effects to nontarget species, have been, and will continue to be analyzed through the National Environmental Policy Act process (Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion).

33. **Comment:** The Draft Plan makes no effort to assess the cumulative impacts to such species as the mountain yellow-legged frog, Yosemite toad, willow flycatcher, the yellow warbler, and hundreds of other species of all this poisoning being conducted in other nearby watersheds or of all the past poisoning in the Silver King drainage or in many other watersheds across the Sierra. **Response:** Addressing the cumulative impacts of recovery actions to other species is not within the scope of this recovery plan. However, these impacts were analyzed through the National Environmental Policy Act process (Environmental Assessment) and section 7(a)(2) of the Endangered Species Act (Biological Assessment, and Biological Opinion).

34. **Comment:** On the whole, this proposed management plan, far from benefiting native amphibians, will only further deteriorate their habitat in several locations. One of the more misleading statements in this Draft Plan is the sentence on p. 9 that "the long-term effects of removal of nonnative and hybrid fish will be beneficial to native amphibians." We know of no studies that show Paiute cutthroat trout are less an impact on amphibians than are other trout. **Response:** Recovery actions include the removal of nonnative trout from Tamarack Lake to prevent their reintroduction downstream into Silver King Creek during high flow events. Mountain yellow-legged frogs historically occupied this lake. Although the exact reason for their disappearance is unknown, it is believed that the stocking of nonnative trout into this once fishless lake contributed to their disappearance. There are no plans to introduce Paiute cutthroat trout into Tamarack or Whitecliff Lakes. Therefore, it is anticipated that a secondary benefit of removing nonnative trout from Tamarack Lake, and managing Tamarack and Whitecliff Lakes to remain fishless, will contribute to the conservation of amphibians.

35. **Comment:** The assumption that "because this recovery plan is partially focused on habitat improvements, it also provides conservation benefits for two candidate species, the Sierra Nevada population of the mountain yellow-legged frog and the Yosemite toad" (Plan at 48) requires substantially more robust supporting analysis than the Plan currently offers. **Response:** See response to Comment 34.

<u>General Comments</u>

36. **Comment:** Have potential impacts from the 1997 flood and/or other natural events that have occurred since the last habitat monitoring was conducted been investigated in the proposed recovery areas, and has it been determined that the fish barriers are still effective? **Response:** Numerous trips into Silver King Canyon by California Department of Fish and Game, Forest Service, and U.S. Fish and Wildlife Service personnel after the 1997 flood have documented that the natural fish barriers in Silver King Canyon are still effective in stopping fish from migrating upstream into historic Paiute cutthroat trout habitat.

37. **Comment:** The project area is projected to reopen to angling after one to three years of treatment. What criteria and methods will be used to monitor the Paiute cutthroat trout populations and protect the restored populations from declining due to even limited fishing pressure? **Response:** The California Department of Fish and Game is responsible for the management of Paiute cutthroat trout. Any regulations relating to angling would be subject to the 4(d) provisions of the Endangered Species Act and state laws and regulations. Permitted recreational fishing will not occur unless the population is stable enough to sustain such an activity and a stream guard is in place to monitor fishing.

38. **Comment:** Have alternative plans been investigated to ensure the removal project is a success and that the required monitoring occurs in the event budgetary constraints limit the funding and/or staff available? **Response:** The U.S. Fish and Wildlfie Service, California Department of Fish and Game, and the Forest Service have all committed to the recovery of Paiute cutthroat trout and will continue to request appropriate funding to carry out recovery actions. The levels of funding which are actually authorized for these efforts on a yearly basis are appropriated by Congress and are beyond agency control.

39. **Comment:** Any use of rotenone by California Department of Fish and Game must conform to the Water Quality Control Plan for the Lahontan Region and Memorandum of Understanding requirements signed by California Department of Fish and Game and the Board. **Response:** Reporting requirements signed by California Department of Fish and Game and the Board were incorporated into the

Terms and Conditions in our Biological Opinion for this project. The project will not be carried out without the appropriate permits from these and other agencies.

40. **Comment:** The collaboration of agency efforts should have resulted in a consistent and supportive group of documents that reflects awareness for all inherent subjects of this recovery plan (wilderness, amphibians, and public interests). **Response:** We believe there is consistent and supporting documentation throughout the documents relating to the Revised Recovery Plan, National Environmental Policy Act compliance (Environmental Assessment), section 7(a)(2) of the Endangered Species Act compliance (Biological Assessment and Biological Opinion), and California State Environmental Quality Act compliance.

41. **Comment:** The California Department of Fish and Game Negative Declaration and Forest Service Environmental Assessment outlined the proposed use of rotenone without a final recovery plan in place. **Response:** There is no legal requirement for a recovery plan to be finalized prior to the implementation of a project which contributes to the recovery of a species.

42. **Comment:** Service should be a part of a joint Environmental Impact Statement/Environmental Impact Report. **Response:** The Forest Service and California Department of Fish and Game are the primary action agencies and have both completed their respective public environmental documentation and review.

43. **Comment:** How does removing fish from the historic range reduce the likelihood of fish being transplanted from elsewhere, like downstream of Silver King Canyon? **Response:** Currently, nonnative fish occur just below Llewellyn Falls. From this location it would be relatively easy to move fish above the falls into occupied habitat. Although the threat of reintroduction will always exist, it is anticipated that this threat will be minimized if nonnative fish are removed because they would have to be transported from below Silver King Canyon. This is a long and treacherous canyon which should make it difficult to transport fish through the canyon to Paiute cutthroat trout occupied habitat.

44. **Comment:** Water quality monitoring is inadequate. **Response:** The adequacy of water quality monitoring has been analyzed through both the National Environmental Policy Act process and section 7(a)(2) of the Endangered Species Act. Additionally, the California Regional Water Quality Control Board, Lahontan Region (LRWQCB) has reviewed the California Department of Fish and Game's proposal and found that the Basin Plan Objectives would be met (LRWQCB July 3, 2003, letter to California Department of Fish and Game).

45. **Comment:** Critical habitat has not been designated for the Paiute cutthroat trout which is a required element under the Endangered Species Act. **Response:** Pursuant to the 1978 amendment of the Endangered Species Act, we are required to designate critical habitat in conjunction with the listing of a species if we find critical habitat to be prudent and determinable. The original rule listing Paiute cutthroat trout as endangered in 1967 did not address the prudency and determinability of critical habitat, because these requirements were not in place at that time. Our ability to designate critical habitat is also practically constrained by availability of staff time and funding to do so. However, any decisions regarding designation of critical habitat are separate and distinct from the process of developing a recovery plan for the species.

46. **Comment:** It is illogical to state "genetic diversity could be subjected to additional severe bottlenecks due to inadequate population size" given that Paiute cutthroat trout evolved as a small population and now inhabit more stream miles than it did historically. **Response:** Populations of Paiute cutthroat trout have gone through severe bottlenecks since they first evolved from Lahontan cutthroat trout. The out-of-basin populations were created through the introduction of very few individuals. The Paiute cutthroat trout does occupy more stream miles than it did historically; however, they are all isolated and there is no gene flow between these isolated populations.

47. **Comment:** There is common doubletalk about restoring genetically pure trout species, where arbitrary standards of purity (usually 99-100 percent) are deemed essential, while at the same time saying genetic diversity is just as essential. **Response:** Genetic purity and diversity are two separate yet equally important components to recovery. Genetic purity refers to genes from nonnative species, such as rainbow trout, being found within the population of Paiute cutthroat trout. If a fish has both Paiute cutthroat trout and rainbow trout genes it is no longer a pure Paiute cutthroat trout but is referred as a hybrid. Genetic diversity refers to the genetic variability of genes within the pure population of Paiute cutthroat trout and is an important component to the long-term viability of the species.

48. **Comment:** The fact that Paiute cutthroat trout were able to evolve, survive, and thrive in 9 miles of stream brings into question the argument that it requires restoration of hundreds and hundreds of stream miles to prevent the extinction of other cutthroat trout species. **Response:** It is not within the scope of the Revised Recovery Plan to address the recovery needs of other subspecies of cutthroat trout.

49. **Comment:** The U.S. Fish and Wildlife Service fails to acknowledge that our own activities, such as stocking of nonnative trout and applying fish poisons, contribute to extinction threats of Paiute cutthroat trout. **Response:** The stocking of

nonnative trout was identified as a threat factor in the listing of Paiute cutthroat trout. Although stocking no longer occurs in Silver King Creek, recovery cannot be achieved without removing the threat of nonnative fish.

50. **Comment:** The plan fails to acknowledge or analyze the potential effects of global warming on Paiute cutthroat trout survival. **Response:** It is possible that climatic change could affect habitat suitability for Paiute cutthroat trout in the future. However, at this time we are unable to predict the likelihood or significance of such consequences with sufficient confidence to analyze their site-specific effects on the species.

51. **Comment:** Was an Environmental Assessment completed for electrofishing surveys which also harms fish and amphibians? **Response:** The California Department of Fish and Game is responsible for the management of Paiute cutthroat trout. Fisheries management, including electrofishing surveys, is subject to the 4(d) provisions of the Endangered Species Act.

52. **Comment:** The Plan states that "there should be no degradation of habitat from existing conditions due to anthropogenic effects". The U.S. Fish and Wildlife Service seems to have a blind spot when it comes to recognizing that its own activities degrade the environment and cause anthropogenic effects. **Response:** The effects of the recovery actions were analyzed through the National Environmental Policy Act and Endangered Species Act processes to determine if these effects are significant and will result in an unacceptable level of adverse impacts to the species. The analysis concluded that the level of impacts would not preclude the recovery of the species.

53. **Comment:** The poisoning plan, restocking, and subsequent sport fishery offer no new protections for Paiute cutthroat trout. **Response:** The removal of nonnatives and subsequent restocking of Paiute cutthroat trout into their historic range will significantly reduce the threat of competition and hybridization from nonnative salmonids. In addition, increasing occupied habitat will reduce the threat of stochastic events (such as fire or flooding) that may occur in the Silver King Creek drainage. The California Department of Fish and Game is responsible for the management of Paiute cutthroat trout. Any regulations relating to angling would be subject to the 4(d) provisions of the Endangered Species Act and state laws and regulations and would not be allowed until certain population criteria were met.

54. **Comment:** The third reason in the Draft Plan for action below Llewellyn Falls is the risk of bottlenecks. These bottlenecks in Paiute cutthroat trout populations are already present. Analysis of Paiute cutthroat trout genetic markers all concluded

that bottlenecks are present in the remaining populations (Israel et al. 2002, Nielsen and Sage 2002). **Response:** The actions in the Revised Recovery Plan seek to maximize the genetic diversity of existing populations (Recovery actions 4.3, 4.3.4, 4.3.5, 4.4, 4.4.4, 4.4.5, 4.5, 4.5.4, 4.5.5) and to minimize the risks from genetic bottlenecks that have occurred since Paiute cutthroat trout first evolved from Lahontan cutthroat.

55. **Comment:** There is confusion in the Draft Plan about hybrid crosses between Paiute cutthroat trout and rainbow trout versus Paiute cutthroat trout and Lahontan cutthroat trout. The Draft Plan states "genetic analysis indicates that Corral Valley Creek now contains pure Paiute cutthroat trout (Israel et al. 2002)." But the Israel et al. (2002) report states "None of the loci screened showed fixed differences between Paiute cutthroat trout and Lahontan cutthroat trout." And in summary it states "Additionally, molecular markers that can distinguish Lahontan cutthroat trout and Paiute cutthroat trout would provide another tool for determining this relationship." Clearly, the Israel et al. (2002) study did not separate Lahontan cutthroat trout from Paiute cutthroat trout. **Response:** The term pure Paiute cutthroat trout refers to the lack of rainbow trout genes. Other studies have documented genetic differences between Paiute cutthroat trout and Lahontan cutthroat trout (Nielson and Sage 2002). Further genetic testing is identified in the Revised Recovery Plan to clarify this issue (Recovery actions 4.3.2, 4.4.2, 4.5.2).

56. **Comment:** The Israel et al. (2002) study even casts doubt on the genetic separation of Paiute cutthroat trout from rainbow trout: "Upon examination of the SCN evidence it does not appear that any population has undergone recent hybridization with rainbow trout; however, introgression from past hybridization events may be difficult to detect when relying on a single genetic marker." **Response:** Israel *et al.* (2002) does not cast doubt on the genetic separation between Paiute cutthroat trout and rainbow trout. Paiute cutthroat trout evolved from Lahontan cutthroat trout. The quote is referring to the fact that one genetic marker was used and that other markers should be explored. Further genetic testing is identified in the Revised Recovery Plan (Recovery actions 4.3.2, 4.4.2, 4.5.2). Using the best available information, there is no evidence of hybridization with rainbow trout in the existing Paiute cutthroat trout populations.

57. **Comment:** The discussion of fish abundances in Upper Fish Valley is confusing and redundant with a later section (Draft Plan pg.3). **Response:** We agree, that section has been removed from the plan.

58. **Comment:** If the purpose of citing the various numbers of fish is to build a case for some needed number of fish, then the values presented are misleading. The Draft Plan leaves it to the reader to add up miles of stream, numbers of fish per

mile, mean number of fish, and locations. **Response:** The purpose of citing various numbers of fish is to give the reader an idea of the size of the existing populations and how they fluctuate from year to year, not to build a case for a required number of fish. The values cited for recovery were taken from peer reviewed literature (Hilderbrand and Kershner 2000) and may need to be revised as additional information becomes available.

59. **Comment:** Without criteria for the meaning of stable, the goal is meaningless and has little utility with a highly fluctuating population. There are enough data points to consider that a population is stable if it is within +/- 2 standard deviations or the 95 % confidence interval of the long term mean. **Response:** Recovery Criterion 2 has been changed slightly and expanded upon to clarify our intentions.

60. **Comment:** In spite of the long time the U.S. Fish and Wildlife Service and California Department of Fish and Game have been managing fish in the Silver King Basin, nearly the only reported data on age and growth is from studies done in the North Fork of Cottonwood Creek where Paiute cutthroat trout are nonnatives. The U.S. Fish and Wildlife Service and California Department of Fish and Game have no data on number of age classes, yet the U.S. Fish and Wildlife Service has selected a rule for judging recovery based on age classes. **Response:** We based this recovery criterion on the best available information in the scientific literature. As stated in the Revised Recovery Plan, the numbers and size of fish required for recovery may need to be revised as additional information becomes available.

61. **Comment:** The definition for a population of at least 2,500 fish > 75mm in length, has not been separately reported for any population in the Silver King drainage, and no rationale has been presented for its choice. There is inconsistency with the size of adult fish. Adult fish are defined as > 150 mm in Silver King Creek and >137 mm in Stairway Creek. Which is it? **Response:** The rationale for the selection of this value was taken from peer reviewed literature (Hilderbrand and Kershner 2000) and may need to be revised as additional information becomes available. Adult fish are defined using various sizes because they grow at different rates in the separate drainages in which they exist.

62. **Comment:** It is difficult to tell if the U.S. Fish and Wildlife Service population goal of at least 2,500 fish applies to the total of the separate populations in the Silver King drainage or to each of the separate populations. **Response:** The goal is to have 2,500 fish greater than 75 mm in length occupying the historic range from Llewellyn Falls downstream to Silver King Canyon. As stated in the Revised Recovery Plan, this number is a preliminary estimate and may be revised as additional information becomes available.

63. **Comment:** The lack of any habitat condition assessment for the last 14 years belies any genuine agency interest in this subspecies. Habitat was a key criterion to recovery of the Paiute cutthroat trout ("Habitat and population trends will be closely monitored" 1985 Plan). But even this critical management goal seems to have been abandoned. **Response:** Habitat was monitored in 1984, 1987, and 1990 (Table 3, Revised Recovery Plan). Nonnative fish removal and subsequent reintroduction of Paiute cutthroat trout into Silver King Creek above Llewellyn Falls in the 1990's took priority over habitat monitoring due to budget and staffing limitations. Habitat monitoring is an important component of the recovery actions for Paiute cutthroat trout (Recovery criterion 3).

64. **Comment:** "The primary threat to Paiute cutthroat trout is hybridization with nonnative trout" (Draft Plan p.49). That threat will remain no matter how large an area the Paiute cutthroat trout occupies. **Response:** We agree that this threat will always remain, however, through education and the removal of nonnative fish throughout its historic range, we believe this threat can be minimized to a level that will allow for a viable population of Paiute cutthroat trout throughout its historic range (Recovery action #6).

65. **Comment:** The Draft Plan does not recognize that a threatened trout species outside its native habitat is a nonnative species and has as much an impact as any other nonnative species. **Response:** We agree that any species outside its native habitat can be considered a nonnative species. Amphibian monitoring will continue in occupied habitat outside of Paiute cutthroat trout historic habitat to evaluate and address any adverse impacts of Paiute cutthroat trout to amphibian species.

66. **Comment:** The apparent lack of baseline information makes it difficult to assess the impacts of past, current, and proposed recovery efforts; the gathering of such information should be a priority. Data needs to be developed to demonstrate that physical, chemical, and biological processes are examined and interactions in the Silver King Creek drainage are understood. **Response:** Ongoing data collection will continue for the purposes of recovery of Paiute cutthroat trout. Data collection will include but will not be limited to water quality, macroinvertebrates, amphibians, riparian and stream habitat quantity and quality, and Paiute cutthroat trout population dynamics.

67. **Comment:** Various predictions of success appear unrealistic given that four decades of fish removal activities were needed to eliminate introduced trout from upper portions of Silver King Creek drainage, that no systematic assessment as to the ecosystem impacts of these previous treatments has occurred (nor is proposed), that project proposals do not even concede the possibility of cumulative impacts

within Silver King Creek drainage from past and ongoing poisonings, and that the measure of success to date has been limited to the removal of all fish and the reestablishment of genetically pure Paiute cutthroat trout rather than the recovery of the Paiute cutthroat trout and the physical, biological, and chemical functionality of its habitat. **Response:** Recovery plans must address the known threats to the species which may include threats to the genetic viability of the species as well as the habitat which supports it. We believe the Revised Recovery Plan provides a multifaceted approach which addresses the various threats to this species. The cumulative impacts of any recovery action is addressed through separate processes including those relating to the National Environmental Policy Act and the Endangered Species Act.

68. **Comment:** The Plan fails to establish metrics of land use pattern and practice (e.g., quantitative and qualitative standards, guidelines, and goals) and for watershed condition that corresponds with maintenance and recovery of habitat condition sufficient for persistence of Paiute cutthroat trout and other at-risk species within the Silver King Creek drainage. **Response:** The primary threat to Paiute cutthroat trout habitat in the Silver King Creek drainage has been the destruction and degradation of habitat through improper livestock grazing management. Grazing has not been allowed since 1995 and the habitat condition has improved dramatically. It is anticipated that the allotment will remain closed to grazing and the habitat will continue to improve which will benefit both Paiute cutthroat trout and other at-risk species such as the mountain yellow-legged frog and Yosemite toad. This supports Recovery Criterion 3, which requires no degradation of habitat from existing conditions due to anthropogenic effects.

69. **Comment:** The Plan should identify specific variables to describe habitat condition, including threshold criteria for suitable and high quality habitat. **Response:** Recovery Action 3.1.1. requires the institution of a stream and riparian habitat monitoring program which uses an established stream monitoring protocol with measurable and repeatable methods. Most of the currently occupied and historic Paiute cutthroat trout habitat is in relatively good condition. The Revised Recovery Plan states that habitat should have no degradation from existing conditions due to anthropogenic effects.

70. **Comment:** The Plan does not emphasize ecosystem recovery. **Response:** We believe the Revised Recovery Plan does emphasize ecosystem recovery by the continued rest from livestock grazing, the eradication of nonnative salmonids and subsequent repatriation of Paiute cutthroat trout, and keeping the high elevation lakes in the drainage fishless for the benefit of the mountain yellow-legged frog and Yosemite toad.

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EFFECTS OF ANTHROPOGENIC FRAGMENTATION AND LIVESTOCK GRAZING ON WESTERN RIPARIAN BIRD COMMUNITIES

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Abstract. Deciduous vegetation along streams and rivers provides breeding habitat to more bird species than any other plant community in the West, yet many riparian areas are heavily grazed by cattle and surrounded by increasingly developed landscapes. The combination of cattle grazing and landscape alteration (habitat loss and fragmentation) are thought to be critical factors affecting the richness and composition of breeding bird communities. Here, we examine the influence of land use and cattle grazing on deciduous riparian bird communities across seven riparian systems in five western states: Montana, Idaho, Nevada, Oregon and California. These riparian systems are embedded in landscapes ranging from nearly pristine to almost completely agricultural. We conducted landscape analysis at two spatial scales: local landscapes (all land within 500 m of each survey location) and regional landscapes (all land within 5 km of each survey location). Despite the large differences among riparian systems, we found a number of consistent effects of landscape change and grazing. Of the 87 species with at least 15 detections on two or more rivers, 44 species were less common in grazed sites, in heavily settled or agricultural landscapes, or in areas with little deciduous riparian habitat. The Veery (*Catharus fuscescens*), Song Sparrow (*Melospiza melodia*), Red-naped Sapsucker (*Sphyrapicus nuchalis*), Fox Sparrow (*Passerella iliaca*), and American Redstart (*Setophaga ruticilla*) were all less common under at least three of these conditions. In contrast, 33 species were significantly more common in one or more of these conditions. Sites surrounded by greater deciduous habitat had higher overall avian abundance and 22 species had significantly higher individual abundances in areas with more deciduous habitat. Yet, areas with more agriculture at the regional scale also had higher total avian abundance, due in large part to greater abundance of European Starling (*Sturnus vulgaris*), American Robin (*Turdus migratorius*), Brown-headed Cowbird (*Molothrus ater*), and Black-billed Magpie (*Pica pica*), all species that use both agricultural and riparian areas. Grazing effects varied considerably among riparian systems, but avian abundance and richness were significantly lower at grazed survey locations. Fifteen species were significantly less abundant in grazed sites while only five species were more abundant therein. Management should focus on (1) preserving and enlarging deciduous habitats, (2) reducing cattle grazing in deciduous habitats, and (3) protecting the few relatively pristine landscapes surrounding large deciduous riparian areas in the West.

Key Words: agriculture; avian abundance and richness; cattle grazing; landscape fragmentation; multi-scale; riparian habitat.

Deciduous riparian areas bordering rivers and streams in the western United States support a higher density of breeding birds than any other habitat type (Carothers and Johnson 1975, Rice et al. 1983, Ohmart and Anderson 1986), and studies explicitly comparing deciduous riparian areas with surrounding upland communities repeatedly have found diversity and density of breeding birds to be greater in riparian communities (Carothers et al. 1974, Johnson et al. 1977, Stamp 1978, Conine et al. 1979, Hehnke and Stone 1979, Knopf 1985; Anderson et al. 1985a,b; Strong and Bock 1990, Cubbedge 1994). The importance of these habitats to the maintenance of avian communities cannot be overemphasized. Deciduous riparian habitat makes up less than 1% of the western land area (Knopf et al. 1988), yet over 50% of western bird species breed primarily or exclusively in deciduous riparian communities (Johnson et al. 1977, Mosconi and Hutto 1982, Johnson 1989, Saab and Groves 1992, Dobkin 1994). Due to the proliferation of dams, intensive water management practices, and the effects of domestic livestock, riparian areas are considered the most heavily degraded ecosystems in the West (Rosenberg et al. 1991, Dobkin 1994, Ohmart 1994, Saab et al. 1995); some western states have already lost as much as 95% of their historic riparian habitat (Rosenberg et al. 1991, Ohmart 1994). The importance of remaining riparian areas for avian and other wildlife populations is thus greatly magnified.

Two of the primary threats to the quality of remaining deciduous riparian habitats are the conversion of land near riparian areas into agricultural and urban land (Tewksbury et al. 1998, Saab 1999), and cattle grazing within riparian areas (Carothers 1977, Crumpacker 1984, Chaney et al. 1990, Saab et al. 1995, Saab 1998). The effects of these activities on individual rivers have often been studied using different metrics, focusing on different groups of birds, and there have been few attempts to combine data across riparian systems to look for common patterns (Hochachka et al. 1999).

Although it is widely recognized that the richness and composition of breeding bird assemblages are at least partially dependent on the landscape within which they are embedded (Robinson et al. 1995a; Donovan et al. 1995b, 1997; Freemark et al. 1995, Faaborg et al. 1995, Saab 1999), it is not clear what scale or scales are appropriate to use when considering the effects of landscapes on bird populations (Freemark et al. 1995, Donovan et al. 2000). Indeed, given the many factors that can affect the structure of bird communities (nest predation, brood parasitism, competition for food and nesting sites, habitat area limitations), landscapes likely affect bird communities at multiple scales (Wiens 1989, 1995; Urban et al. 1987, Turner 1989, Kareiva 1990, Kotliar and Wiens 1990, Barrett 1992, Andrén 1995, Freemark et al. 1995, Hansson et al. 1995). To date, however, few empirical studies have considered the relative importance of multiple landscape scales (but see Tewksbury et al. 1998, Hochochka et al. 1999, Saab 1999, Donovan et al. 2000), and there has been no attempt to examine the relative effects of multiple land-uses across scales when studying the composition of riparian bird communities.

A focal concern in the western United States is cattle grazing. Domestic cattle graze 70% of the land area in the 11 western states (Crumpacker 1984) causing extensive modifications to vegetation (Holechek et al. 1989). These effects are particularly apparent in deciduous riparian areas (Carothers 1977, Crumpacker 1984, Platts and Nelson 1985, Fleischner 1994, Saab et al. 1995). However, it is not clear which grazing effects are dependent on local factors and levels of grazing intensity, and to what extent grazing effects can be generalized across a broad array of riparian systems and grazing regimes.

Here we examine the influence of regional (within 5 km of each study site) and local (within 500 m of each study site) landscapes and the influence of cattle grazing on the richness and relative abundance of bird communities in seven riparian systems dominated by deciduous trees and shrubs. This work is the result of collaboration by five independent research teams working in five western states over the past decade. By combining efforts, we provide the first meta-analysis of human-induced landscape change and cattle grazing on the avian communities breeding in these critical western habitats in the hope of detecting consistent patterns across the West.

METHODS

Riparian Systems, Survey Locations, and Landscape Characterization

The seven riparian systems included in this work vary considerably in size, physical character, local and regional vegetation patterns, and land use (Fig. 1; Appendix 1), but all possess streamside vegetation dominated by woody deciduous species (see Appendix 1 for detailed descriptions of each riparian system).

We analyzed bird species-abundance data from a total of 437 survey locations (Fig. 1; Table 1). Survey locations were separated by at least 150 m and located in vegetation dominated by cottonwood (*Populus* spp.), aspen (*Populus tremuloides*), or a mixture of species including willow (*Salix* spp.), valley oak (*Quercus lobata*), dogwood (*Cornus* spp.), hawthorn (*Crataegus* spp.), cherry (*Prunus* spp.), alder (*Alnus* spp.), and birch (*Betula* spp.). At each survey location, relative abundance was calculated as the total number of each species detected per visit. Surveys were either fixed-radius point counts (five of the seven systems) or 150-m fixed-width line transects (Table 1). We defined a survey as a single visit to a point or transect location. All studies conducted three surveys per year. The radius of point counts was either 40 m or 50 m, and point duration was either five or 10 min (Table 1).

We defined two spatial scales at each study location: regional landscapes (all land within 5 km of each survey location = 7,854 ha) and local landscapes (all land <500 m of each survey location = 78 ha). Regional landscape character was quantified using state GAP databases (Scott et al. 1993) derived from satellite images (Table 1). Local landscape data were gathered from low elevation aerial photography, ortho-photo quadrangle maps, and high resolution digital data, depending on the riparian system. Using a different data set for local analyses allowed us to include smaller features in analyses, such as linear riparian components and individual buildings that could not be detected at the regional scale. Metrics such as average patch size and edge-to-interior ratios depend on mapping resolution, and our data resolution varied considerably among sources (Table 1). Thus we confined our analyses to the percent cover of four landscape components: forest cover, agriculture, human habitation, and deciduous riparian cover. The first three have been used previously to index landscape fragmentation and habitat conversion (Donovan et al. 1995b, 1997; Robinson et al. 1995a, Young and Hutto 1999). Deciduous riparian cover also has been used in landscape studies. Percent cover blends aspects of patch size and isolation, both of which have been found to affect riparian bird communities (Brown and Dinsmore 1986, Gibbs et al. 1991, Craig and Beal 1992, Saab 1999).

Our decision to compare high-resolution local data with low-resolution regional data also reflects the choice available to land managers, where detailed land-use data are available only at local scales. This approach, however, confounds differences in resolution with differences in scale. Therefore, on three riparian systems (Sacramento, San Joaquin, and Bitterroot rivers), we compared GAP data (used for the regional scale) with aerial photography data (used at the local scale) on the same 500 m local landscapes to examine correlations between estimates derived from different

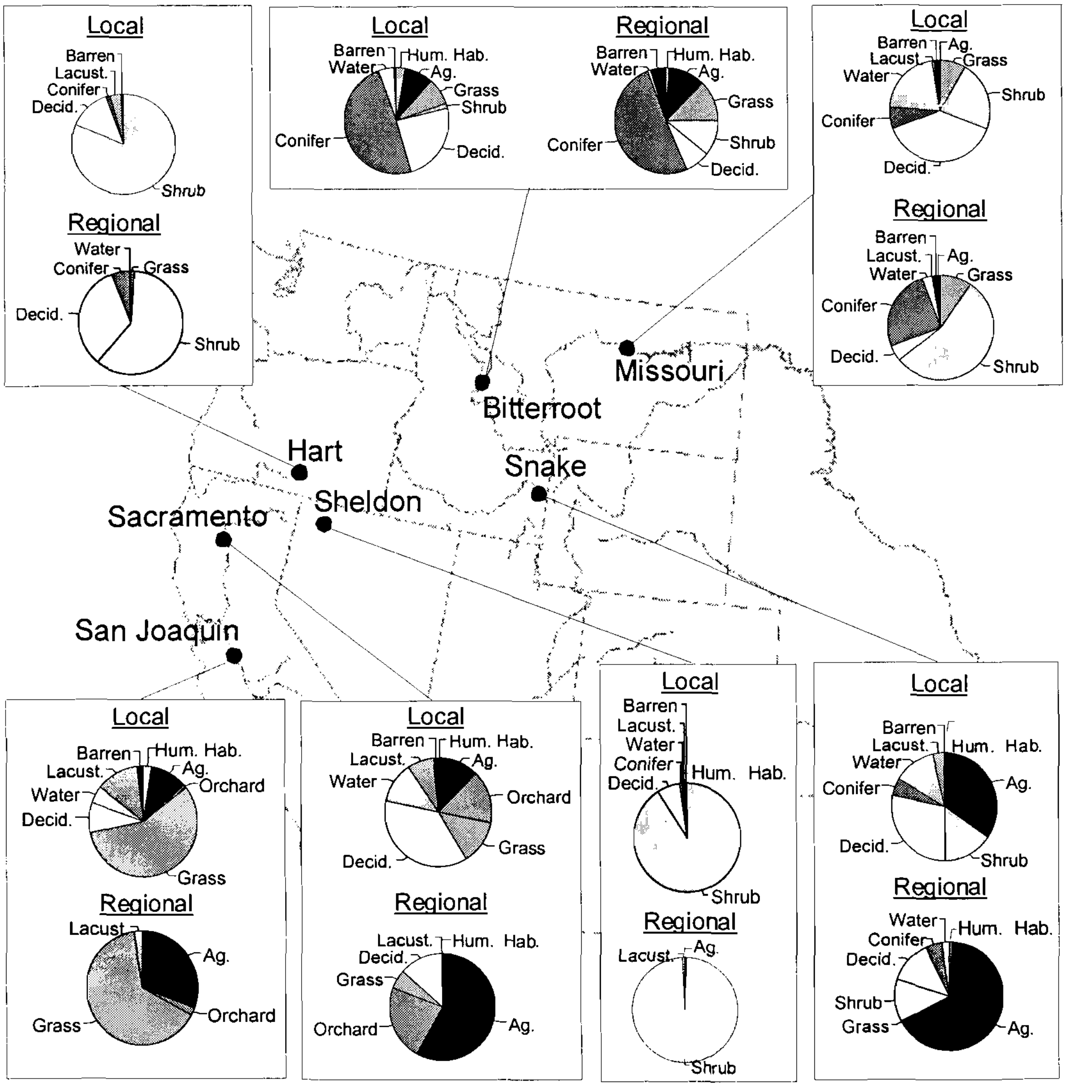

FIGURE 1. River system locations and general landscape character of each river system. Pie charts are mean percent cover for each landscape component averaged across all survey locations, at both local and regional scales. Hum. Hab. = all human habitations, including houses, farms, commercial developments, and industrial areas. Ag. = all agriculture, including row crops and land used for pasture and row crop, but excluding vineyards and orchards. Orchard = all orchards, primarily fruit and nut trees, and vineyards. Grass = all grasslands. Shrub = all shrublands and juniper woodlands, as bird communities were similar. Decid. = all deciduous habitats. Conifer = Conifer forests. Water = all large bodies of water, including river channels. Lacust. = Lacustrine, partially submerged and wet meadow habitat. Barren = permanent snow, ice, rock, or talus.

data types. For the Bitterroot River, the resolution of GAP data is quite high (Table 1), so we expected some concordance between the two techniques. For the Sacramento and San Joaquin Rivers, the GAP resolution is low, and this shift in resolution could affect results considerably. Because the regional scale contains 100 times the area of the local scale, however, lower resolution at the regional landscape scale should have less effect than lower resolution at the local scale.

Livestock Grazing

In five of the seven riparian systems studied, grazing occurred on some but not all of the study sites. Within these five systems, the intensity and timing of grazing differed considerably, from the Missouri River with long term high-intensity grazing on grazed sites and no cattle on rested ("ungrazed") sites for the past 30 years, to the Snake River where grazing intensity dif-

TABLE 1. RIVER SYSTEMS, DATA TYPES, AND SAMPLE SIZES

River system	State	Bird survey type	Duration/length	Years	Sites	Survey locations	Local landscape: Landscape data source	Local landscape: Minimum mapping unit	Regional landscape: Landscape data source	Regional landscape: Minimum mapping unit
Sacramento	CA	Point count	5 min	1993–1997[a]	10	55	CWIS[g]	900 m^2	California GAP	100 ha
San Joaquin	CA	Point count	5 min	1995–1997[b]	6	54	CWIS[g]	900 m^2	California GAP	100 ha
Snake	ID	Point count	10 min	1991–1994[c]	46	148[e]	Aerial photos, Orthophoto Quads.	~650 m^2	Idaho GAP	2 ha, 0.81 ha in riparian
Bitterroot	MT	Point count	10 min	1995–1997[d]	38	120	Aerial photos, Orthophoto quads.	~650 m^2	MT GAP	2 ha, 0.81 ha in riparian
Missouri	MT	Point count	10 min	1998	9	29	MT GAP	2 ha, 0.81 ha in riparian	MT GAP	2 ha, 0.81 ha in riparian
Sheldon	NV	Transect	150 m long	1991 & 1993	5	10[f]	Aerial photography	~650 m^2	Nevada GAP	100 ha
Hart Mountain	OR	Transect	150 m long	1991 & 1993	7	21[f]	Aerial photography	~650 m^2	Western U.S. GAP[h]	100 ha

[a] Surveys conducted on seven sites (58 points) from 1993–1997, surveys conducted on one site (11 points) from 1994–1997, surveys conducted on one site (three points) from 1995 to 1997, and surveys conducted on one site (nine points) from 1996–1997.
[b] Surveys conducted on one site (15 points) from 1995–1997, surveys conducted on four sites (39 points) from 1996–1997, and surveys conducted on one site (nine points) in 1997 only.
[c] Two surveys at each location in 1991, three at each location in all other years.
[d] Surveys conducted on 16 sites (78 points) from 1995–1997, survey conducted on 22 sites (29 points) in 1996 only.
[e] Bird data were provided for each site (averaged across all points on a site).
[f] Surveys are strip transects (see text) run both in 1991 (grazed) and 1993 (ungrazed) and analyzed separately.
[g] California Wetlands Inventory System map of the Central Valley. Map was classified by the California Department of Fish and Game (1997) from spring and fall 1992/1993 30m satellite images. Available on-line at: http://ceres.ca.gov/wetlands/geo_info/cal_wetland_riparian.html.
[h] The Western GAP is an unreleased GAP cover combining all GAP maps in the western United States; Source: Idaho GAP Lab.

fered considerably among sites and was often moderate or light (Appendix 1). The methods of comparison differ as well; in the Hart Mountain and Sheldon systems, the same sites were surveyed in 1991 and 1993, the first and third growing seasons following cessation of long term livestock grazing. We considered the 1991 surveys "grazed" and the 1993 surveys rested. In all other riparian systems, bird abundance was compared in the same years among different locations, rather than in the same locations among different years. Given all these differences, we expected to find great variation among riparian systems in the effects of grazing, and any consistent effects should represent general effects applicable to a wide variety of riparian ecosystems in the West.

ANALYSIS

Relative abundance data were available for each point count or transect survey except on the Snake River, where data were averaged to the study site level. To accommodate this, we performed analyses at the site level for all ripariau systems, and at the survey location level for all areas except the Snake. Both methods gave similar results. However, combining data to the site level resulted in a considerable loss of statistical power, so we present analysis of the survey location data for all rivers except the Snake, which is analyzed at the study site level. Our analysis of species richness includes all areas except the Snake because average richness per survey location could not be calculated from the data available.

All variables were initially screened for deviations from normality using one-sample Kolmogorov-Smirnov tests (Sokal and Rohlf 1995), and transformed where necessary. We used square-root transformations for count data (bird variables), and arcsine square-root transformations for percent data (landscape components). We examined four landscape components—human habitation, agriculture, deciduous forest, and coniferous forest—each at local and regional landscape scales.

Within each riparian system, we examined the effects of landscape differences on the relative abundance of all individual species detected an average of 15 or more times per year on that riparian system. Because we were primarily interested in effects that can be generalized throughout western riparian areas, we limited our analysis to species meeting this criterion on at least two riparian systems (102 species in total). In addition, we examined community level effects by grouping species into different guilds: primary hosts of Brown-headed Cowbirds (see Appendix 2 for scientific names of all species) vs. non-hosts; and long-distance migrants vs. short-distance migrants vs. permanent residents. In examining the effects of grazing, we also divided species into open nesting species vs. primary and secondary cavity nesting species, and low vs. high nesting species. Relative abundance of each species is defined as the average number of individuals detected per survey calculated by averaging values for separate visits within a year and then averaging across years. We also examined overall ricbness, calculated as the cumulative number of species detected at each location over the three surveys within a single year, averaged across years.

Migratory status followed Sauer et al. (2000). Primary hosts included all species listed as common or frequent cowbird hosts in *The Birder's Handbook* (Ehrlich et al. 1988); species listed as uncommon or rare cowbird hosts were termed secondary hosts (not analyzed in this manuscript). For nest height, we used the mean nest height from nesting studies on the riparian systems in this study, and examined the effect of grazing on the abundance of birds nesting below 2.5 m and above 5 m (Appendix 2).

To control for the large differences in methods among riparian systems, we first tested the effects of each landscape component within each riparian system to maintain consistency in sampling. To assess landscape effects on the avian community, we regressed total relative abundance, richness, and the relative abundance of each avian guild against each of the landscape components at both local and regional scales, using all survey locations within each riparian system for each river-specific analysis. To test for grazing effects we used t-tests within each riparian system, comparing community metrics and individual species between grazed and ungrazed sites. We assumed equal variance among population means unless $P < 0.1$ iu Levene tests for equality of variance. Because these analyses are based on overall relative abundance of all species in a guild, the results are heavily influenced by the most common species. To examine landscape and grazing effects on community metrics with all species receiving equal weight, as well as to determiue the response of individual species to differences in landscapes, we designated each survey location as low (lower 25%), middle (25 to 75%) or high (upper 25%) with respect to each landscape component within each riparian system. For tests of landscape effects on overall abundance, and the effects of landscapes and grazing on each guild, we coded each species as either more or less abundant in the low sites when compared to the high sites, then used binomial tests to determine if a significant majority of species within each guild were significantly more abundant in tbe high or low sites. For analysis of individual species, we used Mann-Whitney U-tests to compare the abundance of species in low and high sites for each landscape component within each riparian system and to compare abundance in grazed vs. ungrazed sites. We tested all species on a given riparian system with an average of 15 or greater detections per year. As our purpose was to evaluate the consistency of landscapes and grazing effects across rivers, we limit our results to species tested in at least two riparian systems. This analysis controls for landscape differences among different riparian systems because it compares abundances of birds across the landscape extremes within each riparian system.

To examine landscape and grazing effects across riparian systems, we used Fisher's combined probabilities test (Fisher 1954, Sokal and Rohlf 1995). This test evaluates the P-values from each riparian system against the null hypothesis that there is no general trend of significance across tests (in this case, riparian systems). Tbe value -2 times the sum of the natural logs of all the P values from a group of independent tests of a single hypothesis falls along a cumulative Chi-square distribution with $2k$ degrees of freedom,

TABLE 2. CORRELATIONS AND MEAN DIFFERENCE (1 SE) BETWEEN LANDSCAPE COMPONENTS IDENTIFIED USING HIGH RESOLUTION LOCAL LANDSCAPE DATA AND LOWER RESOLUTION GAP DATA (USED FOR THE REGIONAL SCALE ANALYSIS) BOTH AT THE LOCAL SCALE

	Human habitation		Agriculture		Deciduous riparian		Coniferous forest	
	r	Diff (%)[b]	r	Diff (%)[b]	r	Diff (%)[b]	r	Diff (%)[b]
Bitterroot	0.20*	−5.4 (0.7)	0.78***	−9.0 (1.3)	0.76***	−6.3 (1.2)	0.97***	11.6 (1.0)
Sacramento	—[a]	−1.2 (0.2)	−0.23	5.9 (5.4)	0.11	0.2 (5.7)	—	—
San Joaquin	—[a]	−2.9 (0.3)	0.17	0.8 (3.9)	−0.07	7.6 (3.6)	—	—

Note: * $P < 0.05$, ** $P < 0.01$, *** $P < 0.005$.
[a] Lower resolution data-source picked up no human habitation.
[b] % difference = % component at regional scales (low resolution) − % component at local scales (high resolution).

where k = the number of separate tests (riparian areas) being compared. The combined probabilities test evaluates where the summed value lies along the cumulative Chi-square distribution. Because we are comparing the significance of tests for a general trend in one direction, but trends may be either positive or negative, we had to account for the sign associated with each P value. To do this, we used $-ln$ P for all results whose significance referred to a test opposite in sign from that being evaluated. We evaluated trends in both directions. This procedure produced a more conservative test for an overall pattern across riparian systems, as it is more difficult to reject the null hypothesis of no general effect. Using Fisher's combined probabilities tests also circumvents the problems of combining data with inherent differences in detection probabilities resulting from differences in survey techniques and observers. To determine the most abundant species across river systems, we ranked the abundance of all species within each river system in descending order, and computed mean abundance ranks for all species across rivers (a mean abundance rank of one would mean a species had the highest detection frequency in all rivers it occurred in).

To correct for inflation of significance due to multiple testing, we used sequential Bonferroni adjustment of significance (Rice 1989) for all correlation, regression, and t-tests. Thus for tests of landscape effects, we corrected for a total of 64 tests within each riparian system (four landscape components, two scales, and eight bird community components). We also corrected for 64 tests when examining the significance of the combined probabilities tests across riparian systems. For grazing effects, we corrected for 12 tests (one for each aspect of the bird community examined).

RESULTS

For all studies combined, 180 species were detected across 437 survey locations. Eleven species were detected on all seven river systems. These species, in order of mean abundance rank (lower ranks being more abundant) were the Brown-headed Cowbird, with a mean abundance rank of 7.2; American Robin, 13.7; House Wren, 14.6; Yellow Warbler, 16.1; European Starling, 17.9; Black-headed Grosbeak, 18.9; Bullock's Oriole, 21.3; Mourning Dove, 22.1; Warbling Vireo, 24.1; Brewer's Blackbird, 29.4; and Lazuli Bunting, 30.1. Of the 87 species tested individually for effects of landscape components and grazing, 44 species were significantly less common either in grazed areas, areas with high human habitation or extensive agriculture, or areas with less deciduous riparian habitat; 33 species were more common under these conditions.

CORRELATIONS AMONG LANDSCAPE COMPONENTS AND BETWEEN DATA RESOLUTIONS

Correlations among landscape components varied considerably among riparian systems, depending on the landscape context within which each stream or river was embedded (Fig. 1). Not surprisingly, both within and between scales, the strongest correlations were found where the four components we examined—human habitation, agriculture, deciduous area, and coniferous forest—dominated the landscape (e.g., Snake and Bitterroot rivers), as opposed to landscapes dominated by shrub or grass (Appendix 3). Landscape components varied considerably in their correlations across scales. Relatively homogeneous and broad land uses, such as agriculture, were always correlated positively across scales, whereas clumped and small land-uses, such as human habitation, were correlated weakly across scales in most riparian systems (Appendix 3). Differences in data resolution also affected correlations across scales. When we controlled for scale and compared both local (high resolution) and regional (low resolution) data at the local scale, we found strong positive correlations on the Bitterroot River (Table 2), where regional analysis was relatively fine grained (Table 1). Even with this higher resolution regional data (minimum mapping unit = 2 ha), however, smaller landscape components were underemphasized compared with dominant landscape components (Table 2). Where regional data were coarse-grained, as on the Sacramento and San Luis rivers, correlations were not significant, and differences had high variance because components identified with the high-resolution local data were either missed entirely, or overemphasized by the low resolution landscape data.

Human Habitation

At local scales, the majority of all species (62% ± 5% SE, five rivers) had lower relative abundances in areas with high human habitation compared to areas with low human habitation. This trend was particularly apparent in long-distance migrants (66% ± 6% less abundant in areas with high human habitation, five rivers). These relationships were significant for both groups in binomial tests, but because the Brown-headed Cowbird, Yellow Warbler, and the Black-headed Grosbeak (all very common species) were more abundant in areas with high human habitation, there was no relationship between the total number of detections of all species, or detections of long-distance migrants, vs. local human habitation (Table 3). Human habitation was strongly and positively correlated with the number of Brown-headed Cowbirds detected at both scales (Table 3), and the number of non-host species detections was higher in areas with higher regional human habitation, due primarily to the greater abundance of European Starlings, House Wrens, and American Robins in more densely settled areas (Table 4). The five species showing the greatest reduction in frequency in regional landscapes with high proportions of human settlement were Yellow-rumped Warbler, MacGillivray's Warbler, Warbling Vireo, Swainson's Thrush, and Dusky Flycatcher (Table 4). Populations of each of these species are highly vulnerable to cowbird parasitism (Tewksbury et al. 1998).

Agriculture

High abundances of abundant species such as American Robins, Yellow Warblers, and Brown-headed Cowbirds in areas with agriculture (Table 4) led to highly significant positive relationships between total and guild detection frequency and the amount of agriculture at both scales. However, binomial tests for direction of change of all species in each guild were not significant (Table 3; 53% ± 6% of species had higher abundance in areas with more agriculture), and the only river system to show a significant majority of species increasing with regional agriculture was the Bitterroot (Appendix 4). In addition, regional agriculture was significantly, positively correlated with the abundance of Brown-headed Cowbirds, which were twice as abundant in areas with high proportions of agriculture compared with areas with low proportions of agriculture. Primary hosts, although not related to agriculture at the local scale, showed a strong positive relationship with the amount of agriculture regionally. This positive trend was driven almost entirely by Yellow Warblers, the most abundant host. Yellow Warblers were detected far more often in areas with greater amounts of agriculture and human habitation. In contrast, many less abundant cowbird host species, such as Swainson's Thrush, Warbling Vireo, MacGillivray's Warbler, and Yellow-rumped Warbler, were rarely detected at survey locations with high regional agriculture (Table 4). Overall, there was no indication that the majority of hosts were more or less abundant in landscapes dominated by agriculture (Table 3; Appendix 4).

Non-hosts showed a strong positive relationship with agriculture at both scales (Table 3), primarily due to higher abundances of American Robins, House Wrens, European Starlings, Tree Swallows, and Bullock's Orioles in areas with greater proportions of agriculture (Table 4). The effects of human habitation and agriculture appear similar; in total, 24 species were significantly more abundant in areas with high local or regional agriculture, and 17 of these species were also significantly more abundant in areas with high human habitation.

Deciduous Riparian

Across riparian systems, areas with more deciduous riparian habitat tended to have greater avian abundance and diversity. Fifteen species were significantly more abundant in areas with a high proportion of deciduous habitat at the local scale; six of these species were present in at least four riparian systems: Yellow Warbler, Black-headed Grosbeak, Song Sparrow, Western Wood Pewee, Cedar Waxwing, and Orange-crowned Warbler. Only two species were significantly less abundant in areas with greater local deciduous riparian habitat, MacGillivray's Warbler and Townsend's Warbler. Effects at the regional scale were similar (Tables 3 and 4), though almost half of the individual species increasing were different from those increasing at the local scale.

The amount of local deciduous riparian habitat was positively correlated with virtually all avian guilds at both scales. Binomial tests were less convincing of a significant overall effect, where the only significant relationship was between all species and regional deciduous riparian habitat (Table 3; 57% of species ± 4.3%, five rivers). The lack of significant effects in binomial tests at the local scale was caused primarily by effects on the Sacramento River, where greater local deciduous riparian habitat was associated with lower detection frequencies in 67% of all species (Appendix 4).

Coniferous Forest

At the local scale, the proportion of coniferous forest was not significantly related to total relative abundance, richness, or any guild examined, after correcting for multiple tests. However, at the regional scale, conifer cover had a strong negative effect on cowbird abundance (combined $P < 0.001$). Cowbirds were detected only half as often at survey locations with high conifer forest when compared to locations with low conifer forest (Table 4). Coniferous cover was also related negatively to the abundance of non-hosts, driven primarily by the low abundance of European Starlings, American Robins, and House Wrens in sites with high coniferous cover. In addition, long-distance migrant abundance was associated positively with percent conifer forest (Table 3), due primarily to many more detections of Warbling Vireo, MacGillivray's Warbler, Townsend's Warbler, Violet-green Swallow, and Fox Sparrow in areas with more conifers (Table 4). Binomial tests agreed in direction with regressions on total guild abundance, but were non-significant across rivers, showing considerable variation in results among individual rivers (Appendix 4).

GRAZING

The majority of all species (63% ± 5%) were less abundant in grazed locations (Fig. 2A; combined probabilities test $\chi^2 = 42.8$, $P < 0.001$). After correcting for multiple tests, six species were significantly less abundant at grazed survey locations when all riparian systems were considered, while no species were significantly more abundant at grazed locations (Table 5). In addition, total relative abundance was significantly lower in grazed areas (Fig. 2B; combined probabilities test $\chi^2 = 48.9$, $P < 0.001$), and species richness showed a non-significant trend to be lower in grazed areas (Fig. 2C; combined probabilities test $\chi^2 = 19.8$, $P = 0.01$, not significant after correction for multiple tests). The intensity of grazing effects varied greatly among the seven riparian systems. On the Missouri, Sacramento, and Hart systems, 68–73% of species were less abundant in grazed areas (Fig. 2A; binomial tests, P's < 0.007). The Missouri showed the most dramatic effects, with 13 species significantly less abundant in grazed areas and only one more abundant (Appendix 5), and the average detections per count shifted from 36 on ungrazed survey locations to 21 on grazed survey locations. In contrast, on the Snake and Sheldon riparian systems, species were no more likely to be less or more abundant in these areas (Fig. 2A). On the Sheldon, only two species differed significantly between recently grazed and ungrazed sites, with one species more abundant in each condition (Appendix 5).

Cowbird abundances were not significantly different between grazed and ungrazed locations for any of the five large riparian systems (Fig. 3A). Total primary cowbird hosts, however, were less abundant in grazed areas (Fig. 3B; combined $\chi^2 = 25.3$, $P = 0.005$), with strong effects on the Missouri River ($t = 3.3$, $P = 0.003$) and the Snake River ($t = 3.2$, $P = 0.002$; Appendix 5). While the majority of host species were less abundant on grazed sites in all river systems except the Sheldon, the low number of species in the guild precluded significant effects (Fig. 3C). On the Missouri River, the effects of grazing on hosts was driven primarily by lower abundance of Red-eyed Vireo, American Redstart, Lazuli Bunting, Least Flycatcher, and Yellow Warbler in grazed areas (Appendix 5). Lazuli Buntings and Yellow Warblers were also significantly less abundant in grazed sites along the Snake River, as were Veerys and Song Sparrows (Appendix 5). Total non-host abundance showed no consistent response to grazing pressure (Fig. 3D; combined probabilities test $\chi^2 = 11.3$, $P = 0.33$), but the proportion of species that were more abundant in ungrazed systems was typically higher than expected by chance (Fig. 3E; combined probabilities test $\chi^2 = 20.0$, $P = 0.023$).

Of the migratory guilds, long-distance migrants were the only group significantly less abundant in grazed areas (Total abundance Fig. 4A: combined probabilities test $\chi^2 = 47.7$, $P < 0.001$; binomial mean response Fig. 4B: combined probabilities test $\chi^2 = 26.4$, $P = 0.003$). Across all riparian systems, five of the ten species with significantly lower relative abundances in grazed areas were long-distance migrants (Table 5). The lower relative abundance of long-distance migrants in grazed areas was particularly apparent on the Missouri River, where the average number of long-distance migrants was 21 individuals per survey in ungrazed areas and only 12 per survey in grazed areas (Fig. 4A), and 84% of the species were less abundant in grazed sites (Fig. 4B). In addition to large effects on the Missouri, long-distance migrants were significantly less abundant in grazed sites on the Sacramento ($t = 2.1$, $P = 0.037$), and exhibited similar non-significant trends in both Hart Mountain and Snake River systems ($P = 0.07$ and 0.18, respectively). Residents showed no significant differences between grazed and ungrazed sites for any of the riparian systems (Fig. 4C and 4D). The total abundance of short-distance migrants tended to be lower in grazed areas (Fig. 4E; combined probabilities test $\chi^2 = 19.3$, $P = 0.03$, not significant after correction for multiple tests) with large differences in detection frequency only on the Missouri River ($t = 3.2$, $P = 0.003$). Individual species in this guild were no more likely to be less or more

TABLE 3. EFFECTS OF LANDSCAPE VARIABLES ON TOTAL DETECTIONS, RICHNESS, AND DETECTIONS BY GUILD

Landscape variable		Statistic	All birds[c]	Richness[d]	Cowbirds[e]	Prime hosts[f]	Non-hosts[g]	Long-distance migrant[h]	Residents[i]	Short-distance migrant[j]
Local Human Habitation	Σ[a]	Dir	Neg	Pos	Pos	Pos	Neg	Neg	Neg	Pos
		χ^2	5.25	1.8	28.61	9.21	1.73	11.67	1.79	0.23
		P	0.874	0.985	0.001*	0.512	0.998	0.308	0.998	>0.99
	#[b]	Dir	Neg			Neg	Neg	Neg	Neg	Neg
		χ^2	39.8	N/A	N/A	18.4	18.4	25.6	7.12	12.9
		P	<0.001*	N/A	N/A	0.047	0.047	0.004*	0.525	0.231
Regional Human Habitation	Σ	Dir	Pos	Pos	Pos	Neg	Pos	Pos	Pos	Pos
		χ^2	6.36		25.71	10.73	16.18	5.44	9.30	14.26
		P	0.174	0.080[k]	0.001*	0.030	0.002*	0.245	0.054	0.026
	#	Dir	Neg			Neg	Neg	Neg	Neg	Neg
		χ^2	9.6	N/A	N/A	2.2	4.5	4.7	1.7	5.1
		P	0.144	N/A	N/A	0.903	0.605	0.585	0.944	0.531
Local Agriculture	Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos	Pos	Pos
		χ^2	22.94	15.12	34.98	7.67	31.59	7.52	10.90	38.08
		P	0.011*	0.019	<0.001*	0.661	0.001*	0.676	0.366	<0.001*
	#	Dir	Neg			Neg	Neg	Neg	Pos	Pos
		χ^2	0.5	N/A	N/A	0.79	0.4	9.8	3.1	0.3
		P	>0.99	N/A	N/A	0.999	>0.99	0.279	0.926	>0.99
Regional Agriculture	Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos	Pos	Pos
		χ^2	50.66	17.14	56.91	26.72	34.47	14.46	26.96	55.29
		P	<0.001*	0.029	<0.001*	0.003*	<0.001*	0.153	0.003*	<0.001*
	#	Dir	Pos			Pos	Pos	Neg	Pos	Pos
		χ^2	14.3	N/A	N/A	7.4	1.9	1.9	1.7	6.2
		P	0.159	N/A	N/A	0.690	0.997	0.997	0.998	0.794
Local Deciduous	Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos	Pos	Pos
		χ^2	38.01	15.70	31.33	56.87	14.07	16.71	29.28	34.42
		P	<0.001*	0.204	0.005*	<0.001*	0.445	0.272	0.010	0.002*
	#	Dir	Pos			Pos	Pos	Neg	Pos	Pos
		χ^2	15.7	N/A	N/A	4.51	10.47	2.3	15.08	0.29
		P	0.334	N/A	N/A	0.991	0.727	>0.99	0.237	>0.99
Regional Deciduous	Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos	Pos	Neg
		χ^2	12.89	15.12	20.89	20.34	24.25	17.17	28.30	0.62
		P	0.230	0.056	0.022	0.026	0.007*	0.071	0.002*	>0.99
	#	Dir	Pos			Pos	Pos	Pos	Pos	Pos
		χ^2	20.4	N/A	N/A	1.1	8.5	7.9	6.9	2.7
		P	0.026*	N/A	N/A	>0.99	0.576	0.635	0.735	0.987
Local Conifer	Σ	Dir	Neg	Neg	Neg	Neg	Neg	Neg	Neg	Neg
		χ^2	18.66	13.73	26.49	7.67	18.64	0.57	23.77	22.38

TABLE 3. CONTINUED

Landscape variable		Statistic	All birds[c]	Richness[d]	Cowbirds[e]	Prime hosts[f]	Non-hosts[g]	Long-distance migrant[h]	Residents[i]	Short-distance migrant[j]
		P	0.017	0.033	0.001*	0.466	0.045	>0.99	0.002*	0.004*
	#	Dir	Neg			Pos	Neg	Pos	Neg	Neg
		χ^2	5.1	N/A	N/A	3.0	5.9	12.1	6.2	4.0
		P	0.748	N/A	N/A	0.936	0.655	0.146	0.629	0.857
Regional Conifer	Σ	Dir	Neg	Pos	Neg	Neg	Neg	Pos	Neg	Neg
		χ^2	6.45	7.03	43.87	3.42	23.72	23.30	11.90	21.27
		P	0.597	0.318	<0.001*	0.905	0.003*	0.003*	0.156	0.006*
	#	Dir	Neg			Neg	Neg	Pos	Neg	Neg
		χ^2	8.5	N/A	N/A	1.4	8.4	14.2	3.0	10.4
		P	0.383	N/A	N/A	0.994	0.392	0.076	0.932	0.241

Note: Results from combined probabilities tests of linear regression of summed detections of all species in each guild (Σ) and from binomial tests on direction of change of each species in the guild (#).
* Significant ($P < 0.05$) after Bonferroni correction for multiple tests.
[a] Chi-square value and significance from multiple comparison tests based on regression of landscape value on total abundance within each guild.
[b] Chi-square value and significance from multiple comparison tests based on binomial tests examining the proportion of species more or less abundant in sites with high values for each landscape component.
[c] Average number of all detections per survey.
[d] Average number of species detected per year at a given survey location (3 surveys).
[e] Number of Brown-headed Cowbirds detected.
[f] Number of primary cowbird hosts detected (Appendix 2).
[g] Average number of non-hosts detected (Appendix 2).
[h] Average number of long-distance migrants detected per survey (Appendix 2).
[i] Average number of residents detected per survey (Appendix 2).
[j] Average number of short-distance migrants detected per survey (Appendix 2).
[k] Regression run only on the Bitterroot River; P-value is for regression (Appendix 4).

TABLE 4. INDIVIDUAL SPECIES RESPONSES TO LANDSCAPE COMPONENTS

Landscape component	N	P	Ratio	River system: Bitterroot	Sacramento	San Joaquin	Missouri	Snake	Sheldon	Hart Mountain
High Local Human Habitation										
More Abundant Species										
Bullock's Oriole	4	<0.001*	2.33	0.01/0.11	0.14/0.29	0.30/0.38			0.18/0	
Yellow Warbler	4	<0.001*	3.23	0.18/1.05	0.01/0.01	0.02/0.00			0.73/0.50	
Brown-headed Cowbird	4	<0.001*	1.83	0.40/0.84	0.38/0.59	1.01/1.27			0.82/1.50	
Red-winged Blackbird	4	<0.001*	1.54	0.01/0.20	0.00/0.00	1.22/1.17			1.45/1.25	
Black-headed Grosbeak	4	0.001*	1.62	0.10/0.11	0.42/0.76	0.00/0.11			0.18/0.50	
American Robin	4	0.009	1.79	0.26/0.48	0.10/0.17	0.03/0.05			0.18/0.75	
Western Wood-pewee	3	<0.001*	2.21	0.04/0.27	0.43/0.87				0.45/0.25	
Spotted Towhee	3	0.005*	1.67		0.79/1.30	0.50/0.69			0.09/0.00	
Song Sparrow	3	0.009*	1.99	0.04/0.20		0.60/0.84			0.64/0.50	
Willow Flycatcher	3	0.014	3.13	0.02/0.14	0.01/0.00				0.09/0.00	
Downy Woodpecker	3	0.030	1.90	0.03/0.05	0.06/0.14	0.01/0.00				
Red-shafted Flicker	3	0.034	1.68	0.07/0.23		0.05/0.03			0.27/0.00	
Cedar Waxwing	2	<0.001*	2.51	0.08/0.42	0.16/0.0					
Marsh Wren	2	0.042*	13.21			0.0/0.75			0.09/0.0	
Less Abundant Species										
MacGillivray's Warbler	3	<0.001*	4.92	0.52/0.08	0.00/0.00				0.27/0.25	
Townsend's Warbler	3	0.024	107.20	0.25/0.00	0.00/0.01				0.18/0.00	
Ash-throated Flycatcher	2	0.009	1.28		0.51/0.23	0.75/0.70				
Western Scrub-Jay	2	0.011*	2.03		0.26/0.17	0.59/0.17				
High Regional Human Habitation										
More Abundant Species										
European Starling	2	<0.001*	23.12	0.01/0.22				0.11/1.43		
Western Wood-pewee	2	<0.001*	4.70	0.08/0.36				0.02/0.31		
Bullock's Oriole	2	<0.001*	6.46	0.02/0.09				0.18/0.70		
House Wren	2	<0.001*	18.19	0.01/0.04				0.10/1.42		
Red-winged Blackbird	2	<0.001*	6.13	0.03/0.16				0.02/0.15		
Brown-headed Cowbird	2	<0.001*	1.57	0.50/0.77				0.17/0.58		
American Robin	2	0.002*	2.00	0.35/0.49				0.87/1.62		
Yellow Warbler	2	0.003*	2.14	0.39/0.96				2.32/2.39		
Willow Flycatcher	2	0.004*	1.77	0.05/0.11				0.01/0.01		
Downy Woodpecker	2	0.010*	2.30	0.04/0.08				0.03/0.12		
Tree Swallow	2	0.010*	2.99	0.02/0.06				0.12/0.18		
American Goldfinch	2	0.019	3.49	0.01/0.05				0.61/0.96		

TABLE 4. CONTINUED

Landscape component	N	P	Ratio	River system: Bitterroot	Sacramento	San Joaquin	Missouri	Snake	Sheldon	Hart Mountain
Less Abundant Species										
Yellow-rumped Warbler	2	<0.001*	4.92	0.10/0.02				0.21/0.02		
MacGillivray's Warbler	2	<0.001*	2.56	0.39/0.19				0.14/0.01		
Warbling Vireo	2	<0.001*	2.00	0.53/0.36				1.07/0.17		
Swainson's Thrush	2	<0.001*	2.39	0.21/0.11				0.04/0.00		
Dusky Flycatcher	2	<0.001*	5.64	0.44/0.08				0.09/0.04		
Ruffed Grouse	2	<0.001*	6.90	0.07/0.01				0.06/0.00		
Red-naped Sapsucker	2	0.006*	2.09	0.14/0.08				0.26/0.06		
Veery	2	0.006*	2.47	0.11/0.03				0.45/0.15		
Song Sparrow	2	0.012	1.34	0.09/0.14				1.01/0.21		
American Crow	2	0.021*	1.43	0.00/0.00				0.26/0.07		
Western Tanager	2	0.023*	3.06	0.07/0.02				0.08/0.03		
High Local Agriculture										
More Abundant Species										
American Robin	4	<0.001*	2.19	0.23/0.65	0.09/0.18	0.07/0.03		1.12/1.90		
Bullock's Oriole	4	<0.001*	2.76	0.00/0.10	0.21/0.37	0.31/0.24		0.29/0.88		
House Wren	4	<0.001*	3.52	0.00/0.05	0.17/0.23	0.79/1.18		0.17/1.53		
European Starling	4	<0.001*	8.28	0.00/0.22	0.11/0.04	0.20/0.08		0.09/1.99		
Brown-headed Cowbird	4	<0.001*	1.75	0.39/0.81	0.35/0.51	0.96/1.38		0.34/0.52		
Yellow Warbler	4	<0.001*	1.97	0.19/1.21	0.03/0.00	0.03/0.03		2.68/2.29		
Great Horned Owl	4	0.004*	N/A	0.00/0.01	0.00/0.02	0.00/0.08		0.00/0.03		
Tree Swallow	4	0.006*	1.33	0.00/0.08	0.68/0.43	0.48/0.55		0.11/0.21		
Black-headed Grosbeak	4	0.006*	1.72	0.10/0.16	0.36/0.80	0.02/0.08		0.25/0.14		
Spotted Sandpiper	4	0.010*	6.94	0.00/0.13	0.01/0.01	0.02/0.00		0.03/0.01		
Downy Woodpecker	4	0.014	2.24	0.04/0.08	0.03/0.12	0.01/0.00		0.04/0.13		
American Goldfinch	4	0.030	1.57	0.00/0.02	0.32/0.28	0.09/0.25		0.79/0.90		
Western Wood-pewee	3	<0.001*	5.26	0.03/0.40	0.29/0.69			0.09/0.42		
Red-winged Blackbird	3	<0.001*	1.26	0.00/0.20		1.18/0.66		0.07/0.05		
Red-shafted Flicker	3	<0.001*	2.02	0.05/0.26		0.04/0.02		0.28/0.15		
Song Sparrow	3	<0.001*	1.73	0.02/0.23		0.72/1.51		0.95/0.12		
Cedar Waxwing	3	0.004*	1.51	0.12/0.36	0.07/0.00			0.36/0.15		
Willow Flycatcher	3	0.014	3.48	0.03/0.13	0.01/0.02			0.00/0.01		
Spotted Towhee	3	0.033	1.98	0.00/0.00	0.02/1.22	0.63/0.96				
Black-billed Magpie	2	0.001*	1.77	0.26/0.40				0.22/0.54		
Eastern Kingbird	2	0.001*	76.67	0.00/0.02				0.00/0.04		
White-breasted Nuthatch	2	0.032*	3.29	0.01/0.03	0.04/0.09					
Less Abundant Species										
Warbling Vireo	3	<0.001*	2.16	0.61/0.41	0.07/0.03			0.74/0.16		
Dusky Flycatcher	3	<0.001*	7.05	0.39/0.06	0.01/0.00			0.06/0.03		

TABLE 4. CONTINUED

Landscape component	N	P	Ratio	River system						
				Bitterroot	Sacramento	San Joaquin	Missouri	Snake	Sheldon	Hart Mountain
Yellow-rumped Warbler	3	0.005*	2.26	0.09/0.06	0.01/0.00			0.16/0.02		
N. Rough-winged Swallow	3	0.017*	N/A		0.02/0.00	0.13/0.00		0.03/0.00		
Townsend's Warbler	3	0.019	46.12	0.29/0.00	0.00/0.01	0.00/0.01				
MacGillivray's Warbler	2	<0.001*	11.87	0.60/0.06				0.12/0.01		
Veery	2	0.001*	3.60	0.11/0.04				0.40/0.07		
Ruffed Grouse	2	0.001*	5.29	0.07/0.02				0.06/0.00		
Nuttall's Woodpecker	2	0.003*	1.87		0.60/0.41	0.31/0.08				
Chipping Sparrow	2	0.005*	9.25	0.10/0.01				0.04/0.01		
Violet-green Swallow	2	0.007*	9.30		0.01/0.00			0.32/0.03		
Ruby-crowned Kinglet	2	0.010*	25.32	0.16/0.01	0.01/0.00					
Red-naped Sapsucker	2	0.012*	1.77	0.12/0.09				0.22/0.05		
Western Scrub-Jay	2	0.017*	1.82		0.28/0.11	0.57/0.35				
Orange-crowned Warbler	2	0.017	7.42	0.13/0.02	0.02/0.00					
High Regional Agriculture										
More Abundant Species										
Brown-headed Cowbird	5	<0.001*	1.97	0.26/0.79	0.32/0.61	0.94/1.44	0.44/0.50	0.25/0.63		
Bullock's Oriole	5	<0.001*	1.59	0.00/0.08	0.19/0.31	0.44/0.27	0.39/0.59	0.20/0.74		
House Wren	5	<0.001*	1.12	0.00/0.07	0.24/0.58	0.99/0.95	2.44/2.45	0.18/1.16		
Yellow Warbler	5	<0.001*	1.35	0.07/0.97	0.02/0.00	0.05/0.03	3.03/3.73	2.10/2.67		
American Robin	5	<0.001*	1.13	0.33/0.65	0.08/0.23	0.03/0.07	2.14/1.36	0.81/1.86		
American Goldfinch	5	<0.001*	1.37	0.00/0.06	0.33/0.37	0.05/0.43	1.78/2.41	0.56/1.32		
European Starling	5	0.002*	2.45	0.00/0.25	0.13/0.01	0.17/0.26	0.44/0.45	0.18/1.26		
Tree Swallow	5	0.015*	1.26	0.00/0.06	0.87/0.48	0.42/0.73	0.03/0.09	0.13/0.30		
Western Wood-pewee	4	<0.001*	1.21	0.04/0.34	0.32/0.99		1.78/1.18	0.04/0.39		
Spotted Towhee	4	0.002*	1.55	0.01/0.00	0.62/1.40	0.51/0.85	0.97/1.41			
Common Yellowthroat	4	0.004*	1.04	0.00/0.03	0.05/0.08	0.03/0.24	1.14/1.36			
Red-winged Blackbird	4	0.004*	1.38	0.00/0.13	0.01/0.00	0.91/0.84		0.02/0.16		
Downy Woodpecker	4	0.037	0.92	0.05/0.09	0.04/0.16		0.33/0.09	0.04/0.10		
Black-billed Magpie	3	<0.001*	3.09	0.13/0.66			0.03/0.09	0.36/0.29		
Eastern Kingbird	3	<0.001*	2.81	0.00/0.02			0.11/0.36	0.00/0.03		
Black-capped Chickadee	3	0.003*	1.33	0.13/0.66			0.72/0.05	0.29/0.30		
Willow Flycatcher	3	0.012	7.40	0.01/0.11	0.01/0.00			0.00/0.01		
Less Abundant Species										
Swainson's Thrush	5	<0.001*	6.17	0.34/0.02	0.00/0.01	0.10/0.04	0.11/0.09	0.07/0.00		
Warbling Vireo	4	<0.001*	4.73	0.59/0.16	0.05/0.01		0.31/0.00	0.96/0.16		
MacGillivray's Warbler	3	<0.001*	8.68	0.52/0.06	0.01/0.00			0.09/0.00		
Violet-green Swallow	3	0.003*	17.63		0.01/0.00		0.14/0.00	0.84/0.05		
American Crow	3	0.004*	8.57	0.01/0.01		0.10/0.02		0.26/0.02		
Yellow-rumped Warbler	3	0.007*	2.98	0.16/0.07	0.01/0.00			0.17/0.03		

TABLE 4. CONTINUED

Landscape component	N	P	Ratio	River system: Bitterroot	Sacramento	San Joaquin	Missouri	Snake	Sheldon	Hart Mountain
Townsend's Warbler	2	<0.001*	166.26	0.47/0.00	0.00/0.01					
Western Kingbird	2	0.015*	2.22		0.38/0.24	1.69/0.69				
Western Scrub-Jay	2	0.030	1.77		0.27/0.18	0.63/0.33				
High Local Deciduous Riparian										
More Abundant Species										
Yellow Warbler	7	<0.001*	1.25	0.03/0.77	0.00/0.01	0.00/0.03	2.71/3.21	1.90/2.81	1.00/0.50	2.00/0.30
Black-headed Grosbeak	7	0.014*	1.80	0.05/0.11	0.72/0.44	0.02/0.17	0.21/0.57	0.08/0.23	0.00/0.33	0.09/0.70
Song Sparrow	6	<0.001*	1.85	0.01/0.21		0.53/1.33	0.50/0.79	0.35/0.48	1.00/0.50	0.18/0.00
Western Wood-pewee	6	0.015	1.60	0.03/0.21	0.62/0.54		0.71/2.29	0.27/0.24	0.67/0.17	0.27/0.40
Cedar Waxwing	4	<0.001*	1.58	0.06/0.25	0.27/0.02		0.36/0.79	0.24/0.34		
Orange-crowned Warbler	4	0.034	2.81	0.12/0.04	0.00/0.02				0.33/0.17	0.00/1.10
Black-capped Chickadee	3	<0.001*	3.48	0.10/0.51			0.00/0.43	0.27/0.36		
Red-eyed Vireo	3	<0.001*	19.60	0.00/0.04			0.07/1.29	0.00/0.01		
Red-naped Sapsucker	3	<0.001*	4.23	0.06/0.21				0.07/0.22		0.09/0.60
Gray Catbird	3	0.007*	3.72	0.00/0.04			0.21/0.57	0.05/0.19		
Veery	2	<0.001*	16.23	0.00/0.08				0.03/0.38		
Fox Sparrow	2	<0.001*	N/A					0.00/0.12		0.00/0.90
Least Flycatcher	2	0.006*	3.68	0.00/0.03			0.71/2.50			
American Redstart	2	0.011*	14.27	0.02/0.19			0.00/0.43			
Bewick's Wren	2	0.031*	1.57		0.49/0.72	0.45/0.75				
Less Abundant Species										
MacGillivray's Warbler	4	0.001*	3.90	0.58/0.13	0.00/0.00			0.04/0.04	0.33/0.17	0.18/0.00
Townsend's Warbler	4	0.004*	12.66	0.40/0.00		0.00/0.01			0.00/0.17	0.18/0.00
Western Kingbird	3	0.026	1.87		0.39/0.14	0.92/0.56			0.00/0.17	
High Regional Deciduous Riparian										
More Abundant Species										
Western Wood-pewee	5	<0.001*	2.23	0.01/0.33	0.51/0.92		1.07/2.00	0.23/0.05		0.20/0.70
American Robin	5	0.020	1.19	0.31/0.65	0.05/0.25		1.36/1.93	1.66/0.94		1.60/1.90
Song Sparrow	4	<0.001*	2.30	0.01/0.20			0.00/0.21	0.45/1.03		0.30/0.00
Yellow Warbler	4	<0.001*	1.27	0.07/0.90			2.43/3.43	2.53/2.13		1.40/0.50
Red-shafted Flicker	4	0.011*	1.47	0.08/0.23			0.79/1.36	0.20/0.32		0.90/0.80
Cedar Waxwing	4	0.012	2.25	0.04/0.23	0.01/0.00		0.29/0.64	0.27/0.32		
Black-capped Chickadee	3	<0.001*	3.14	0.17/0.69			0.29/1.29	0.26/0.26		
Red-eyed Vireo	3	0.004*	67.64	0.00/0.03			0.00/0.71	0.01/0.02		
Willow Flycatcher	3	0.009*	9.86	0.00/0.10	0.02/0.00			0.01/0.00		
Red-naped Sapsucker	3	0.012*	1.87	0.03/0.09				0.12/0.20		0.40/0.70
Red-winged Blackbird	3	0.046	0.45	0.00/0.13				0.15/0.05		0.80/0.00
White-breasted Nuthatch	3	0.046	3.32	0.00/0.03	0.07/0.10		0.00/0.14			

TABLE 4. CONTINUED

Landscape component	N	P	Ratio	River system: Bitterroot	Sacramento	San Joaquin	Missouri	Snake	Sheldon	Hart Mountain
Black-billed Magpie	2	<0.001*	2.45	0.17/0.69				0.39/0.34		
Less Abundant Species										
Townsend's Warbler	2	<0.001*	159.60	0.45/0.00	0.00/0.01					
Orange-crowned Warbler	2	0.006*	2.33	0.11/0.04						1.10/0.50
MacGillivray's Warbler	2	0.019	4.52	0.54/0.09				0.05/0.08		
Ruby-crowned Kinglet	2	0.024	4.50	0.18/0.01						0.00/0.10
High Local Conifer Forest										
More Abundant Species										
Swainson's Thrush	4	<0.001*	4.92	0.01/0.31			0.00/0.14	0.00/0.07		0.19/0.20
Warbling Vireo	4	<0.001*	1.96	0.21/0.57			0.00/0.14	0.17/0.71		1.25/1.90
MacGillivray's Warbler	3	<0.001*	13.61	0.05/0.50				0.01/0.11		0.00/0.10
Yellow-rumped Warbler	3	0.007*	2.86	0.03/0.09				0.02/0.12		0.06/0.10
Dusky Flycatcher	3	0.023	0.97	0.02/0.29				0.02/0.03		1.81/1.30
Western Tanager	3	0.034	3.97	0.01/0.08				0.02/0.04		0.06/0.20
Ruffed Grouse	2	<0.001*	11.67	0.01/0.09				0.00/0.06		
Ruby-crowned Kinglet	2	0.007*	6.22	0.01/0.19						0.06/0.10
Veery	2	0.028	2.14	0.02/0.00				0.17/0.59		
Violet-green Swallow	2	0.029	5.30				0.07/0.00	0.05/0.43		
Less Abundant Species										
Western Wood-pewee	4	<0.001*	1.64	0.40/0.03			1.79/1.07	0.33/0.06		0.50/1.00
American Robin	4	<0.001*	1.88	0.72/0.25			2.07/0.93	1.57/0.95		1.94/1.70
House Wren	4	<0.001*	1.60	0.07/0.00			2.50/1.93	1.24/0.05		4.50/4.50
Bullock's Oriole	4	<0.001*	1.90	0.14/0.00			0.64/0.36	0.69/0.25		0.50/0.80
European Starling	4	<0.001*	2.44	0.25/0.00			0.71/0.14	1.27/0.11		0.50/1.40
Yellow Warbler	4	<0.001*	1.41	1.30/0.07			3.21/3.29	2.49/2.80		0.75/1.20
Red-shafted Flicker	4	<0.001*	1.53	0.30/0.06			1.43/0.64	0.17/0.25		1.06/1.10
Downy Woodpecker	4	0.003*	2.11	0.11/0.04			0.00/0.14	0.13/0.03		0.38/0.20
Mourning Dove	4	0.003*	2.01	0.02/0.00			2.21/1.29	0.55/0.14		0.06/0.10
Brown-headed Cowbird	4	0.004*	1.61	0.84/0.36			0.43/0.64	0.54/0.42		1.00/0.70
Cedar Waxwing	3	<0.001*	1.32	0.33/0.08			0.86/0.79	0.19/0.38		
Black-billed Magpie	3	0.009*	1.76	0.44/0.23			0.00/0.14	0.47/0.22		
Black-capped Chickadee	3	0.013	1.74	0.44/0.23			0.64/0.07	0.29/0.32		
American Goldfinch	3	0.024	1.43	0.05/0.00			2.07/1.93	1.08/0.81		
Red-winged Blackbird	2	<0.001*	5.64	0.24/0.00				0.09/0.09		
Willow Flycatcher	2	0.001*	5.14	0.16/0.02				0.01/0.01		
Least Flycatcher	2	0.001*	2.30	0.03/0.00				2.50/1.14		
Spotted Sandpiper	2	0.007*	6.49	0.12/0.00				0.02/0.04		
Great Blue Heron	2	0.017*	8.67	0.04/0.00				0.02/0.01		

TABLE 4. CONTINUED

Landscape component	N	P	Ratio	River system: Bitterroot	Sacramento	San Joaquin	Missouri	Snake	Sheldon	Hart Mountain
High Regional Conifer Forest										
More Abundant Species										
Swainson's Thrush	4	<0.001*	1.89	0.01/0.34			0.00/0.14	0.00/0.08		0.42/0.10
Warbling Vireo	4	<0.001*	1.81	0.25/0.65			0.14/0.64	0.14/0.85		1.11/1.30
MacGillivray's Warbler	3	<0.001*	7.25	0.04/0.60				0.00/0.15		0.11/0.00
Dusky Flycatcher	3	<0.001*	1.41	0.02/0.31				0.02/0.06		1.37/2.10
Western Tanager	3	0.002*	1.28	0.01/0.10				0.00/0.04		0.16/0.00
Chipping Sparrow	3	0.003*	6.77	0.01/0.12			0.07/0.00	0.00/0.03		
Pine Siskin	3	0.003*	4.31	0.06/0.35				0.02/0.04		0.05/0.00
Yellow-rumped Warbler	3	0.009*	2.72	0.03/0.17				0.02/0.17		0.11/0.00
Townsend's Warbler	2	<0.001*	7.96	0.00/0.43						0.11/0.00
Orange-crowned Warbler	2	<0.001*	1.73	0.00/0.10						0.53/1.10
Ruffed Grouse	2	0.002*	9.95	0.01/0.09				0.00/0.03		
Violet-green Swallow	2	0.003*	12.60				0.00/0.07	0.04/0.51		
Mountain Chickadee	2	0.008*	N/A	0.00/0.05						0.00/0.30
Ruby-crowned Kinglet	2	0.025	3.03	0.01/0.18						0.11/0.00
Fox Sparrow	2	0.032	2.25					0.05/0.15		0.32/0.80
Less Abundant Species										
Western Wood-pewee	4	<0.001*	1.92	0.47/0.04			1.07/1.57	0.39/0.05		0.58/0.40
Yellow Warbler	4	<0.001*	1.97	1.34/0.05			2.43/3.21	2.75/2.26		1.68/0.30
Bullock's Oriole	4	<0.001*	3.16	0.14/0.00			0.21/0.43	0.75/0.20		0.79/0.30
European Starling	4	<0.001*	11.34	0.25/0.00			1.14/0.00	1.33/0.20		0.63/0.10
Brown-headed Cowbird	4	<0.001*	2.01	0.84/0.23			0.36/0.71	0.63/0.23		1.05/0.90
American Robin	4	<0.001*	1.39	0.73/0.26			1.36/2.79	1.81/1.00		1.95/1.80
House Wren	4	<0.001*	1.37	0.07/0.00			2.21/3.00	1.18/0.11		3.32/3.80
Downy Woodpecker	4	0.012*	2.55	0.10/0.06			0.36/0.00	0.10/0.04		0.32/0.20
Cedar Waxwing	3	<0.001*	1.26	0.31/0.01			0.29/1.00	0.25/0.32		
Red-winged Blackbird	3	<0.001*	38.45	0.24/0.00				0.13/0.03		0.42/0.00
American Goldfinch	3	<0.001*	1.67	0.05/0.00			1.29/1.71	1.25/0.40		
Eastern Kingbird	3	0.003*	3.57	0.02/0.00			0.36/0.14	0.05/0.00		
Willow Flycatcher	2	<0.001*	11.96	0.16/0.01				0.01/0.00		
Spotted Sandpiper	2	0.001*	27.55	0.12/0.00				0.03/0.01		

Notes: Includes all species with study-wide differences in average abundance between the lower 25% of plots (*Low*) and the upper 25% of plots (*High*) when all plots within each river system are ranked from lowest to highest for each landscape variable. The N is the number of rivers in which the species and landscape component were present. P-values are from Fisher's combined probability tests across rivers. We report the ratio of detection frequency (detections per survey) in all of the less abundant class (*Low* or *High*) to detection frequency in all of the more abundant class as 1:x, where x = Ratio. In addition, detection frequency in each river system for *Low* and *High* plots (*Low*/*High*) is indicated.

* Significant after Bonferroni correction for multiple tests.

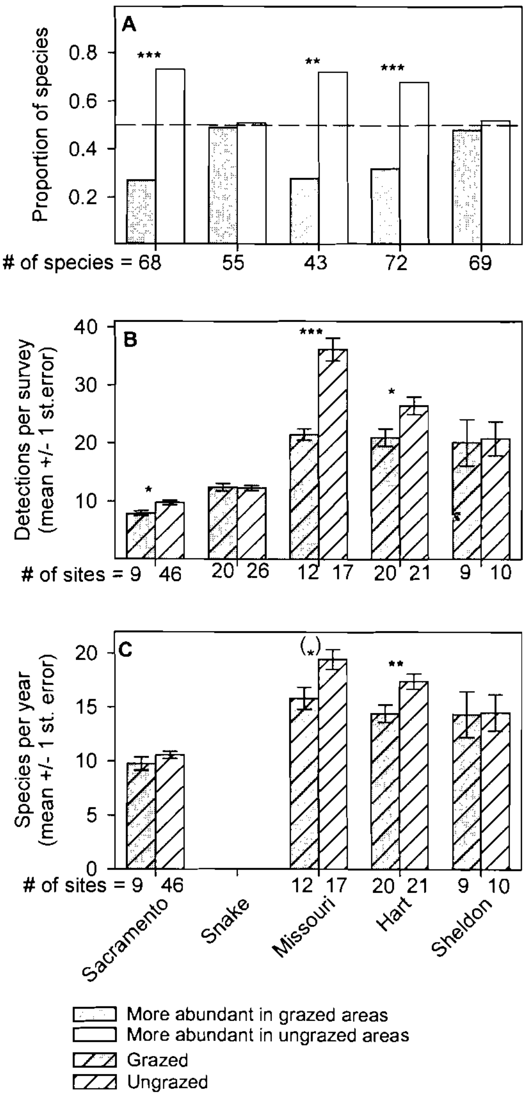

FIGURE 2. Total response of all species to grazing in each riparian system. Proportion of all species more abundant in grazed or ungrazed plots (A), average number of birds detecied per survey (B), and the average number of species detected over the course of a single year at a given location (C) for grazed and ungrazed plots in each river system. * P < 0.05, ** P < 0.01, *** P < 0.005. (*) = P-value not significant after correction for multiple tests.

abundant in grazed sites (Fig. 4F; combined probabilities test χ^2 = 7.5, P = 0.679).

Total abundance of open cup nesters was significantly higher in ungrazed survey locations (Fig. 5A; combined probabilities test χ^2 = 46.4, P < 0.0005) and an average of 65% (± 8%) of open-cup nesting species were less abundant in grazed areas (Fig. 5B; combined probabilities test χ^2 = 35.3 P < 0.001). Primary cavity nesting species trended in the same direction (Fig. 5C; combined probabilities test χ^2 = 20.4, P = 0.026, not significant after correction for multiple tests), and secondary cavity nesters showed conflicting patterns on different riparian systems with no overall effect (Fig. 5E; combined probabilities test χ^2 = 4.4, P = 0.92). Binomial tests suggested no overall trend for cavity nesters (Fig. 5D and 5F), though the number of species in each guild was too small for rigorous analysis. On the Missouri, total abundances of open cup and primary cavity nesters were significantly greater on ungrazed sites (t's > 4.2, P's < 0.001) and 22 of 25 open-cup nesting species were more abundant in ungrazed sites. Open-cup nesting abundance was also lower on the Hart Mountain (total abundance: t = 2.6, P = 0.013) and Sacramento River (t = 2.1, P = 0.04) systems, with 30 of 40 species less abundant in grazed areas on Hart Mountain (binomial test P = 0.003) and 27 of 40 species less abundant in grazed locations on the Sacramento (binomial test P = 0.04).

The overall abundance of all species nesting below 2.5 m was significantly lower in grazed sites compared to ungrazed sites (Fig. 6A; combined probabilities test χ^2 = 26.4, P = 0.003) and 67% of species in this category (± 5%) were less abundant in grazed sites (combined probabilities test χ^2 = 17, P = 0.07), with all rivers showing the same trend (Fig. 6B). In contrast, the combined abundance of all species with average nesting heights higher than 5 m showed only a non-significant trend to be lower in grazed areas (Fig. 6C; combined probabilities test χ^2 = 18.6, P = 0.045, not significant after correction for multiple tests), and only 58% (± 9%) of species in this guild were less abundant in grazed sites, with the Snake and Sheldon systems showing either opposite trends or no effect (Fig. 6D; combined probabilities test χ^2 = 5.8, P = 0.23).

DISCUSSION

This synthesis includes seven different western riparian systems, each embedded in a different landscape. In each system, data were gathered by different investigators using similar but not identical methodologies. Despite these differences, our results demonstrate that both landscape character and livestock grazing have some consistent, potentially West-wide effects on bird communities. Although some of these effects are similar to those found in the Midwest (landscape effects on Brown-headed Cowbirds, for example), others will require further study to determine the mechanisms responsible for the patterns (the effects of grazing and agriculture

TABLE 5. SPECIES SHOWING OVERALL TREND IN RESPONSE TO GRAZING

Less common in grazed areas			More common in grazed areas		
Species	Rivers	P	Species	Rivers	P
American Robin	5	0.005*	Dusky Flycatcher	4	0.040
Western Wood-pewee	5	0.031	Western Meadowlark	3	0.056
Black-headed Grosbeak	5	0.080	Brewer's Sparrow	2	0.110
Song Sparrow	4	0.020			
Hairy Woodpecker	4	0.031*			
Mallard	4	0.055			
Red-shafted Flicker	4	0.115			
MacGillivray's Warbler	4	0.129			
Cedar Waxwing	3	0.073			
Cordilleran Flycatcher	2	0.003*			
Red-eyed Vireo	2	0.008*			
Fox Sparrow	2	0.014*			
Green-tailed Towhee	2	0.015*			
Black-capped Chickadee	2	0.017			
Gray Catbird	2	0.032			
Ovenbird	2	0.177			
Turkey Vulture	2	0.197			

Note: Species are ranked by the number of riparian systems included in the analysis (minimum of two) and significance ($P < 0.2$). * Denotes significant after Bonferroni correction for multiple tests.

on Yellow Warblers, for example). Below, we summarize effects of different landscape components and provide a brief synthesis of our findings.

SCALE AND RESOLUTION

Until recently, there has been a significant gap between theoretical work stressing the scale-dependent nature of landscape effects (Wiens 1989, 1995; Dunning et al. 1992) and empirical studies that confine analysis to a single landscape scale (Donovan et al. 1995b, Robinson et al. 1995a, Thompson et al. 2000, Hejl and Young 1999; but see Tewksbury et al. 1998, Young and Hutto 1999, Donovan et al. 2000). The abundance and composition of bird communities are affected by multiple processes across different landscape scales (Dunning et al. 1992, Freemark et al. 1995); even a single process, such as nest predation, acts across multiple scales dependent on the range size and habitat affinities of the primary predators (Andrén 1995, Tewksbury et al. 1998). This variation in the scaling of processes suggests that conservation planning will be best served by examination of multiple scales. Multiple-scale landscape analyses allows the discovery of relationships that are relatively scale-insensitive, and thus more easily applied in management contexts, and it allows determination of appropriate scales when processes such as brood parasitism or nest predation are considered.

Our results show that different landscape components influence bird abundance and diversity at different scales. Overall, 40% of species significantly affected by landscape factors at one scale were not affected by these factors at the other scale (Table 4), suggesting that examination of landscapes at only a single spatial scale may result in loss of considerable information. Importantly, our examination of two landscape scales does not allow us to determine the point when considering more land area decreases rather than increases the explanatory power of a certain landscape variable, as we can only say that a larger landscape is better than a smaller one, or the other way around. Analyses comparing the effect sizes of landscape components at multiple scales would allow estimation of the relative importance of landscape features at different distances from an area of interest.

The appropriate scale is also a function of mapping resolution. Linear landscape components and components that typically have small patch sizes are usually underestimated when mapping resolution is coarse. It is not particularly surprising that we found no significant correlations between data gathered using the low resolution California GAP data and the detailed CWIS data (Table 2), as the resolution of the California GAP data (100 ha minimum patch size) is greater than the entire area of our local landscapes (78 ha). This coarse resolution is inappropriate for local scale habitat mapping, but it may still be appropriate for larger landscape scales as long as the biases are recognized. At our regional scale, where we used these data, we mapped 8000 ha around each survey location, which allowed for a mosaic of patches even

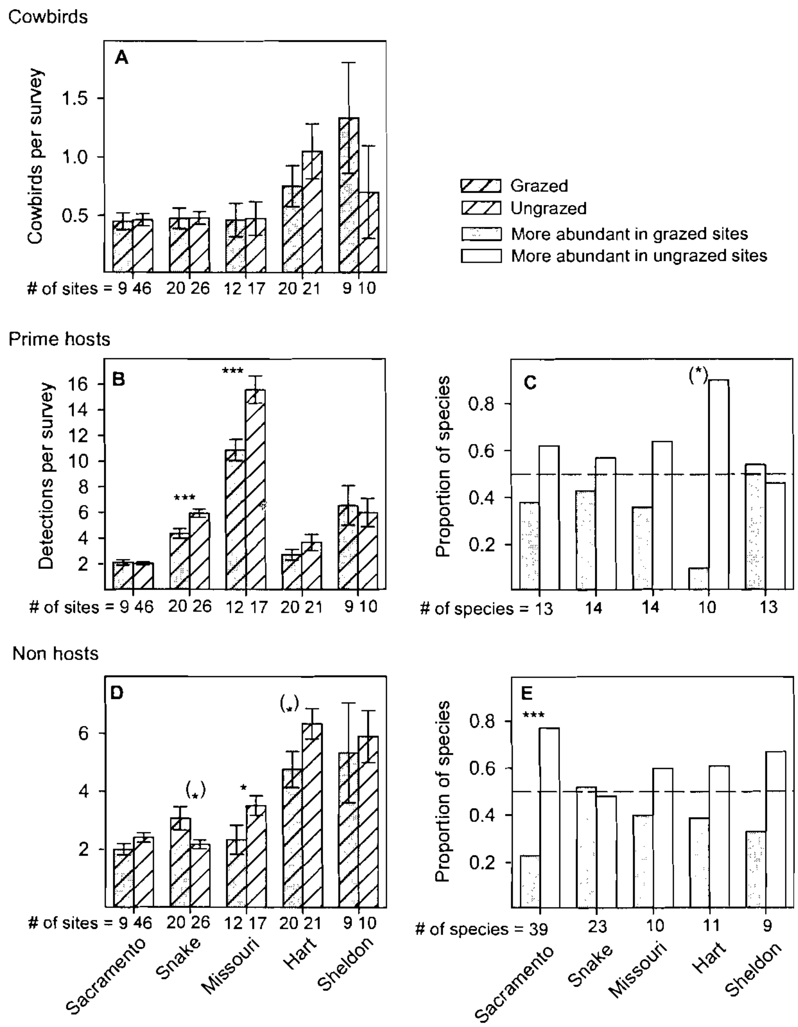

FIGURE 3. Grazing effects on cowbirds, prime hosts, and non-hosts. Total detections per survey on grazed and ungrazed sites (A, B, and D), and proportion of species in each guild more abundant in grazed or ungrazed sites (C and E), for cowbirds (A), prime hosts (B and C), and non-hosts (D and E) in each river system. * P < 0.05, ** P < 0.01, *** P < 0.005. (*) = P-value not significant after correction for multiple tests.

when these patches were 100 ha and larger. At this level, large differences in the regional landscape are fully apparent, but features such as dispersed housing or small riparian areas are not detected. Thus the effect of changing regional agriculture or coniferous forest cover is well represented in the coarse-grained data, while changes in linear deciduous riparian areas may go undetected. As landscape data of higher resolution become more broadly available, comparisons across regions should be possible using the same data sources for all landscape sizes, eliminating the confounding issues of shifting mapping resolution and allowing explicit comparison of scale.

Human Habitation and Agriculture

Our finding that overall avian abundance was positively related to regional agricultural abundance runs counter to findings from the East (Croonquist and Brooks 1991, 1993), but is not without precedent in the western United States (Carothers et al. 1974). These results may be better understood by examining the individual species with large differences in abundance, rather than by focusing on guilds (Mannan and Meslow 1984). The high congruence in the species increasing due to agriculture and human habitation is partly a function of the positive correlation that typically exists between agriculture

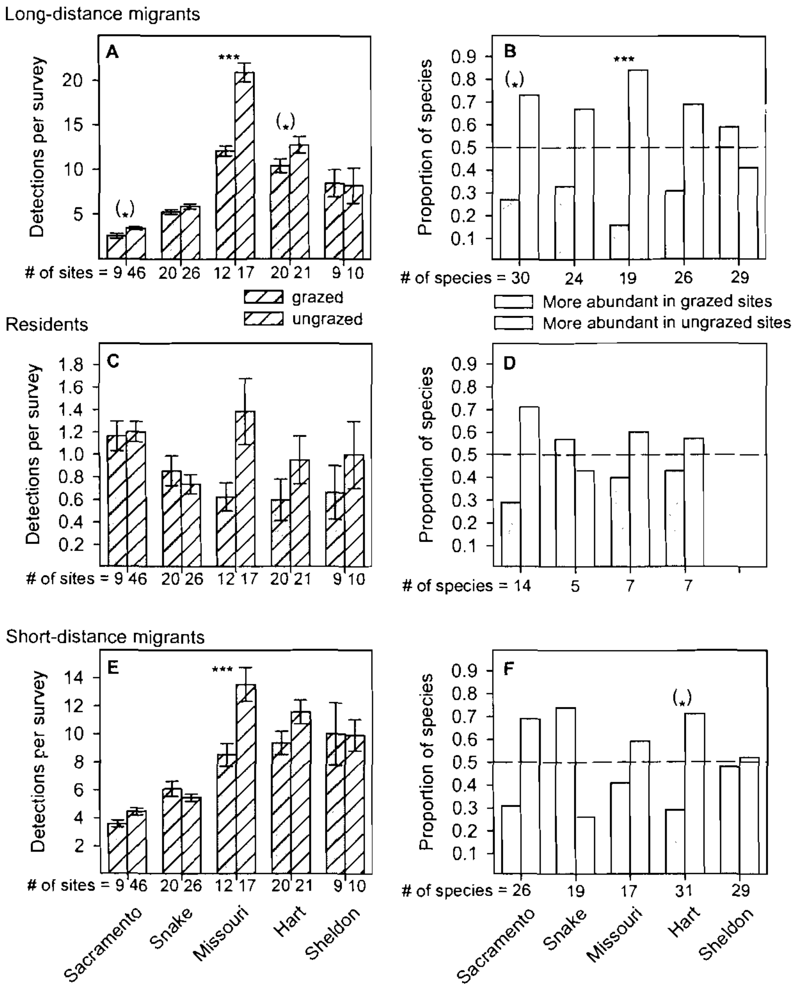

FIGURE 4. Grazing effects on long-distance migrants, residents, and short-distance migrants. Total detections per survey on grazed and ungrazed sites (A, C, and E), and proportion of species within each guild more abundant in grazed or ungrazed sites (B, D, and F), for long-distance migrants (A and B), year-round residents (C and D), and short-distance migrants (E and F) in each river system. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.005$. (*) = P-value not significant after correction for multiple tests.

and houses (Appendix 3). It is likely, however, that many species with higher relative abundance in areas with more agriculture also show similar numerical responses to high human habitation. Brown-headed Cowbirds use both agricultural and farm areas for foraging (Thompson 1994), and European Starlings often forage in suburban and agricultural areas (Fischl and Caccamise 1985). Indeed, most of the species that are more abundant in areas with high agriculture or human habitation often utilize multiple habitats; American Robins, Black-billed Magpies, starlings, and cowbirds are all examples. Increases in starlings may have consequences for other secondary cavity nesters, as starlings can exclude less aggressive species from cavities (Ingold 1989, 1994, 1998; Nilsson 1984, Kerpez and Smith 1990, Rich et al. 1994, Dobkin et al. 1995). Indeed, densities of Violet-green Swallows were significantly lower in sites with high agriculture at either scale—the same sites in which starlings were significantly more abundant (Table 4).

Higher Brown-headed Cowbird detection frequency in areas with more agriculture has been found previously across both local and regional scales (Conine et al. 1979, Donovan 1997, Tewksbury et al. 1999, Hejl and Young 1999, Hochachka et al. 1999, Young and Hutto 1999). Our finding that the detection frequency of pri-

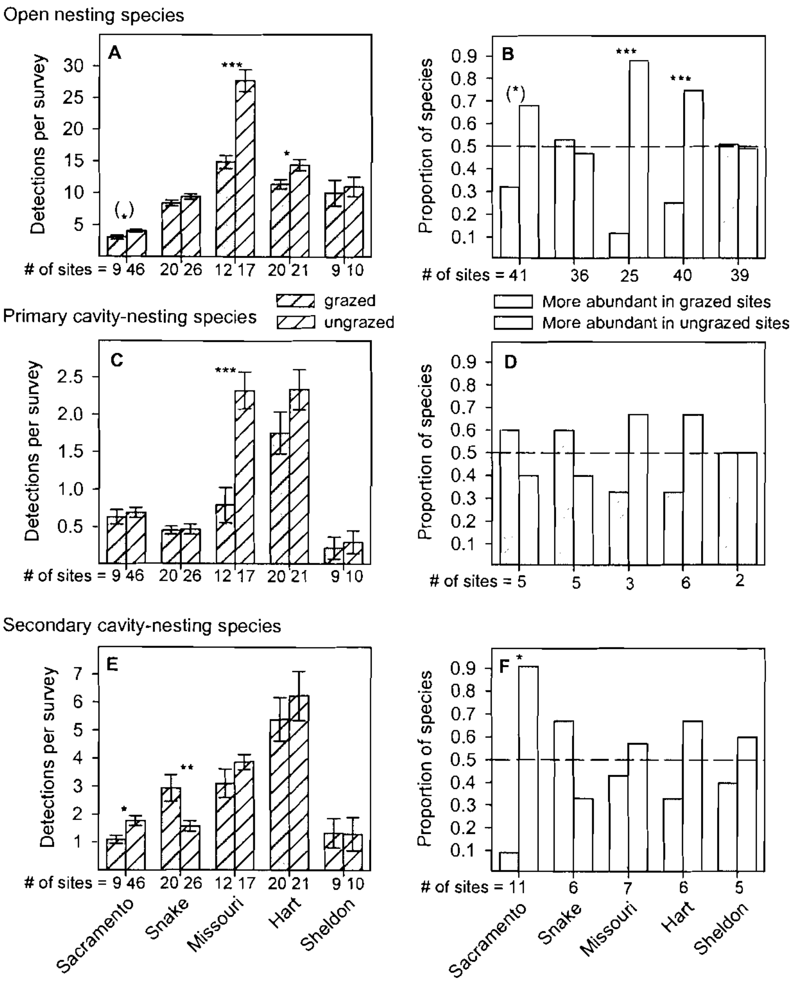

FIGURE 5. Grazing effects on open nesting species, primary cavity nesters, and secondary cavity nesting species. Total detections per survey on grazed and ungrazed sites (A, C, and E), and proportion of species in each guild more abundant in grazed or ungrazed sites (B, D, and F), for open-cup nesting species (A and B), primary cavity nesting species (C and D), and secondary cavity nesting species (E and F) in each river system. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.005$. (*) = P-value not significant after correction for multiple tests.

mary hosts was not lower in areas where cowbirds were common is consistent with other comparisons of cowbird density and host density (Donovan et al. 1997, Tewksbury et al. 1999, Young and Hutto 1999), and does not indicate that cowbirds have no effect on host communities (De Groot et al. 1999). The demographic effect of brood parasitism varies greatly among different host species (Lorenzana and Sealy 1999), and we first expect lower abundances of species that are particularly susceptible to parasitism. Indeed, the Dusky Flycatcher, Swainson's Thrush, Veery, Warbling Vireo, Orange-crowned Warbler, MacGillivray's Warbler, and American Redstart all suffer complete or nearly complete brood loss when parasitized (J. J. Tewksbury, unpubl. data) and are all less abundant in areas with high human habitation or high agriculture (Table 4), areas where cowbirds are abundant. In contrast, Yellow Warblers are more resistant to the demographic effect of brood parasitism (Clark and Robertson 1981, Sealy 1995), and they were more abundant in areas with high human habitation and agriculture. Importantly, human habitation and agriculture are often concentrated near productive riparian habitat with large flood-plains, areas where many long-distance migrants susceptible to parasitism are more abundant. Thus the trend for Yellow Warblers (more abundant in these areas) may characterize the natural response of other species, as they respond to larger riparian areas, but the ef-

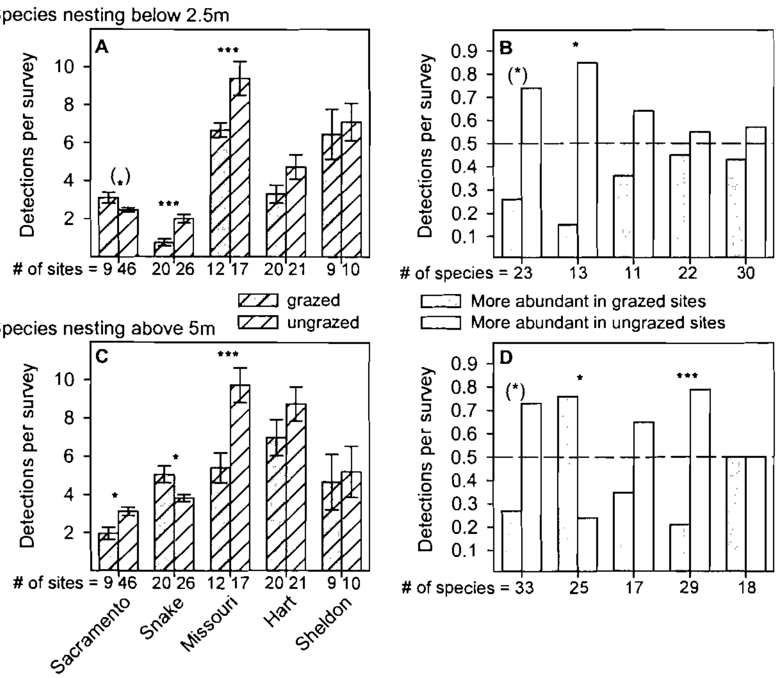

FIGURE 6. Effects of grazing on low and high nesting species. Total detections per survey on grazed and ungrazed sites (A and C), and proportion of species in each guild more abundant in grazed or ungrazed sites (B and D), for species nesting below 2.5m (A and B), and species nesting above 5 m (C and D) in each river system. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.005$. (*) = P-value not significant after correction for multiple tests.

fect of cowbirds may counter this trend. In addition, negative correlations between cowbird and host detection frequencies suggest that rates of brood parasitism are positively related to cowbird detection frequencies. While this assumption is reasonable across most levels of cowbird detection, where cowbird numbers are high, further increases may not change parasitism rates. This may be the case along the Sacramento River, where high-levels of parasitism at all sites may have already caused large regional declines in many species (Gaines 1974), so that current variation in cowbird detection frequency is uncorrelated with parasitism rates.

The largest limitations in understanding the effects of changing landscapes on riparian bird communities are the correlations among components of the landscape. In our study, we cannot separate unambiguously the effects of agriculture and human habitation because of the high correlation between these components (Appendix 3). In some cases, however, correlations between landscape components differ significantly among riparian systems, allowing insights into which relationships are causative, and which are simply due to covariation in landscape components. For example, local deciduous habitat is correlated strongly with higher host abundance (Table 3). In the Bitterroot Valley, agriculture is correlated positively with the amount of local deciduous habitat (Appendix 3; $r = 0.47$, $P < 0.001$), and, as a result, we see positive associations between host abundance and regional agriculture. Conversely, along the Snake River, local agriculture and deciduous habitats are negatively correlated (Appendix 3; $r = -0.55$, $P < 0.001$), and we see a strong negative relationship between host abundance and agriculture at the local scale (Appendix 4). Thus, changes in host abundance are likely caused by differences in the amount of deciduous habitat, not the amount of agriculture, but the effects are difficult to separate where these components are positively correlated.

Deciduous Riparian Area

Deciduous riparian area at the local scale is a function of the width of the riparian corridor; thus the positive correlations between avian abundance and deciduous habitat likely are consequences of greater habitat availability and heterogeneity associated with larger riparian corridors (Tyser 1983, Brown and Dinsmore 1986, Dobkin and Wilcox 1986, Craig and Beal 1992, Keller et al. 1993). All of the species that were significantly more abundant at survey locations with high local deciduous habitat are species traditionally considered riparian associates. The guild-level examination of the effects of increasing local deciduous area and increasing regional agriculture suggested similar effects (Table 3), but the individual species responding to these landscape components were quite different (Table 4). Fifteen species had significantly higher abundance in larger deciduous areas, and 17 species were higher in abundance in areas with more regional agriculture. However, only three species were more abundant under both these conditions (Table 4). Thus the bird communities in areas with high agricultural abundance share little in common with the communities in areas with large amounts of deciduous habitat, and guild-based analysis may lead to erroneous conclusions unless the responses of individual species are examined.

Within riparian systems, the breeding bird community found in smaller deciduous tracts was most often a subset of the birds found in larger tracts. Only three species were less abundant in sites with more local deciduous forest, and one of these, the Townsend's Warbler, is typically associated with coniferous habitats. Thus, preserving and restoring large tracts of deciduous habitat likely will do more to preserve riparian-associated species than will any other action. In addition, large deciduous patches also may reduce parasitism in parts of the patch as distance from the nearest cowbird feeding area increases.

Coniferous Forest

Studies in the Midwest have found that areas with higher conifer abundance, at scales similar to our regional scale, have lower cowbird abundance and parasitism (Donovan et al. 1995b, 1997; Robinson et al. 1995a). Recent work in the western United States, however, has suggested that the abundance of human habitation (Tewksbury et al. 1998), agriculture (Hejl and Young 1999, Young and Hutto 1999, Donovan et al. 2000), and the abundance of suitable hosts (Barber and Martin 1997; Tewksbury et al. 1998, 1999) are better predictors of parasitism pressure than is conifer abundance. Our study supports both bodies of work—cowbirds were not significantly less abundant in areas with more local coniferous forest, but they were related positively to both human habitation and agriculture, and they were also higher in larger riparian areas, where host abundance is also higher. At a regional scale, cowbirds did show a strong negative correlation with the amount of coniferous forest on the landscape, similar to results from the Midwest and East. This relationship at the regional scale is most likely due to the strong negative correlation between regional coniferous forest cover and agriculture on the Snake and the Bitterroot rivers (Appendix 3), the two rivers where cowbirds and coniferous forest are negatively related (Appendix 4). In the Bitterroot River system, rates of brood parasitism have been directly related to the amount of human habitation on the landscape, not the amount of coniferous forest (Tewksbury et al. 1998). The effects of coniferous forest on individual species were very similar across scales, with over 70% of species affected showing significant effects at both scales.

Grazing

Variation in the intensity, duration, and timing of grazing has been shown to influence bird communities (Saab et al. 1995), and its effects are particularly apparent in deciduous systems (Fleischner 1994). Our study includes a diversity of grazing regimes, and the effects on bird communities generally match the intensity and duration of the grazing. In the Missouri River, grazed sites have had cattle on them for over 50 years, and ungrazed sites have been free of grazing for over 25 years. This is reflected in the severe effects of grazing on the bird communities. In contrast, grazing-related differences were few in the Sheldon system, where long-term livestock grazing has left a highly degraded set of riparian habitats. Ungrazed survey locations were only in their third year of rest, and the general lack of differences in avian community composition reflected the very limited recovery made by the riparian plant community (D. Dobkin et al., unpubl. data).

Our finding that grazing had no effect on detection frequencies of Brown-headed Cowbirds in any riparian system runs counter to most previous studies (Page et al. 1978, Mosconi and Hutto 1982, Knopf et al. 1988, Schulz and Leninger 1991; but see Taylor 1986). However, we measured grazing pressure on individual study sites, not on the landscape as a whole; thus cowbirds may be foraging in grazed sites but searching for nests in ungrazed sites, where hosts are generally more abundant. Thus grazed and ungrazed sites may offer different resources for cowbirds; previous research in the Bitterroot River system has shown that cowbird abundance is strongly related to host abundance, as well as distance from agriculture (Tewksbury et al. 1999), supporting this possibility.

As cowbirds are not consistently more abundant in grazed areas, the much lower primary host abundance in grazed areas may not be simply the result of higher parasitism pressure, but instead may be due to interactions between vegetation differences and predation rates (Knopf

1985), lack of appropriate settling cues in grazed sites, or indirect interactions between food availability, foraging behavior, and nest predation (Martin 1992). Many primary hosts are also long-distance migrants, and we found that this group was lower in abundance in grazed areas as well. Saab et al. (1995) found the same result after reviewing the literature, and suggested that this could be due to the high proportion of open-cup nesters among long-distance migrants and greater sensitivity of open-cup nesters to grazing. Our data are consistent with this interpretation: open-cup nesters were more heavily affected by grazing than were primary or secondary cavity nesters. Open-cup nesters accounted for 96% of species and 81% of detections for long distance migrants, 82% of species and 28% of detections for short-distance migrants, and only 58% of species and 37% of all detections for residents.

Along the Missouri River, differences in primary cavity nesters between grazed and ungrazed areas were as great as differences in open-cup nesters, a finding that contrasts sharply with previous work (Good and Dambush 1943, Mosconi and Hutto 1982, Medin and Clary 1991). The strong community-wide effects seen on the Missouri may be related to changes in vegetation that take place with continued grazing over long time scales (Ohmart 1994). High-nesting birds and primary cavity nesters may escape the immediate effects of grazing, but as cottonwood and aspen forests age, lack of recruitment of new trees causes a reduction in small and eventually large tree classes, which will affect the density of cavity nesters (Sedgwick and Knopf 1990, Dobkin et al. 1995) and the density of high-nesting species in general. This process may be well advanced in grazed locations along the Missouri, but is unlikely where grazing has been less continuous. Our results comparing low-nesting species to high-nesting species further support this possibility. Low open-cup nesting species have been shown to be particularly sensitive to grazing due to the large effects cattle have on the lower strata of vegetation (Sedgwick and Knopf 1987, Saab et al. 1995, Saab 1998). We also found that while both low and high nesting species had lower detection frequencies in grazed areas, these differences were greater for low nesting species. Along the Missouri, however, equally strong differences were found for both low- and high-nesting species, suggesting that long-term grazing may have affected canopy structure, snag retention, and recruitment of trees into the canopy (Ohmart 1994).

Cowbirds and Landscapes

Cowbirds could pose regional threats to riparian avifaunas due to their ubiquitous nature, their tendency to reach high densities in riparian areas (Tewksbury et al. 1999, Ward and Smith 2000), and the effects of parasitism both on individual hosts (Pease and Grzybowski 1995, Woodworth 1999) and on community composition (De Groot et al. 1999). Because of this, much work has examined landscape-scale effects on cowbird abundance and parasitism pressure locally (Gustafson and Crow 1994, Coker and Capen 1995, Gates and Evans 1998, Hejl and Young 1999; Tewksbury et al. 1998, 1999; Young and Hutto 1999), regionally (Donovan et al. 1995b, 1997, 2000; Robinson et al. 1995a, Thompson et al. 2000) and nationally (Hochachka et al. 1999). The majority of this work investigated only one or two factors that could limit cowbird abundance, in contrast to our results, which suggest that multiple landscape components may be important in the western United States.

To date, the species that are most often affected by parasitism appear to be extremely habitat limited (Robinson et al. 1995b), suggesting that the primary cause of population decline is not parasitism but habitat loss. With the steady increase in human encroachment upon riparian systems, and the highly mobile nature and generalist feeding strategy of the cowbird (Thompson 1994, Robinson et al. 1995b), we already have lost most of our opportunity to set aside large riparian areas in landscapes that are remote enough to preclude cowbirds altogether. Thus most communities will be affected by cowbirds, and attention should shift to strategies for minimizing the effect of cowbirds at local and regional scales. We suggest that preserving and enhancing the size of deciduous areas that are surrounded by few human habitations and little agriculture will have the greatest benefit for host populations, as cowbirds in these landscapes are likely limited by feeding habitat. In largely agricultural landscapes, cowbirds are more likely limited by availability of host nests, not feeding areas; thus moderate reductions in feeding areas in these areas (feedlots, bird-feeders, corrals, livestock pastures) may have little effect on rates of brood parasitism.

Management Implications and Species of Particular Concern

Our data suggest that the greatest threats to western deciduous riparian systems are (1) continued deciduous habitat loss and reduction in riparian area, (2) continued cattle grazing in remaining deciduous systems, and (3) increasing concentration of homes and farms along major riparian systems in the western United States. All of these factors are likely to have negative effects on bird communities in deciduous riparian areas, but rarely is it possible to extrapolate

TABLE 6. SUMMARY OF ALL SPECIES SIGNIFICANTLY LESS ABUNDANT IN AREAS WITH MORE HUMAN HABITATION, MORE AGRICULTURE, OR LESS DECIDUOUS HABITAT AT EITHER SCALE, OR IN GRAZED HABITATS, TESTED IN AT LEAST TWO RIPARIAN SYSTEMS

Species	More human habitation	More agriculture	Less deciduous	Grazing	Negative responses	Net negative–positive responses	West BBS[a]
Red-naped Sapsucker	0/–	–/0	–/–		4	4	
MacGillivray's Warbler	–/–	–/–	+/+		4	2	
Song Sparrow	+/–	+/0	–/–	–	4	2	**
Western Scrub-jay	–/0	–/–			3	3	
Veery	0/–	–/0	–/0		3	3	
Warbling Vireo	0/–	–/–			3	3	
Red-eyed Vireo			–/–	–	3	3	**
Yellow-rumped Warbler	0/–	–/–			3	3	
Black-capped Chickadee		–0/+	–/–	–	3	2	**
Townsend's Warbler	–/0	–/–	+/+		3	1	
Ruffed Grouse	0/–	–/0			2	2	*
American Crow	0/–	0/–			2	2	
Violet-green Swallow		–/–			2	2	
Swainson's Thrush	0/–	0/–			2	2	
Gray Catbird			–/0	–	2	2	
Fox Sparrow			–/0	–	2	2	
Dusky Flycatcher	0/–	–/0		+	2	1	*
Orange-crowned Warbler		–/0	–/+		2	1	*
Western Wood-pewee	+/+	+/+	–/0	–	2	–2	**
American Robin	+/+	+/+	0/–	–	2	–2	
Cedar Waxwing	+/0	+/0	–/–		2	–2	
Yellow Warbler	+/+	+/+	–/–		2	–2	
Nuttall's Woodpecker		–/0			1	1	
Hairy Woodpecker				–	1	1	
Least Flycatcher			–/0		1	1	
Ash-throated Flycatcher	–/0				1	1	
Cordilleran Flycatcher				–	1	1	
Western Kingbird		0/–			1	1	
Northern Rough-winged Swallow		–/0			1	1	
Bewick's Wren			–/0		1	1	
American Redstart			–/0		1	1	
Western Tanager	0/–				1	1	
Green-tailed towhee				–	1	1	
Chipping Sparrow		–/0			1	1	**
White-breasted Nuthatch		+/0	0/–		1	0	
Ruby-crowned Kinglet		–/0	0/+		1	0	
Black-billed Magpie		+/+	0/–		1	–1	
Black-headed Grosbeak	+/0	+/0	–/0		1	–1	
Red-shafted Flicker	+/0	+/0	0/–		1	–2	*
Red-winged Blackbird	+/+	+/+	0/–		1	–3	**
Willow Flycatcher	+/+	+/+	0/–		1	–3	**

Notes: Significantly ($P = 0.05$) lower detection frequency (–), significantly higher detection frequency (+), and no significant difference in detection frequency (0) are listed for each species in which at least 2 river systems were used in the analysis. Significant effects at local and regional scales are listed (local/regional). Species are ranked by the number of negative responses and the net (negative – positive) responses.

[a] Trend estimates from the Western Breeding Bird Survey region (Sauer et al. 2000). Species with a declining trend ($P < 0.25$) in the past 20 years, or over the course of the entire survey period, are single-starred (*) and species showing significant declines ($P < 0.05$) are marked with double stars (**).

from local studies to regional population trends. The data provided here allow us to highlight consistent trends, and by summarizing the responses to individual land uses we can also identify those species that appear to be at particular risk due to human landscape modification and livestock grazing (Table 6). We ranked each species based on the number of negative responses (lower abundance due to grazing, higher amounts of human habitation or agriculture, or lower amounts of deciduous habitat) making the assumption that species vulnerable to multiple human land-uses should receive greater attention than species vulnerable to only one type of land-use. Ten species had at least three negative responses. Of these, the Veery, MacGillivray's

Warbler, Song Sparrow, Warbling Vireo, and Red-eyed Vireo may be the most at risk, as all but the Warbling Vireo nest lower in dense vegetation (Ehrlich et al. 1988; J. J. Tewksbury unpubl. data) and all frequently suffer brood parasitism (Friedmann et al. 1977; J. J. Tewksbury unpubl. data). These species were all less abundant in landscapes with high human habitation and agriculture or low amounts of riparian habitat, and three respond negatively to livestock grazing. In addition, all of these, as well as the Red-naped Sapsucker, are found almost exclusively in deciduous vegetation. We suggest that these species should be monitored closely in western riparian habitats, and research should be initiated to examine mechanisms behind these patterns.

CONCLUSIONS

Management that focuses on enhancing the size of remaining deciduous riparian areas and reducing cattle grazing on these areas is likely to produce the greatest benefits for bird species dependent on western deciduous riparian habitats. In addition, strict limitations on building in floodplains will reduce the need for absolute flood control on riparian systems, which results in reduced riparian area. Protecting the few areas where riparian systems run through landscapes that are relatively free of human disturbance should be a high conservation priority both to protect the last unaltered pieces of one of the most endangered and important breeding habitats for western birds, and to preserve these few natural landscapes as benchmarks to use in examining the effects of land conversion. Without natural landscapes, we may lose sight of the conditions we are attempting to preserve.

ACKNOWLEDGMENTS

The results summarized here were produced by five independent field teams working over the past decade, and our work would not have been possible without the sharp eyes and strong ears of the many people who conducted surveys in these riparian systems. We thank R. Hutto, B. Kus, and L. George for their comments on earlier drafts of this manuscript.

APPENDIX 1

DESCRIPTIONS OF INDIVIDUAL RIPARIAN SYSTEMS

Sacramento

Location: all study sites are between Red Bluff and Colusa, California. Most sites are in remnant forest patches in the Sacramento National Wildlife Refuge.

Vegetation: the floodplain is a complex of early- to late-successional deciduous forests dominated successively by willows (*Salix* spp.) and cottonwood (*Populus* spp.), sycamore (*Platanus* spp.), ash (*Fraxinus* spp.), and valley oak (*Quercus lobata*). Adjacent upper terraces are dominated by valley oak. See Gaines (1974) for a detailed description of study sites.

Grazing: moderate to heavy cattle grazing for the past 15+ years on grazed sites. Ungrazed sites had been without cattle for at least 3 years before data collection.

San Joaquin

Location: all survey locations are in the northern portion of California's San Joaquin Valley, on levee roads adjacent to riparian stringers, grasslands, and recently (last decade) re-flooded grasslands in the San Luis National Wildlife Refuge.

Vegetation: similar to Sacramento River, dominated by willows and cottonwood, sycamore, ash, and valley oak. Willows and marsh vegetation are more common than valley oak.

Grazing: moderate to heavy cattle grazing for the past 15+ years on grazed sites. Ungrazed sites have been without cattle for at least 3 years before data collection.

Snake

Location: Sites are in an 80-km stretch just downstream of the Idaho/Wyoming border in eastern Idaho. For a detailed description of sites see Saab (1999).

Vegetation: Cottonwood (*Populus angustifolia*) forests. Understory species include dogwood (*Cornus stolonifera*), thin-leafed alder (*Alnus incana*), water birch (*Betula occidentalis*), and willows.

Grazing: moderate to heavy grazing for the past 30+ years on grazed sites. Ungrazed sites have been without cattle for at least three years before data collection.

Bitterroot

Location: survey locations were located along a 40-km stretch of the Bitterroot River and smaller tributaries throughout the Bitterroot Valley between Corvallis to the north and continuing past Darby to the south. See Tewksbury et al. (1998, 1999) for details of study sites.

Vegetation: cottonwood and willow dominate sites along the Bitterroot River, with dogwood, thin-leafed alder, and water birch in smaller quantities. Along tributaries, cottonwood, aspen, and willow are dominant.

Grazing: all study sites were ungrazed or rested for at least five years; thus the Bitterroot River is not included in our analysis of grazing effects.

Missouri

Location: ungrazed survey locations were located on the Charles M. Russell National Wildlife Refuge, and grazed survey locations were in a 40-km stretch of river bordering the refuge to the west.

Vegetation: riparian stands consist of mid- to late-seral riparian vegetation (Hansson et al. 1995) dominated by Great Plains cottonwood (*Populus deltoides*), green ash (*Fraxinus pennsylvanica*), and willow. Floodplains are bounded by the steep, highly eroded "Missouri Breaks," which rise to 300m from the floodplain and support upland vegetation dominated by shrubs.

Grazing: moderate to heavy grazing for the past 30–120 years on grazed sites, ungrazed sites have had no cattle for the past 30 years.

Hart Mountain

Location: all Hart Mountain sites were located in the northwestern Great Basin on the 115,000 ha Hart Mountain National Antelope Refuge (42°25′ N, 119°40′ W) in southeastern Oregon. Data were used from surveys conducted along small streams in five separate drainages.

Vegetation: riparian woodlands occurred as narrow ribbons of riparian habitat, primarily aspen and willows, surrounded by sagebrush (*Artemisia* spp.) steppe, or as dense stands of smaller-stature trees on sideslopes and snowpocket areas in the higher reaches of riparian drainages. For additional details see Dobkin et al. (1995, 1998).

Grazing: in the autumn of 1990, livestock were removed completely from the Hart Mountain refuge, ending continuous livestock use dating back to the 1870s. For this study, we classified data from 1991 (the first growing season following livestock removal) as "grazed," and data from 1993 (the third growing season following livestock removal) as "rested" or "ungrazed." We did not use data for 1992.

Sheldon

Location: all Sheldon sites were on the Sheldon National Wildlife Refuge located in the northwestern corner of Nevada, approximately 55 km southeast of Hart Mountain. Riparian areas occur mostly as narrow valleys and canyons bordered by the steep rimrock of tablelands.

Vegetation: riparian habitat is severely limited at Sheldon, and nearly all riparian habitat in this study consisted of degraded willow-dominated areas.

Grazing: as at Hart Mountain, livestock were removed from the Sheldon Refuge in the autumn of 1990 following continuous livestock use dating back to the 1870s. For this study, we classified data from 1991 (the first growing season following livestock removal) as "grazed," and data from 1993 (the third growing season following livestock removal) as "rested" or "ungrazed." We did not use data for 1992.

APPENDIX 2. COMMON AND SCIENTIFIC NAMES AND ECOLOGICAL ATTRIBUTES OF ALL SPECIES ANALYZED

Common Name	Scientific Name	Rivers	Mean abundance rank	Migration guild	Host guild	Nest Type	Nest Height
Pied-billed Grebe	*Podilymbus podiceps*	1	61.5		non host		
Western Grebe	*Aechmophorus occidentalis*	1	84		non host		
American White Pelican	*Pelecanus erythrorhynchos*	1	72		non host	open	<2.5m
Double-crested Cormorant	*Phalacrocorax auritus*	1	53		non host	open	
American Bittern	*Botaurus lentiginosus*	1	54		non host	open	<2.5m
Great Blue Heron	*Ardea herodias*	4	44.5		non host	open	>5m
Great Egret	*Ardea alba*	1	34		non host	open	>5m
Black-crowned Night-Heron	*Nycticorax nycticorax*	1	47		non host	open	>5m
Turkey Vulture	*Cathartes aura*	3	46	Short-distance	non host		
Canada Goose	*Branta canadensis*	1	54		non host	open	<2.5m
Wood Duck	*Aix sponsa*	3	58.5		non host	secondary cavity	>5m
Gadwall	*Anas strepera*	2	56		non host	open	<2.5m
Mallard	*Anas platyrhynchos*	6	35.6		non host	open	<2.5m
Green-winged Teal	*Anas crecca*	1	63		non host	open	<2.5m
Cinnamon Teal	*Anas cyanoptera*	1	23		non host	open	<2.5m
Hooded Merganser	*Lophodytes cucullatus*	1	86		non host	secondary cavity	
Common Merganser	*Mergus merganser*	2	59.3		non host	secondary cavity	>5m
Red-breasted Merganser	*Mergus serrator*	1	69		non host		
Osprey	*Pandion haliaetus*	2	61.5	Short-distance	non host	open	>5m
Bald Eagle	*Haliaeetus leucocephalus*	1	81.5	Short-distance	non host	open	>5m
Northern Harrier	*Circus cyaneus*	2	48.3	Short-distance	non host		<2.5m
Sharp-shinned Hawk	*Accipiter striatus*	1	48.5	Short-distance	non host	open	>5m
Cooper's Hawk	*Accipiter cooperii*	2	83.3	Short-distance	non host	open	>5m
Northern Goshawk	*Accipiter gentilis*	2	37	Resident	non host	open	>5m
Red-shouldered Hawk	*Buteo lineatus*	1	50	Short-distance	non host		
Swainson's Hawk	*Buteo swainsoni*	1	45	Long-distance	non host	open	>5m
Red-tailed Hawk	*Buteo jamaicensis*	6	43.1	Short-distance	non host	open	>5m
Golden Eagle	*Aquila chrysaetos*	1	53	Short-distance	non host	open	>5m
American Kestrel	*Falco sparverius*	5	47.8	Short-distance	non host	secondary cavity	>5m
Ring-necked Pheasant	*Phasianus colchicus*	1	23	Resident	non host	open	<2.5m
Ruffed Grouse	*Bonasa umbellus*	2	39	Resident	non host	open	<2.5m
Wild Turkey	*Meleagris gallopavo*	1	69	Resident	non host	open	<2.5m
California Quail	*Callipepla californica*	2	23.5	Resident	non host	open	<2.5m
Virginia Rail	*Rallus limicola*	1	39		non host	open	<2.5m
American Coot	*Fulica americana*	1	51		non host		
Killdeer	*Charadrius vociferus*	4	53.1	Short-distance	non host	open	<2.5m
Solitary Sandpiper	*Tringa solitaria*	1	41		non host		
Spotted Sandpiper	*Actitis macularia*	4	48.1		non host	open	<2.5m
Common Snipe	*Gallinago gallinago*	2	55.8		non host	open	<2.5m

APPENDIX 2. CONTINUED

Common Name	Scientific Name	Rivers	Mean abundance rank	Migration guild	Host guild	Nest Type	Nest Height
Wilson's Phalarope	*Phalaropus tricolor*	1	64		non host	open	<2.5m
Red-necked Phalarope	*Phalaropus lobatus*	2	46.5		non host		
Caspian Tern	*Sterna caspia*	1	58		non host	open	<2.5m
Plain Pigeon	*Columba inornata*	1	86	Resident	non host		
Mourning Dove	*Zenaida macroura*	7	22.1	Short-distance		open	
Yellow-billed Cuckoo	*Coccyzus americanus*	2	61.5	Long-distance	non host	open	>5m
Great Horned Owl	*Bubo virginianus*	5	49.9	Resident	non host	open	>5m
Barred Owl	*Strix varia*	1	61	Resident	non host	secondary cavity	>5m
Long-eared Owl	*Asio otus*	2	63.5		non host	open	
Short-eared Owl	*Asio flammeus*	1	79	Short-distance	non host	open	<2.5m
Lesser Nighthawk	*Chordeiles acutipennis*	1	61.5	Long-distance	non host	open	<2.5m
Common Nighthawk	*Chordeiles minor*	1	68	Long-distance	non host	open	<2.5m
White-throated Swift	*Aeronautes saxatalis*	2	46	Long-distance	non host		
Black-chinned Hummingbird	*Archilochus alexandri*	1	31	Long-distance	non host	open	>5m
Anna's Hummingbird	*Calypte anna*	1	66	Resident	non host	open	
Calliope Hummingbird	*Stellula calliope*	3	53.5	Long-distance	non host	open	<2.5m
Broad-tailed Hummingbird	*Selasphorus platycercus*	3	60	Long-distance	non host	open	
Rufous Hummingbird	*Selasphorus rufus*	2	40	Long-distance	non host	open	>5m
Belted Kingfisher	*Ceryle alcyon*	3	53		non host		
Lewis's Woodpecker	*Melanerpes lewis*	1	80	Short-distance	non host	primary cavity	>5m
Acorn Woodpecker	*Melanerpes formicivorus*	1	45.5	Resident	non host	primary cavity	>5m
Red-naped Sapsucker	*Sphyrapicus nuchalis*	3	17.5	Short-distance	non host	primary cavity	>5m
Red-breasted Sapsucker	*Sphyrapicus ruber*	1	64	Short-distance	non host	primary cavity	>5m
Nuttall's Woodpecker	*Picoides nuttallii*	2	10.5	Resident	non host	primary cavity	
Downy Woodpecker	*Picoides pubescens*	6	35	Resident	non host	primary cavity	>5m
Hairy Woodpecker	*Picoides villosus*	5	36.9	Resident	non host	primary cavity	>5m
Northern Flicker	*Colapetes auratus*	6	20.5	Short-distance	non host	primary cavity	>5m
Pileated Woodpecker	*Dryocopus pileatus*	1	77	Resident	non host	primary cavity	>5m
Olive-sided Flycatcher	*Contopus cooperi*	1	53.5	Long-distance		open	
Western Wood-Pewee	*Contopus sordidulus*	6	13	Long-distance		open	>5m
Willow Flycatcher	*Empidonax traillii*	4	48.6	Long-distance	primary host	open	<2.5m
Least Flycatcher	*Empidonax minimus*	2	34.3	Long-distance		open	
Hammond's Flycatcher	*Empidonax hammondii*	2	51	Long-distance	non host	open	>5m
Gray Flycatcher	*Empidonax wrightii*	1	20.5	Long-distance	non host	open	
Dusky Flycatcher	*Empidonax oberholseri*	5	27.7	Long-distance		open	
Cordilleran Flycatcher	*Empidonax occidentalis*	3	46	Long-distance		open	<2.5m
Pacific-slope Flycatcher	*Empidonax difficulis*	1	45.5	Long-distance		open	<2.5m
Ash-throated Flycatcher	*Myiarchus cinerascens*	2	7	Long-distance	non host	open	
Say's Pheobe	*Sayornis saya*	1	63	Short-distance			

APPENDIX 2. CONTINUED

Common Name	Scientific Name	Rivers	Mean abundance rank	Migration guild	Host guild	Nest Type	Nest Height
Black Phoebe	*Sayornis nigricans*	2	40.5	Resident			
Western Kingbird	*Tyrannus verticalis*	3	26.7	Long-distance		open	>5m
Eastern Kingbird	*Tyrannus tyrannus*	3	47.3	Long-distance	primary host	open	>5m
Loggerhead Shrike	*Lanius ludovicianus*	2	34.5	Short-distance	non host	open	
Warbling Vireo	*Vireo gilvus*	7	24.1	Long-distance	primary host	open	>5m
Red-eyed Vireo	*Vireo olivaceus*	4	45	Long-distance	primary host	open	>5m
Gray Jay	*Perisoreus canadensis*	1	90.5	Resident	non host	open	
Steller's Jay	*Cyanocitta stelleri*	1	70	Resident	non host	open	>5m
Western Scrub-Jay	*Aphelocoma californica*	2	12.5	Resident	non host		
Clark's Nutcracker	*Nucifraga columbiana*	1	62.5	Resident	non host	open	>5m
Black-billed Magpie	*Pica hudsonia*	5	22	Resident	non host	open	>5m
Yellow-billed Magpie	*Pica nuttalli*	1	21	Resident	non host	open	
American Crow	*Corvus brachyrhynchos*	4	50.8	Short-distance	non host	open	>5m
Common Raven	*Corvus corax*	3	54.3	Resident	non host	open	>5m
Horned Lark	*Eremophila alpestris*	1	53	Short-distance		open	<2.5m
Tree Swallow	*Tachycineta bicolor*	6	20.7	Short-distance		secondary cavity	>5m
Violet-green Swallow	*Tachycineta thalassina*	4	30.8	Long-distance	non host	secondary cavity	>5m
Northern Rough-winged Swallow	*Stelgidopteryx serripennis*	4	44.3	Long-distance	non host		
Bank Swallow	*Riparia riparia*	1	29	Long-distance			
Cliff Swallow	*Petrochelidon pyrrhonota*	6	44.8	Long-distance			
Barn Swallow	*Hirundo rustica*	3	47.5	Long-distance			
Black-capped Chickadee	*Poecile atricapilla*	3	12.8	Resident		secondary cavity	
Mountain Chickadee	*Poecile gambeli*	3	53.2	Resident	non host	secondary cavity	<2.5m
Chestnut-backed Chickadee	*Poecile rufescens*	1	44.5	Resident	non host	secondary cavity	<2.5m
Oak Titmouse	*Baeolophus inornatus*	2	32	Resident			
Bushtit	*Psaltriparus minimus*	1	21	Resident		open	
Red-breasted Nuthatch	*Sitta canadensis*	4	48.3	Short-distance	non host	primary cavity	>5m
White-breasted Nuthatch	*Sitta carolinensis*	3	42	Resident		secondary cavity	>5m
Pygmy Nuthatch	*Sitta pygmaea*	1	60	Resident	non host	primary cavity	>5m
Brown Creeper	*Certhia americana*	1	75.5	Short-distance		open	>5m
Rock Wren	*Salpinctes obsoletus*	1	29	Short-distance			
Bewick's Wren	*Thryomanes bewickii*	2	5	Short-distance		secondary cavity	<2.5m
House Wren	*Troglodytes aedon*	7	14.6	Long-distance		secondary cavity	
Winter Wren	*Troglodytes troglodytes*	1	48	Short-distance		open	<2.5m
Marsh Wren	*Cistothorus palustris*	2	33	Short-distance	non host	open	<2.5m
Golden-crowned Kinglet	*Regulus satrapa*	1	23	Short-distance		open	>5m
Ruby-crowned Kinglet	*Regulus calendula*	4	37.9	Short-distance		open	>5m

APPENDIX 2. CONTINUED

Common Name	Scientific Name	Rivers	Mean abundance rank	Migration guild	Host guild	Nest Type	Nest Height
Blue-gray Gnatcatcher	*Polioptila caerulea*	1	63	Long-distance	primary host	open	
Western Bluebird	*Sialia mexicana*	1	39	Short-distance		secondary cavity	<2.5m
Mountain Bluebird	*Sialia currucoides*	2	34.3	Short-distance		secondary cavity	
Townsend's Solitaire	*Myadestes townsendi*	2	73	Short-distance	non host	open	<2.5m
Veery	*Catharus fuscescens*	2	20	Long-distance	primary host	open	<2.5m
Swainson's Thrush	*Catharus ustulatus*	6	32.3	Long-distance		open	
American Robin	*Turdus migratorius*	7	13.7	Short-distance		open	
Gray Catbird	*Dumetella carolinensis*	3	34.3	Long-distance		open	<2.5m
Northern Mockingbird	*Mimus polyglottos*	1	26	Resident		open	
Sage Thrasher	*Oreoscoptes montanus*	1	45	Short-distance		open	<2.5m
Brown Thrasher	*Toxostoma rufum*	1	18	Short-distance		open	<2.5m
California Thrasher	*Toxostoma redivivum*	1	46	Resident	non host	open	<2.5m
European Starling	*Sturnus vulgaris*	7	17.9	Short-distance	non host	secondary cavity	>5m
Cedar Waxwing	*Bombycilla cedrorum*	4	16	Short-distance		open	
Orange-crowned Warbler	*Vermivora celata*	4	32.1	Long-distance		open	<2.5m
Nashville Warbler	*Vermivora ruficapilla*	2	64.5	Long-distance		open	<2.5m
Yellow Warbler	*Dendroica petechia*	7	16.1	Long-distance	primary host	open	
Yellow-rumped Warbler	*Dendroica coronata*	5	33.6	Short-distance	primary host	open	>5m
Townsend's Warbler	*Dendroica townsendi*	5	51.5	Long-distance		open	>5m
Hermit Warbler	*Dendroica occidentalis*	1	58.5	Long-distance		open	>5m
American Redstart	*Setophaga ruticilla*	2	19	Long-distance	primary host	open	
Ovenbird	*Seiurus aurocapillus*	2	45	Long-distance	primary host	open	<2.5m
Northern Waterthrush	*Seiurus noveboracensis*	1	51	Long-distance		open	<2.5m
MacGillivray's Warbler	*Oporornis tolmiei*	5	37.4	Long-distance		open	<2.5m
Common Yellowthroat	*Geothlypis trichas*	5	22	Long-distance	primary host	open	<2.5m
Wilson's Warbler	*Wilsonia pusilla*	5	26.6	Long-distance		open	<2.5m
Yellow-breasted Chat	*Icteria virens*	5	38.7	Long-distance	primary host	open	<2.5m
Western Tanager	*Piranga ludoviciana*	6	39.8	Long-distance		open	>5m
Green-tailed Towhee	*Pipilo chlorurus*	2	36	Long-distance		open	<2.5m
Spotted Towhee	*Pipilo maculatus*	5	32.7	Short-distance	primary host	open	<2.5m
California Towhee	*Pipilo crissalis*	2	34	Long-distance		open	<2.5m
Chipping Sparrow	*Spizella passerina*	4	46	Long-distance	primary host	open	
Brewer's Sparrow	*Spizella breweri*	2	20.5	Long-distance		open	<2.5m
Vesper Sparrow	*Pooecetes gramineus*	3	45.3	Short-distance	primary host	open	<2.5m
Black-throated Sparrow	*Amphispiza bilineata*	1	35	Short-distance		open	<2.5m
Sage Sparrow	*Amphispiza belli*	1	39	Short-distance		open	<2.5m
Lark Sparrow	*Chondestes grammacus*	3	48	Long-distance		open	<2.5m
Savannah Sparrow	*Passerculus sandwichensis*	2	33.8	Short-distance		open	<2.5m
Fox Sparrow	*Passerella iliaca*	2	20.5	Short-distance		open	<2.5m

APPENDIX 2. CONTINUED

Common Name	Scientific Name	Rivers	Mean abundance rank	Migration guild	Host guild	Nest Type	Nest Height
Song Sparrow	*Melospiza melodia*	6	15.4	Short-distance	primary host	open	<2.5m
White-crowned Sparrow	*Zonotrichia leucophrys*	1	10	Short-distance		open	<2.5m
Golden-crowned Sparrow	*Zonotrichia atricapilla*	2	66	Short-distance		open	
Black-headed Grosbeak	*Pheucticus melanocephalus*	7	18.9	Long-distance		open	
Blue Grosbeak	*Guiraca caerulea*	2	38.5	Long-distance	primary host	open	
Lazuli Bunting	*Passerina amoena*	7	30.1	Long-distance	primary host	open	<2.5m
Red-winged Blackbird	*Agelaius phoeniceus*	6	29.1	Short-distance	primary host	open	<2.5m
Tricolored Blackbird	*Agelaius tricolor*	1	52	Resident		open	<2.5m
Western Meadowlark	*Sturnella neglecta*	5	37.2	Short-distance		open	<2.5m
Yellow-headed Blackbird	*Xanthocephalus xanthocephalus*	1	63	Long-distance		open	<2.5m
Brewer's Blackbird	*Euphagus cyanocephalus*	7	29.4	Short-distance	primary host	open	
Common Grackle	*Quiscalus quiscula*	1	15	Short-distance		open	>5m
Brown-headed Cowbird	*Molothrus ater*	7	7.21	Short-distance	non host		
Bullock's Oriole	*Icterus bullockii*	7	21.3	Long-distance		open	>5m
Purple Finch	*Carpodacus purpureus*	1	78	Short-distance		open	>5m
Cassin's Finch	*Carpodacus cassinii*	3	35.7	Short-distance	non host	open	>5m
House Finch	*Carpodacus mexicanus*	5	40.4	Short-distance		open	>5m
Red Crossbill	*Loxia curvirostra*	1	9	Short-distance	non host	open	>5m
Pine Siskin	*Carduelis pinus*	4	31.6	Short-distance		open	>5m
Lesser Goldfinch	*Carduelis psaltria*	1	17	Short-distance		open	>5m
American Goldfinch	*Carduelis tristis*	5	17.8	Short-distance	primary host	open	>5m
Evening Grosbeak	*Coccothraustes vespertinus*	1	11	Short-distance		open	>5m
House Sparrow	*Passer domesticus*	1	41	Resident	non host		

Notes: the number of river systems in which the species was detected (Rivers) and the mean rank of each species (Mean abundance rank; 1 = most often detected species on a river). All species were ranked in descending order of detection frequency within each river system. Mean rank abundance is the average rank across all rivers where the species were detected. Species membership in migration, cowbird host, nest type, and nest height guilds is also included.

APPENDIX 3. PEARSON CORRELATIONS (AND P-VALUES) FOR ALL LANDSCAPE VARIABLES WITHIN EACH RIVER SYSTEM (SEE TABLE 1 FOR SAMPLE SIZES)

River system	Landscape variable	Local human habitation	Local agriculture	Local deciduous forest	Local coniferous forest	Regional human habitation	Regional agriculture	Regional deciduous forest
Sacramento	Local agriculture	0.383 (0.004)						
	Local deciduous forest	−0.139 (0.311)	−0.322 (0.016)					
	Local coniferous forest	. .	. .	. .				
	Regional human habitation	−0.220 (0.106)	−0.001 (0.993)	−0.251 (0.064)	. .			
	Regional agriculture	0.714 (<0.001)	0.688 (<0.001)	−0.125 (0.364)	. .	−0.166 (0.227)		
	Regional deciduous forest	0.774 (<0.001)	0.179 (0.190)	−0.239 (0.079)	. .	−0.166 (0.225)	0.630 (<0.001)	
	Regional coniferous forest	. .	. .	. .	. .	. .	. .	. .
San Joaquin	Local agriculture	0.612 (<0.001)						
	Local deciduous forest	0.275 (0.044)	0.369 (0.006)					
	Local coniferous forest	. .	. .	. .				
	Regional human habitation	. .	. .	. .	. .			
	Regional agriculture	0.215 (0.119)	0.577 (<0.001)	0.213 (0.1216)	. .	. .		
	Regional deciduous forest	. .	. .	. .	. .	. .	. .	
	Regional coniferous forest	. .	. .	. .	. .	. .	. .	. .
Snake	Local agriculture	0.074 (0.604)						
	Local deciduous forest	−0.304 (0.029)	−0.555 (<0.001)					
	Local coniferous forest	−0.022 (0.876)	−0.706 (<0.001)	0.035 (0.805)				
	Regional human habitation	0.286 (0.040)	0.665 (<0.001)	−0.099 (0.485)	−0.648 (<0.001)			
	Regional agriculture	−0.377 (0.006)	0.589 (<0.001)	0.075 (0.598)	−0.562 (<0.001)	0.596 (<0.001)		
	Regional deciduous forest	0.417 (0.002)	−0.409 (0.003)	−0.114 (0.421)	0.343 (0.013)	−0.410 (0.003)	−0.884 (<0.001)	

APPENDIX 3. CONTINUED

River system	Landscape variable	Local human habitation	Local agriculture	Local deciduous forest	Local coniferous forest	Regional human habitation	Regional agriculture	Regional deciduous forest
	Regional coniferous forest	0.398 (0.003)	−0.546 (<0.001)	−0.094 (0.509)	0.561 (<0.001)	−0.556 (<0.001)	−0.927 (<0.001)	0.798 (<0.001)
Bitterroot	Local agriculture	0.675 (<0.001)						
	Local deciduous forest	0.437 (<0.001)	0.473 (<0.001)					
	Local coniferous forest	−0.623 (<0.001)	−0.851 (<0.001)	−0.688 (<0.001)				
	Regional human habitation	0.242 (0.008)	0.439 (<0.001)	0.335 (<0.001)	−0.581 (<0.001)			
	Regional agriculture	0.470 (<0.001)	0.603 (<0.001)	0.700 (<0.001)	−0.717 (<0.001)	0.561 (<0.001)		
	Regional deciduous forest	0.418 (<0.001)	0.537 (<0.001)	0.690 (<0.001)	−0.651 (<0.001)	0.544 (<0.001)	0.951 (<0.001)	
	Regional coniferous forest	−0.574 (<0.001)	−0.759 (<0.001)	-0.637 (<0.001)	0.869 (<0.001)	−0.623 (<0.001)	−0.909 (<0.001)	−0.833 (<0.001)
Missouri	Local agriculture	. .						
	Local deciduous forest	. .	0.049 (0.801)					
	Local coniferous forest	. .	0.150 (0.438)	−0.575 (0.001)				
	Regional human habitation	. .	. .	. .	. .			
	Regional agriculture	. .	0.594 (<0.001)	−0.061 (0.754)	0.186 (0.333)	. .		
	Regional deciduous forest	. .	0.104 (0.593)	0.824 (<0.001)	−0.407 (0.028)	. .	−0.003 (0.988)	
	Regional coniferous forest	. .	0.201 (0.295)	0.777 (<0.001)	−0.158 (0.413)	. .	0.233 0.2245	0.809 (<0.001)
Hart	Local coniferous forest	. .	. .	−0.628 (<0.001)				
	Regional human habitation	. .	. .	. .	. .			
	Regional agriculture	. .	. .	. .	. .	. .		

APPENDIX 3. CONTINUED

River system	Landscape variable	Local human habitation	Local agriculture	Local deciduous forest	Local coniferous forest	Regional human habitation	Regional agriculture	Regional deciduous forest
	Regional deciduous forest	.	.	0.226	0.373	.	.	
		.	.	(0.155)	(0.016)	.	.	
	Regional coniferous forest	.	.	0.599	−0.123	.	.	0.433
		.	.	(<0.001)	(0.445)	.	.	(0.005)
Sheldon	Local deciduous forest	−0.248	.					
		(0.305)	.					
	Local coniferous forest	0.344	.	−0.195				
		(0.150)	.	(0.424)				
	Regional human habitation	.	.	.	.			
		.	.	.	.			
	Regional agriculture	−0.380	.	0.924	−0.199	.		
		(0.109)	.	(<0.001)	(0.413)	.		
	Regional deciduous forest	.	.	.	.	.	.	
		.	.	.	.	.	.	
	Regional coniferous forest	.	.	.	.	.	.	.
		.	.	.	.	.	.	.

APPENDIX 4. EFFECTS OF LANDSCAPE VARIABLES WITHIN INDIVIDUAL RIVER SYSTEMS ON TOTAL DETECTIONS (TOTAL BIRDS), TOTAL RICHNESS (RICHNESS), BROWN-HEADED COWBIRDS, COWBIRD HOST GUILDS, AND MIGRATION GUILDS (SEE APPENDIX 2)

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
Local Human Habitation									
Sacramento									
Σ	Dir	Pos		Pos	Pos	Neg	Pos		
	B	0.13		0.19	0.19	−0.27	0.20		
	R^2	0.016		0.036	0.035	0.072	0.039		
	P	0.361		0.166	0.173	0.048	0.150		
#	Dir	Neg			Neg	Neg	Neg	Neg	Neg
	Inc	38%			31%	34%	33%	29%	36%
	P	0.038			0.267	0.074	0.100	0.180	0.230
San Joaquin									
Σ	Dir			Pos	Pos	Neg		Neg	Pos
	B			0.25	0.26	−0.12		−0.29	0.12
	R^2			0.063	0.031	0.015		0.086	0.015
	P			0.062	0.049	0.375		0.028	0.371
#	Dir	Neg				Neg	Neg		
	Inc	43%				41%	35%		
	P	0.313				0.377	0.263		
Snake									
Σ	Dir	Neg		Neg	Neg	Pos	Neg	Pos	Neg
	B	−0.38		−0.17	−0.52	0.11	−0.44	0.17	−0.23
	R^2	0.143		0.029	0.265	0.013	0.187	0.030	0.051
	P	0.005*		0.224	<0.001*	0.415	0.001*	0.211	0.104
#	Dir	Neg			Neg	Neg	Neg	Neg	Neg
	Inc	33%			21%	35%	29%	14%	37%
	P	0.015			0.057	0.210	0.064	0.125	0.359
Bitterroot									
Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos	Pos	Pos
	B	0.26	0.15	0.38	0.40	0.25	0.11	0.16	0.19
	R^2	0.070	0.021	0.141	0.158	0.060	0.012	0.024	0.035
	P	0.004	0.143	<0.001*	<0.001*	0.007	0.238	0.091	0.040
#	Dir	Pos			Pos	Pos			
	Inc	57%			75%	61%			
	P	0.213			0.077	0.243			
Sheldon									
Σ	Dir	Neg	Neg		Neg	Neg	Neg	Neg	Neg
	B	−0.51	−0.42		−0.42	−0.43	−0.31	−0.33	−0.39
	R^2	0.259	0.176		0.175	0.183	0.099	0.110	0.150
	P	0.026	0.074		0.075	0.068	0.190	0.166	0.101

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
#	Dir	Neg			Neg	Neg	Neg		Neg
	Inc	19%			0%	18%	18%		23%
	P	<0.001			<0.001	0.004	0.001		0.011
Regional Human Habitation									
Snake									
Σ	Dir			Pos	Neg	Pos	Neg	Pos	Pos
	B			0.36	−0.48	0.20	−0.17	0.23	0.23
	R^2			0.126	0.230	0.040	0.034	0.051	0.054
	P			0.009	<0.001*	0.153	0.228	0.105	0.095
#	Dir	Neg				Neg	Neg		
	Inc	40%				35%	38%		
	P	0.178				0.210	0.307		
Bitterroot									
Σ	Dir	Pos	Pos	Pos	Pos	Pos	Neg	Pos	Pos
	B	0.16	0.17	0.33	0.17	0.28	−0.10	0.16	0.24
	R^2	0.026	0.030	0.106	0.030	0.079	0.009	0.024	0.058
	P	0.078	0.080	<0.001*	0.060	0.002	0.289	0.091	0.008
#	Dir								
	Inc								
	P								
Local Agriculture									
Sacramento									
Σ	Dir	Pos	Pos	Pos			Pos		
	B	0.21	0.16	0.24			0.28		
	R^2	0.044	0.025	0.057			0.079		
	P	0.126	0.245	0.080			0.037		
#	Dir					Neg			
	Inc					38%			
	P					0.230			
San Joaquin									
Σ	Dir			Pos	Pos				Pos
	B			0.27	0.24				0.19
	R^2			0.074	0.070				0.035
	P			0.042	0.070				0.167
#	Dir						Neg		
	Inc						38%		
	P						0.381		
Snake									

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
Σ	Dir	Pos		Pos	Neg	Pos	Neg	Pos	Pos
	B	0.23		0.25	−0.54	0.48	−0.13	0.39	0.37
	R^2	0.053		0.063	0.294	0.225	0.013	0.132	0.134
	P	0.097		0.069	<0.001*	<0.001*	0.364	0.004	0.007
#	Dir	Neg				Neg	Neg		
	Inc	40%				36%	26%		
	P	0.310				0.286	0.035		
Bitterroot									
Σ	Dir	Pos	Pos	Pos	Pos	Pos		Pos	Pos
	B	0.26	0.29	0.38	0.33	0.31		0.22	0.33
	R^2	0.068	0.082	0.142	0.106	0.095		0.049	0.109
	P	0.004	0.003	<0.001*	<0.001*	0.001*		0.015	<0.001*
#	Dir	Pos			Pos	Pos		Pos	Pos
	Inc	69%			71%	71%		69%	61%
	P	<0.001*			0.143	0.018		0.210	0.281
Regional Agriculture									
Sacramento									
Σ	Dir	Pos	Pos	Pos			Pos	Pos	Pos
	B	0.34	0.25	0.24			0.27	0.30	0.22
	R^2	0.113	0.063	0.057			0.075	0.091	0.049
	P	0.012*	0.065	0.078			0.044	0.026	0.105
#	Dir			Neg					Neg
	Inc			38%					33%
	P			0.265					0.189
San Joaquin									
Σ	Dir		Neg	Pos		Pos	Neg		Pos
	B		−0.2	0.26		0.15	−0.16		0.33
	R^2		0.041	0.068		0.023	0.037		0.111
	P		0.141	0.052		0.260	0.246		0.012*
#	Dir								
	Inc								
	P								
Snake									
Σ	Dir	Pos		Pos			Pos		Pos
	B	0.51		0.55			0.27		0.51
	R^2	0.256		0.298			0.068		0.260
	P	<0.001*		<0.001*			0.049		<0.001*
#	Dir	Neg				Neg	Neg		

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
	Inc	0.4				35%	33%		
	P	0.178				0.210	0.152		
Bitterroot									
Σ	Dir	Pos	Pos	Pos	Pos	Pos		Pos	Pos
	B	0.23	0.28	0.43	0.33	0.44		0.38	0.30
	R^2	0.083	0.081	0.202	0.109	0.191		0.148	0.089
	P	0.001*	0.004	<0.001*	<0.001*	v*		<0.001*	0.001*
#	Dir	Pos			Pos	Pos	Pos	Pos	Pos
	Inc	69%			71%	71%	64%	69%	65%
	P	<0.001*			0.143	0.018*	0.164	0.210	0.170
Missouri									
Σ	Dir	Pos	Pos		Pos	Pos	Pos	Neg	Pos
	B	0.45	0.28		0.45	0.26	0.39	−0.33	0.49
	R^2	0.202	0.078		0.203	0.065	0.158	0.111	0.235
	P	0.015	0.142		0.014*	0.180	0.038	0.078	0.008*
#	Dir	Pos			Pos			Neg	Pos
	Inc	60%			71%			20%	76%
	P	0.222			0.180			0.375	0.049
Local deciduous riparian									
Sacramento									
Σ	Dir	Neg	Neg	Neg		Neg	Neg		Pos
	B	−0.33	−0.31	−0.17		−0.13	−0.41		0.19
	R^2	0.107	0.093	0.028		0.017	0.167		0.036
	P	0.015*	0.024	0.218		0.345	0.002*		0.163
#	Dir	Neg			Neg	Neg	Neg		Neg
	Inc	33%			33%	29%	25%		38%
	P	0.007*			0.388	0.038*	0.023*		0.383
San Joaquin									
Σ	Dir	Pos	Pos	Pos	Pos	Pos		Pos	Pos
	B	0.27	0.12	0.13	0.244	0.27		0.28	0.24
	R^2	0.073	0.014	0.017	0.059	0.073		0.078	0.056
	P	0.044	0.396	0.343	0.070	0.045		0.038	0.080
#	Dir	Pos			Pos	Pos		Pos	
	Inc	67%			80%	69%		82%	
	P	0.009*			0.109	0.052		0.065	
Snake									
Σ	Dir				Pos	Neg	Pos	Neg	Neg
	B				0.40	−0.42	0.25	−0.25	−0.17
	R^2				0.160	0.175	0.053	0.062	0.029

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
	P				0.003*	0.002*	0.072	0.072	0.224
#	Dir	Pos			Pos		Pos		
	Inc	58%			71%		63%		
	P	0.281			0.180		0.307		
Bitterroot									
Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos	Pos	Pos
	B	0.37	0.34	0.40	0.51	0.34	0.13	0.32	0.30
	R^2	0.140	0.113	0.161	0.259	0.113	0.017	0.100	0.090
	P	<0.001*	0.001*	<0.001*	<0.001*	<0.001*	0.164	<0.001*	0.001*
#	Dir	Pos			Pos	Pos	Pos	Pos	Pos
	Inc	68%			75%	71%	61%	75%	68%
	P	0.001*			0.070	0.015	0.296	0.077	0.089
Missouri									
Σ	Dir	Pos	Pos		Pos	Pos	Pos	Pos	Pos
	B	0.59	0.27		0.34	0.25	0.65	0.37	0.33
	R^2	0.344	0.073		0.117	0.064	0.416	0.139	0.110
	P	0.001*	0.115		0.070	0.187	<0.001*	0.047	0.079
#	Dir	Pos					Pos		
	Inc	63%					72%		
	P	0.144					0.096		
Hart									
Σ	Dir				Neg	Neg		Pos	
	B				−0.43	−0.14		0.27	
	R^2				0.182	0.019		0.071	
	P				0.005*	0.390		0.093	
#	Dir	Neg			Neg			Pos	Neg
	Inc	40%			11%			80%	31%
	P	0.131			0.039*			0.375	0.063
Sheldon									
Σ	Dir	Pos		Pos	Pos	Pos	Pos		Pos
	B	0.42		0.54	0.50	0.33	0.21		0.62
	R^2	0.180		0.290	0.252	0.110	0.044		0.381
	P	0.071		0.017*	0.028	0.166	0.383		0.005*
#	Dir					Pos	Neg		
	Inc					62%	37%		
	P					0.383	0.248		
Regional deciduous riparian									
Sacramento									
Σ	Dir	Pos		Pos		Neg	Pos	Pos	

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
	B	0.16		0.17		−0.21	0.26	0.17	
	R^2	0.025		0.029		0.045	0.067	0.027	
	P	0.253		0.217		0.122	0.056	0.228	
#	Dir				Neg	Neg			
	Inc				30%	39%			
	P				0.343	0.281			
Snake									
Σ	Dir	Neg		Neg			Neg	Neg	Neg
	B	−0.47		−0.46			−0.31	−0.12	−0.41
	R^2	0.221		0.213			0.088	0.014	0.170
	P	<0.001*		0.001*			0.026	0.399	0.002*
#	Dir								
	Inc								
	P								
Bitterroot									
Σ	Dir	Pos	Pos	Pos	Pos	Pos		Pos	Pos
	B	0.26	0.27	0.48	0.32	0.41		0.39	0.25
	R^2	0.066	0.073	0.234	0.103	0.167		0.149	0.062
	P	0.005	0.006	<0.001*	<0.001*	<0.001*		<0.001*	0.006
#	Dir	Pos			Pos	Pos	Pos	Pos	Pos
	Inc	68%			71%	71%	63%	69%	65%
	P	0.001*			0.143	0.018	0.216	0.210	0.170
Missouri									
Σ	Dir	Pos	Pos		Pos	Pos	Pos	Pos	Pos
	B	0.61	0.38		0.39	0.34	0.65	0.38	0.33
	R^2	0.368	0.146		0.151	0.118	0.413	0.144	0.110
	P	<0.001*	0.041		0.037	0.068	<0.001*	0.043*	0.079
#	Dir	Pos				Pos	Pos	Pos	
	Inc	69%				78%	73%	100%	
	P	0.030				0.180	0.118	0.125	
Hart									
Σ	Dir			Neg	Neg		Pos		Neg
	B			−0.20	−0.15		0.17		−0.24
	R^2			0.039	0.023		0.040		0.057
	P			0.213	0.342		0.279		0.134
#	Dir						Pos		
	Inc						63%		
	P						0.359		

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distance migrant	Residents	Short-distance migrant
Local Coniferous Forest									
Snake									
Σ	Dir			Neg	Pos	Neg		Neg	Neg
	B			−0.21	0.45	−0.20		−0.23	−0.17
	R^2			0.044	0.202	0.042		0.051	0.030
	P			0.132	0.001*	0.143		0.104	0.217
#	Dir	Pos			Pos		Pos		
	Inc	64%			71%		71%		
	P	0.059			0.180		0.064		
Bitterroot									
Σ	Dir	Neg	Neg	Neg	Neg	Neg		Neg	Neg
	B	−0.29	−0.34	−0.38	−0.40	−0.29		−0.26	−0.29
	R^2	0.081	0.116	0.144	0.162	0.084		0.069	0.086
	P	0.002	<0.001*	<0.001*	<0.001*	0.001*		0.004	0.001*
#	Dir	Neg			Neg	Neg		Neg	Neg
	Inc	36%			31%	33%		31%	38%
	P	0.007*			0.210	0.055		0.210	0.265
Missouri									
Σ	Dir	Neg				Neg	Neg	Neg	Neg
	B	−0.31				−0.26	−0.32	−0.37	−0.18
	R^2	0.095				0.066	0.090	0.136	0.034
	P	0.103				0.180	0.094	0.049	0.341
#	Dir								
	Inc								
	P								
Hart									
Σ	Dir				Pos		Pos	Neg	Neg
	B				0.18		0.18	−0.15	−0.21
	R^2				0.032		0.032	0.024	0.044
	P				0.266		0.250	0.337	0.186
#	Dir						Pos	Neg	
	Inc						76%	17%	
	P						0.027*	0.219	
Regional Coniferous Forest									
Snake									
Σ	Dir	Neg		Neg			Neg		Neg
	B	−0.50		−0.57			−0.35		−0.44
	R^2	0.251		0.327			0.118		0.178
	P	<0.001*		<0.001*			0.010		0.002*

APPENDIX 4. CONTINUED

Landscape variable and river system	Statistic	Total birds	Richness	Cowbirds	Prime hosts	Non-hosts	Long-distauce migrant	Residents	Short-distance migrant
#	Dir								
	Inc								
	P								
Bitterroot									
Σ	Dir	Neg	Neg	Neg	Neg	Neg	Pos	Neg	Neg
	B	−0.23	−0.27	−0.38	−0.29	−0.35	0.14	−0.28	−0.29
	R^2	0.055	0.072	0.146	0.086	0.124	0.019	0.078	0.084
	P	0.010	0.006	<0.001*	0.001*	<0.001*	0.137	0.002	0.001*
#	Dir	Neg				Neg		Neg	Neg
	Inc	35%				32%		33%	32%
	P	0.004*				0.035		0.302	0.089
Missouri									
Σ	Dir	Pos	Pos	Pos	Pos	Pos	Pos		Pos
	B	0.77	0.69	0.18	0.69	0.19	0.81		0.521
	R^2	0.554		0.034	0.480	0.036	0.664		0.271
	P	<0.001*	<0.001*	0.339	<0.001*	0.325	<0.001*		0.004*
#	Dir	Pos			Pos		Pos		
	Inc	71%			69%		89%		
	P	0.015*			0.267		0.001*		
Hart									
Σ	Dir	Neg	Neg		Neg	Neg	Neg		Neg
	B	−0.25	−0.21		−0.40	−0.32	−0.21		−0.31
	R^2	0.064	0.043		0.160	0.100	0.032		0.097
	P	0.109	0.194		0.010*	0.044*	0.199		0.048*
#	Dir	Neg			Neg	Neg			Neg
	Inc	36%			25%	35%			31%
	P	0.036*			0.289	0.170			0.063

Notes: Results are from linear regression (Σ), with directionality of change (Dir: Pos = higher relative abundance in areas with more of the landscape variable; Neg = lower relative abundance in areas with more of the landscape variable), standardized regression coefficient (B), R^2, and P-value shown; and from Binomial tests across all species in the guild (#) for directionality (more or less abundant) with high amounts of each landscape variable, with directionality of the majority of species (Dir), the percent of species more abundant in areas with high amounts of the landscape variable (Inc), and the P-value for the binomial test shown. All results with a trend ($P < 0.4$) are shown. * = significant after Bonferroni adjustment for multiple tests.

APPENDIX 5. Grazing Effects on Individual Species, by River

River system	Detection/survey		Mann-Whitney U-test		
	Ungrazed	Grazed	U	W	P
Sacramento					
Less Abundant in Grazed Areas					
Tree Swallow	0.5873	0.0597	66.5	111.5	0.001
Black-headed Grosbeak	0.6412	0.2735	95.5	140.5	0.011
Downy Woodpecker	0.0960	0.0094	105.5	150.5	0.013
American Robin	0.1555	0.0409	123.0	168.0	0.044
California Towhee	0.0406	0.0000	144.0	189.0	0.060
Mourning Dove	0.1550	0.0472	127.5	172.5	0.062
Bank Swallow	0.0821	0.0000	153.0	198.0	0.089
White-breasted Nuthatch	0.0761	0.0189	146.5	191.5	0.122
Turkey Vulture	0.1250	0.0189	155.5	200.5	0.152
European Starling	0.1061	0.0094	157.0	202.0	0.156
Western Wood-pewee	0.5840	0.3741	148.0	193.0	0.179
More Abundant in Grazed Areas					
California Quail	0.0457	0.2169	78.5	1159.5	0.001
Warbling Vireo	0.0341	0.0880	105.5	1186.5	0.004
Wilson's Warbler	0.1370	0.2578	91.0	1172.0	0.007
Bewick's Wren	0.6334	0.8708	116.5	1197.5	0.039
Lazuli Bunting	0.3520	0.4999	123.5	1204.5	0.057
Lesser Goldfinch	0.1702	0.3804	142.5	1223.5	0.130
Snake					
Less Abundant in Grazed Areas					
Veery	0.4791	0.1161	118.0	328.0	0.001
Song Sparrow	0.8020	0.3124	117.0	327.0	0.001
Fox Sparrow	0.1606	0.0131	134.5	344.5	0.002
Black-capped Chickadee	0.3667	0.2178	150.5	360.5	0.015
Lazuli Bunting	0.1176	0.0678	154.0	364.0	0.016
Yellow Warbler	2.7632	2.2466	152.0	362.0	0.017
Mallard	0.0474	0.0118	174.5	384.5	0.029
Black-headed Grosbeak	0.2386	0.1465	162.5	372.5	0.030
Belted Kingfisher	0.0293	0.0091	189.0	399.0	0.041
Gray Catbird	0.1490	0.0763	172.5	382.5	0.047
Cedar Waxwing	0.3268	0.1940	172.0	382.0	0.050
Ruffed Grouse	0.0321	0.0056	203.5	413.5	0.058
Violet-green Swallow	0.2309	0.0971	179.0	389.0	0.059
Broad-tailed Hummingbird	0.0118	0.0022	213.0	423.0	0.096
MacGillivray's Warbler	0.0532	0.0149	196.0	406.0	0.107
Spotted Sandpiper	0.0302	0.0158	198.0	408.0	0.118
Swainson's Thrush	0.0403	0.0068	211.0	421.0	0.147
More Abundant in Grazed Areas					
House Wren	0.4621	1.1689	107.0	458.0	0.001
Mourning Dove	0.2488	0.5509	149.5	500.5	0.014
Pine Siskin	0.0044	0.0529	180.5	531.5	0.019
Black-billed Magpie	0.2475	0.4988	160.0	511.0	0.026
European Starling	0.3474	1.2135	163.5	514.5	0.032
Cassin's Finch	0.0201	0.0326	208.0	559.0	0.167
Missouri					
Less Abundant in Grazed Areas					
Mourning Dove	2.2941	0.7917	19.5	97.5	<0.001
American Robin	2.3824	1.0833	27.0	105.0	0.001
Red-eyed Vireo	0.7059	0.0417	45.5	123.5	0.004
Red-shafted Flicker	1.6765	0.4583	39.0	117.0	0.004
Least Flycatcher	2.1176	1.2083	43.5	121.5	0.008
Brown Thrasher	0.7059	0.1250	48.0	126.0	0.009
Western Wood-pewee	1.8824	1.0833	48.0	126.0	0.011
Lazuli Bunting	1.6765	0.9167	46.0	124.0	0.011
Ovenbird	0.4706	0.0000	60.0	138.0	0.013
House Wren	2.7647	2.0000	54.0	132.0	0.028
Black-headed Grosbeak	0.7353	0.1667	57.5	135.5	0.029
Bullock's Oriole	0.6471	0.2083	57.0	135.0	0.031

APPENDIX 5. CONTINUED

River system	Detection/survey		Mann-Whitney U-test		
	Ungrazed	Grazed	U	W	P
American Redstart	0.4706	0.0833	63.0	141.0	0.034
Yellow Warbler	3.7941	2.5833	58.0	136.0	0.047
Yellow-breasted Chat	2.8529	2.0417	59.0	137.0	0.051
Hairy Woodpecker	0.4118	0.0833	65.0	143.0	0.052
Gray Catbird	0.6176	0.1250	70.5	148.5	0.108
Common Grackle	0.6471	0.3333	76.5	154.5	0.132
Black-capped Chickadee	0.6471	0.2083	74.5	152.5	0.161
American Goldfinch	2.3529	1.5417	72.0	150.0	0.177
More Abundant in Grazed Areas					
Eastern Kingbird	0.1176	0.3333	67.0	220.0	0.048
Spotted Towhee	0.9706	1.3750	67.5	220.5	0.115
Hart					
Less Abundant in Grazed Areas					
Cordilleran Flycatcher	0.3333	0.0000	140.0	350.0	0.005
Hairy Woodpecker	0.4286	0.0500	130.5	340.5	0.005
Green-tailed Towhee	0.5714	0.1500	121.5	331.5	0.006
Rock Wren	0.2619	0.0000	170.0	380.0	0.043
Wilson's Warbler	0.2381	0.0500	170.5	380.5	0.092
Red-tailed Hawk	0.2857	0.1000	171.0	381.0	0.138
More Abundant in Grazed Areas					
Swainson's Thrush	0.1905	0.4000	160.0	391.0	0.091
Black-headed Grosbeak	0.3333	0.6500	154.0	385.0	0.094
Sheldon					
Less Abundant in Grazed Areas					
Western Wood-pewee	0.6000	0.0000	22.5	67.5	0.017
More Abundant in Grazed Areas					
Brewer's Sparrow	0.2000	0.7778	23.0	78.0	0.039
Yellow Warbler	0.3000	0.7778	27.0	82.0	0.096

Notes: Values are mean detections per survey, and results of Mann-Whitney U-test for differences between grazed and ungrazed. All species detected at least 15 times on a given river system with a $P < 0.2$ from a Mann-Whitney U-test are included.

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HOW FINE SEDIMENT IN RIVERBEDS IMPAIRS GROWTH AND SURVIVAL OF JUVENILE SALMONIDS

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Abstract. Although excessive loading of fine sediments into rivers is well known to degrade salmonid spawning habitat, its effects on rearing juveniles have been unclear. We experimentally manipulated fine bed sediment in a northern California river and examined responses of juvenile salmonids and the food webs supporting them. Increasing concentrations of deposited fine sediment decreased growth and survival of juvenile steelhead trout. These declines were associated with a shift in invertebrates toward burrowing taxa unavailable as prey and with increased steelhead activity and injury at higher levels of fine sediment. The linear relationship between deposited fine sediment and juvenile steelhead growth suggests that there is no threshold below which exacerbation of fine-sediment delivery and storage in gravel bedded rivers will be harmless, but also that any reduction could produce immediate benefits for salmonid restoration.

Key words: fine sediment; Oncorhynchus mykiss; *Pacific salmonids; parr; river food web; sedimentation; steelhead trout.*

Introduction

Throughout western North America, historically large populations of native anadromous salmonids are in severe decline or extinct. In the United States alone, 26 Evolutionarily Significant Units of Pacific salmonid are currently threatened or endangered (National Marine Fisheries Service 2003). These declines are in large part attributable to degradation of spawning and rearing habitat (Nehlsen et al. 1991, Frissell 1993), a major cause of which is increased loading and storage of fine sediments (Miller et al. 1989, Bisson et al. 1992, Waters 1995).

The storage of fine sediments (particle sizes <2 mm median diameter) in gravel-bedded rivers is normally a transient phenomenon, as sediments enter and leave river channels naturally. Without frequent resupply from upstream sources or termination of gravel mobilizing flows, fine sediment is carried downstream to lowland reaches or the sea. Yet anthropogenic activities have greatly increased the storage of fine sediment in rivers throughout the world. Where it comes to rest in river reaches, fine sediment can transform the topography and porosity of the gravel riverbed in ways that profoundly affect the emergent ecosystem, particularly during biologically active periods of seasonal low flow. It is during these periods of low flow that demographically critical juvenile rearing occurs for salmonids.

Despite scientific, political, and commercial motivation to quantify the relationship between fine-sediment loading and juvenile salmon production in river systems (and in particular, thresholds beyond which impairment occurs), no causal relationship has been established. Research on the influence of deposited fine sediment on juvenile salmonids has consisted primarily of laboratory work and correlative field studies comparing salmonid assemblages before and after or upstream and downstream of a fine-sediment influx, or among rivers with differing bed compositions. This work has suggested that fine-sediment deposition negatively impacts juvenile salmonids and the food webs supporting them (Crouse et al. 1981, Murphy and Hall 1981, Reeves et al. 1993), but field experimental support has been lacking. As a result, mechanisms by which these effects arise are also poorly understood. This is due in part to difficulty in isolating the impacts of fine sediment from other co-varying physical factors (e.g., flow velocity and turbulence, channel depth, plan form morphology) that can also influence salmonid performance.

Here we provide the results of an experiment designed to isolate the impact of fine sediment on a juvenile salmonid (*Oncorhynchus mykiss*) in its natural habitat. We manipulated fine bed sediment in replicate

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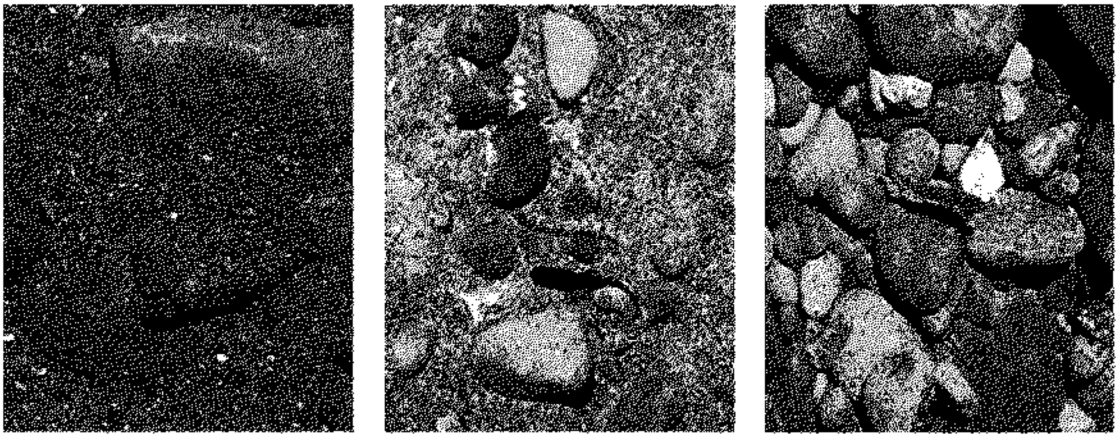

FIG. 1. Juvenile steelhead over 100%, 80%, and 0% embedded substrates (left to right).

channels in the South Fork Eel River, California (39°43′45″ N, 123°38′40″ W) and measured the growth and behavior of steelhead parr and the density and composition of aquatic invertebrate assemblages on which they prey.

METHODS

In June 2000, we constructed six 2 m long × 1 m wide channels in the downstream ends of each of four river pools distributed over a 3-km reach. The 24 channels consisted of vinyl flooring secured flush to the riverbed with rebar stakes and metal pipe, with walls protruding 15 cm above the water surface. Each channel was assigned to one of six fine-sediment loadings in a complete randomized block design. We filled each channel to a depth of 15 cm with pebbles, gravels, and cobbles collected from the adjacent riverbed and sifted through a 6-mm sieve. The size range of these coarse framework particles was 6–90 mm diameter. Median size class by weight was 22–32 mm diameter. The highest sediment loading treatment ("100% embeddedness") received enough fine sediment so that only the upper surfaces of the coarse framework particles were visible. At this level, further additions of fine sediment would not alter the topography or porosity of the bed in a biologically meaningful manner. The other five treatments at each site received 80, 60, 40, 20, and 0% of that volume (Fig. 1). Fine sediment consisted of particles with diameter <2 mm. Median size class by weight was 0.60–1.18 mm diameter.

After allowing 25 d for invertebrate colonization and algal growth, we closed the upstream and downstream ends of each channel with 6-mm mesh walls that were permeable to invertebrate drift and smaller prey fishes but did not allow passage of juvenile steelhead. We seined steelhead parr from the river and measured and weighed each. We then stocked each channel with two parr (1 parr/m^2), approximating densities in the adjacent open habitat. At stocking, standard length ranged from 37 to 54 mm and mass from 0.65 to 2.53 g. The two fish in each channel were chosen to differ in length by 5–9 mm. Fish were confined for 46 d, during which time we conducted extensive behavioral observations. Observers approached the channels, remained motionless for 5 min, and then began 10-min continuous observation periods, in which swimming was distinguished from holding and sheltering behavior and all feeding movements and intraspecific interactions were recorded. A minimum of four such observations per experimental channel were conducted during the time fish were confined.

At experiment's end, all steelhead parr were measured, weighed, and released. Any fish that died over the course of the experiment were immediately replaced. Injury was inferred as the cause of death when a fish died after developing fin rot. We observed these infections on fish wounded during conspecific attacks. The infection whitened dorsal or caudal fins, spread from the fin into the body tissue, and resulted in death within 2 d. Analyses of growth included only steelhead that were enclosed in experimental channels for a minimum of 25 d. All growth measurements are reported on a per day basis.

We sampled the invertebrate community in each channel from sediment cores. Just prior to stocking steelhead in the channels and on the day after steelhead were removed, three circular cores (14 cm diameter) were taken from each channel and pooled. The sediment was elutriated to dislodge invertebrates, and all organisms were collected and stored in 70% ETOH. All organisms were classified to family and assigned to one of three broad functional groups (i.e., burrowing, armored, and vulnerable) based on life history traits influencing availability to steelhead fry. This information was gleaned from published reports (Merritt and Cummins 1996, Resh et al. 1997) and from direct field observation. Burrowing taxa included oligochaete worms, freshwater clams (Sphaeriidae), one genus of silt-encased Trichoptera (Sericostomatidae: Gumaga), one family of Megaloptera (Sialidae), two families of

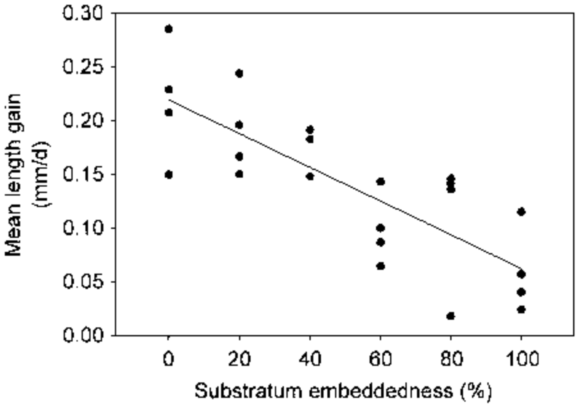

FIG. 2. Growth of juvenile steelhead trout in relation to direct manipulation of substratum embeddedness ($R^2 = 0.63$, $P < 0.0001$). Changes in growth in mass were similar ($R^2 = 0.59$, $P < 0.0001$). Analyses of relative growth, which accounts for differences in initial size, and of instantaneous growth rate produced similar linear patterns ($R^2 = 0.52$, $P = 0.0001$ and $R^2 = 0.53$, $P < 0.001$, respectively). One experimental channel with 40% substratum embeddedness contained no fish that survived the minimum 25 d and is thus excluded from the analysis.

Diptera (Ceratopogonidae and Tipulidae), and one family of Odonata (Gomphidae). Armored taxa included two families of snails (Planorbidae and Physidae), two families of stone-encased Trichoptera (Helicopsychidae and Limnephilidae), and wood-encased Limnephilidae over 10 mm in length. Vulnerable prey included three families of wood-encased Trichoptera under 10 mm in length (Lepidostomidae, Brachycentridae, and Limnephilidae), one family of free-living Trichoptera (Rhyacophilidae), four families of Ephemeroptera (Heptageniidae, Baetidae, Trichorythidae, and Leptophlebiidae), two families of Plecoptera (Perlidae and Chloroperlidae), three families of Coleoptera (Elmidae, Haliplidae, and Psephenidae), two families of Diptera (Blephariceridae and Chironomidae), and three families of Odonata (Aeshnidae, Lestidae, and Coenagrionidae). Individual insect dry biomass was determined based on length regressions published in the literature or generated in this study. Biomass of burrowing organisms was log transformed to meet assumptions of regression analysis.

Results

Steelhead growth decreased steeply and roughly linearly with increasing fine-sediment concentration (Fig. 2). This result was consistent with the effects of sedimentation on the food supply available to steelhead. With increasing fine sediment, invertebrate assemblages shifted from available prey organisms (i.e., epibenthic grazers and predators) to unavailable burrowing taxa (Fig. 3), so that steelhead confined to channels with higher levels of sedimentation experienced lower food availability than those in less embedded channels.

In addition to reducing prey availability, deposited fine sediments increased steelhead activity. At higher levels of embeddedness, fine sediments filled spaces under and between coarse cobbles, producing a flat and featureless bed. As interstitial refuges and prey declined, steelhead spent less time sheltering behind or under cobbles and more time actively swimming (Fig. 4a). Steelhead also exhibited higher levels of intraspecific aggression, including attacks (Fig. 4b), as prey availability and visual separation between fish decreased with higher fine-sediment levels. This likely explains the increased incidence of at least one mortality event in more heavily embedded channels (logistic regression, $P < 0.05$, $n = 24$; Fig. 5).

Discussion

Anadromous salmonids have a complex life history that exposes them to a wide range of threats across multiple life stages. As a commercially, culturally, and ecologically valuable group of animals, considerable research effort has been devoted to quantifying the impacts of these various threats (e.g., dams, fish farms, overharvesting, hatchery fish, invasive organisms, and river and estuarine pollution, degradation, and habitat loss) on wild salmon populations. It is widely known that salmonid stocks decline when land use increases fine-sediment delivery to gravel-bedded rivers (Bisson and Sedell 1984, Reeves et al. 1993, Waters 1995), but mechanistic understanding of the role of fine sediment

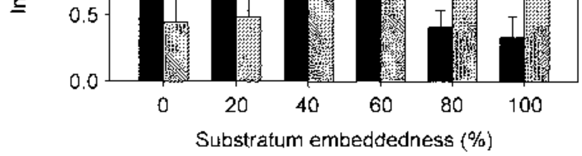

FIG. 3. Biomass of invertebrates from sediment core samples taken at the experiment's end (mean + 1 SE). There were significant linear relationships between fine sediment and the biomass of individual functional groups of invertebrates. As fine sediment increased (greater embeddedness), biomass of vulnerable prey declined ($R^2 = 0.42$, $P < 0.001$) and biomass of unavailable burrowing organisms increased ($R^2 = 0.23$, $P = 0.02$). A similar pattern was found in the prestocking samples taken on 30 June; there was a significant and negative relationship between fine sediment and vulnerable prey biomass ($R^2 = 0.35$, $P = 0.003$) and a significant and positive relationship between fine sediment and burrowing organism biomass ($R^2 = 0.37$, $P = 0.002$). Fine sediment had no influence on the biomass of armored grazers. Similar taxon-specific responses to fine sediment have been observed in other studies (Bjornn et al. 1977, Mebane 2001).

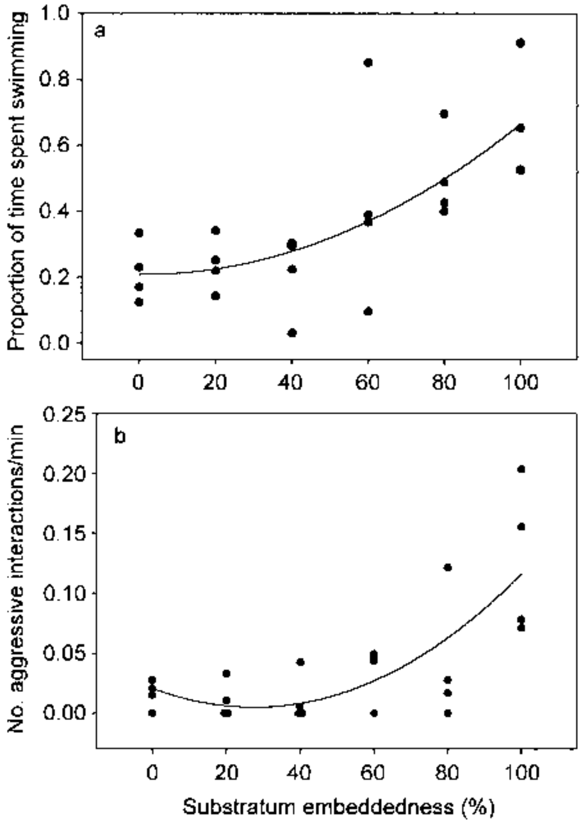

FIG. 4. Behavior of steelhead parr in experimental channels. Data represent mean values for each experimental channel. (a) Fish activity (swimming time) is represented by the best-fit line from a second-order polynomial regression ($R^2 = 0.45$, $P = 0.004$). The difference in activity between steelhead in 100% embeddedness channels and those in 0% embeddedness channels translates to a 47% higher energy expenditure, based on metabolic data for the same size class of sockeye salmon (*O. nerka*) under similar environmental conditions (Brett and Glass 1973), energy equivalents of animal oxygen consumption (Elliot and Davison 1975), and assuming a standard 12-h period of nightly inactivity. (b) Intraspecific aggression is represented by the best-fit line from a second-order polynomial regression ($R^2 = 0.56$, $P = 0.0002$).

in these declines has been restricted to the embryo stage.

When deposited in riverbeds, fine sediment can reduce survival of embryos and emergence of fry from redds (nests in the riverbed) by decreasing dissolved oxygen and water exchange and entrapping emerging fry (Chapman 1988). Survival of embryos may not, however, limit salmonid populations. Even where sediment influxes destroy many redds, higher survival rates from redds in suitable substrate in heterogeneous reaches could compensate for these losses (Magee et al. 1996). Fry and parr from successful redds must then contend with changes in rearing habitat imposed by fine sediment. Even fry hatched from redds in unimpacted tributaries and side channels are susceptible, as they ultimately rear in larger channels (Reiser and Bjornn 1979, Hackelroad and La Marr 1993), where larger drainage areas and lower gradients increase the likelihood of fine-sediment loading and storage. Steelhead trout may be particularly vulnerable, as they remain in natal streams up to two years longer than other anadromous salmonids. By confining juvenile steelhead over discreet patches of riverbed with experimentally imposed fine-sediment concentrations, we were able to investigate the mechanisms underlying previously observed patterns of salmonid declines in response to fine-sediment loading and storage.

The decreases in steelhead growth and survival we observed with increasing fine-sediment deposition were associated with lower prey availability and higher activity, aggression, and risk of injury. Declines in growth rates lower survival of salmonids and other fishes (Werner and Gilliam 1984, Walters and Korman 1999). Larger body size confers higher survival of over-wintering (Quinn and Peterson 1996) and smolting (Ward and Slaney 1988, Yamamoto et al. 1999) juvenile salmonids. Recent demographic models indicate that these juveniles may be the best age classes to target for effective conservation measures. Even modest reductions in juvenile mortality (i.e., 6–11%) are predicted to reverse population declines in Snake River chinook salmon (*Oncorhynchus tshawytscha*), regardless of adult dam passage success and egg survival (Kareiva et al. 2000). Differences in growth and survival imposed by fine sediment could therefore have important population-level impacts.

The flux of fine sediment into and out of river systems, while a natural process, has been greatly exacerbated by humans. Land uses that increase erosion, particularly road construction (Burns 1972, Megahan and Kidd 1972, Beschta 1978, Reid et al. 1981), increase fine-sediment loading, while flow regulation and diversion diminish transport and removal. The steep dissected terrain and weak parent material of drainages along California's North Coast make rivers of this region particularly vulnerable to land-management prac-

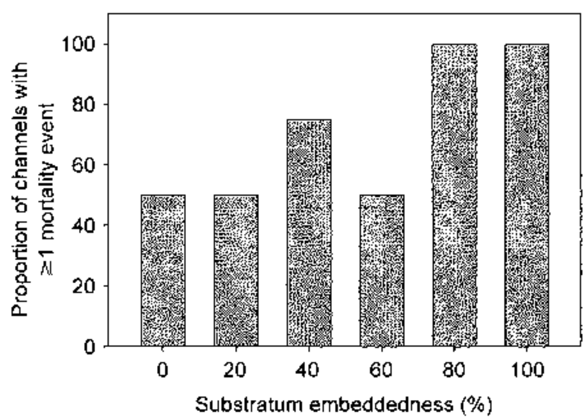

FIG. 5. Steelhead mortality in experimental channels, in relation to fine sediment.

tices that increase erosion. Increased storage of fines dramatically alters river ecosystems. This state is transient over geomorphic time scales, as fine-sediment pulses move down through steeper, gravel-bedded portions of drainage networks and eventually are discharged into lowland floodplains, estuaries, and the sea. In addition, certain steep microhabitats may not retain fine sediments even in rivers receiving heavy loading. In this sense, the river is potentially self-cleaning. If land uses that increase loading or decrease transport of fine sediments continue unabated, however, areas of formerly suitable juvenile rearing habitat may be lost from the riverbed long enough to cause irreversible population declines in resident salmonids. This concern is particularly important for juvenile salmonids, whose territoriality limits their ability to crowd into shrinking areas of good habitat.

Many states and countries throughout the world have regulations directed at fine-sediment management, intended in part to protect native and introduced salmon. None of these regulations derives from known quantitative relationships between the amount of loaded or stored fine sediment and the performance of salmonids in the receiving river. In particular, it is not known whether there might be an acceptable level of increase in fine-sediment loading that causes no damage to salmonids or the ecosystems supporting them. Nearly all sediment management regulations, however, make this reasonable assumption (U.S. EPA 1999).

Our experiment demonstrates that fine-sediment deposition, even at low concentrations, can decrease growth and survival of juvenile salmonids. We find no threshold below which fine-sediment addition is harmless. These results suggest that any augmentation of fine-sediment deposition in steelhead bearing rivers in this region will further impair this potentially population-limiting life stage, while land management practices that decrease fine-sediment loading or storage in channels may benefit salmonid populations.

Acknowledgments

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CFER Cooperative Forest Ecosystem Research

Influence of Forest Management on Headwater Stream Amphibians at Multiple Spatial Scales

Background

Amphibians are important components of headwater streams in forest ecosystems of the Pacific Northwest (PNW). They comprise the highest vertebrate biomass and density in these systems and are integral to trophic dynamics both as prey and as predators. The most commonly encountered amphibians in PNW headwater streams include the Pacific giant salamander (*Dicamptodon tenebrosus*), the tailed frog (*Ascaphus truei*), the southern torrent salamander (*Rhyacotriton variegatus*), and the Columbia torrent salamander (*R. kezeri*).

Several studies of headwater stream amphibians have examined species-habitat associations in managed and unmanaged forests. Results from some of these studies suggest that logging practices at the stand scale may impact species presence and abundance by directly or indirectly altering stream and riparian habitat. Habitat associations also have been well studied at the stream-reach scale; however, the influence of broader spatial-scale patterns (such as landscape structure) on amphibians is unclear. Because management activities at broad scales can influence habitat at finer scales, identifying the effects of these activities on headwater amphibians at different spatial scales is fundamental to the development of appropriate riparian management practices.

CFER scientists Margo Stoddard and John Hayes investigated the relationships between headwater stream amphibians (Pacific giant salamanders, tailed frog adults, tailed frog tadpoles, and torrent salamanders) and habitat characteristics measured at four spatial scales (2-m sample unit, patch, sub-drainage, and drainage; Figure 1). The goals of the study were to: 1) identify and rank the importance of habitat characteristics in predicting amphibian occurrence at fine spatial scales; 2) identify and rank the importance of geophysical and management-related characteristics in predicting amphibian occurrence at three broader spatial scales; 3) examine patterns across scales; and 4) evaluate habitat models at each spatial scale that could be used to develop riparian and upslope management strategies that maintain adequate habitat for stream amphibians.

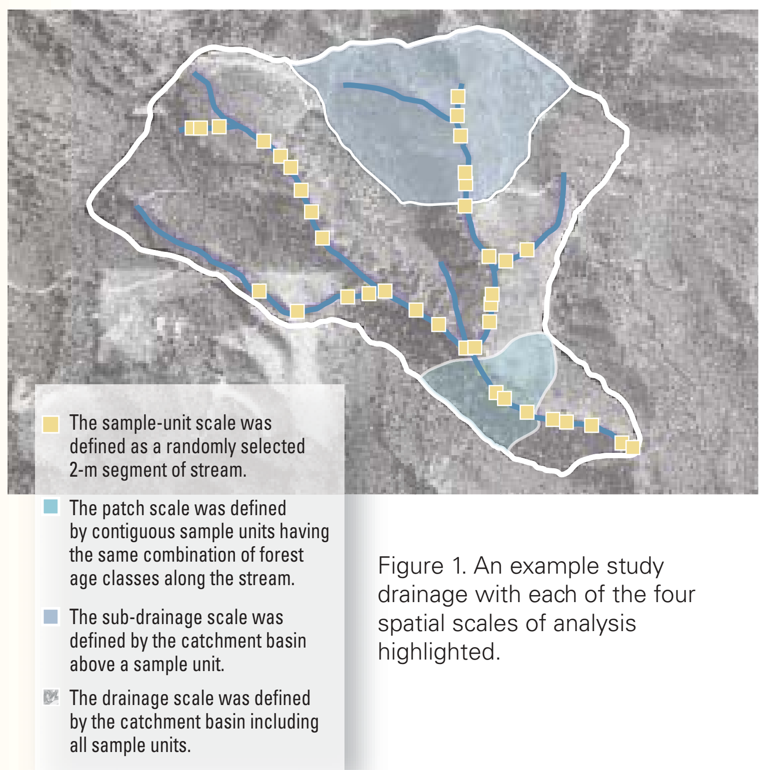


Figure 1. An example study drainage with each of the four spatial scales of analysis highlighted.

In 1998, a population of potential study sites was identified from maps of land ownership, streams, and forest-age classes. Potential study sites included all third-order drainages in the Eugene and Salem Districts of the Bureau of Land Management (BLM) on the east slope of the Oregon Coast Range. To assure a range of management conditions was represented, drainages were stratified into low, moderate, and high management intensities based on the percentage of forest >55 years old in each drainage (Figure 2). Drainages in each management intensity category were then randomly selected from a list of all potential sites for a total of sixteen drainages (five intensively logged drainages, five moderately logged drainages, and six drainages subjected to low-logging intensity; Figure 3).

Figure 2. Digital orthophotographs of three of the study drainages with A) low, B) moderate, and C) high management intensities based on the percentage of forest >55 years old. Light areas on photos represent clearcut and young forest stands.

Amphibian sampling and microhabitat classification were conducted in 702 2-meter units. Each unit was carefully searched for amphibians by overturning stream substrates. Instream and streambank habitat characteristics representing geomorphic, vegetative, topographic, and physical characteristics also were measured. Macrohabitat classification was based on characteristics measured at the patch, sub-drainage, and drainage spatial scales. Variables examined at the patch scale included stand age(s) around the stream, presence of a forested band >55 years old and at least 150 ft (46 m) in width on each side of the stream, aspect, and stream gradient. Variables at the sub-drainage and drainage scales included the proportion of young (<15 years old) forest, road density, percent side-slope, aspect, and proportion of stream length bordered by forested bands >55 years old and >150 ft in width.

At each spatial scale, species-habitat association models were developed, and the importance of each habitat variable in determining species occurrence was determined. Models were ranked using Akaike's Information Criterion (AIC), and variable importance was assessed with Akaike weights (Burnham and Anderson 2002, *Model Selection and Multi-Model Inference: a Practical Information-Theoretic Approach*. Springer-Verlag. 488 pp.).

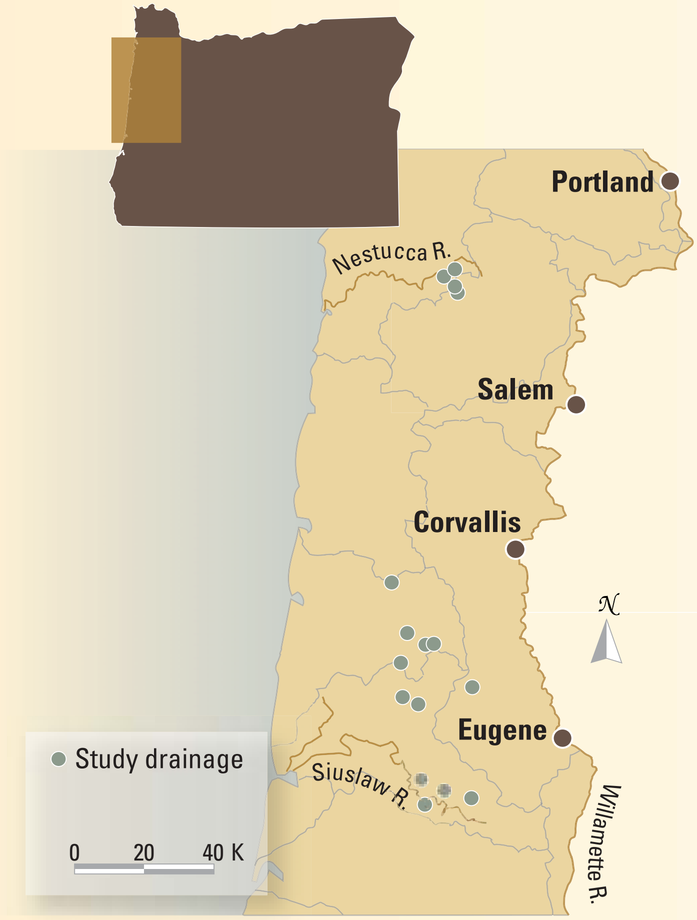


Figure 3. Location of the sixteen study drainages selected for study. Drainages were randomly selected and represent a range of management intensities.

Results

At the finest spatial scale, amphibians were most frequently found in stream segments with greater proportions of large substrates (>3.2 cm in diameter) in the streambed. At the patch and sub-drainage scales, variables related to the geophysical characteristics of streams or drainages (e.g., gradient and aspect) were important for all species or life stages except tailed frog adults. In general, at broad spatial scales, variables related to forest condition (e.g., the presence of a forested band >150 ft in width on each side of the stream or the percentage of stream length with forested bands >150 ft in width) around streams were important. Table 1 provides a summary of the variables most important in predicting occurrence at each spatial scale for each species or life stage.

Table 1. Blue boxes highlight the variables most important in predicting amphibian occurrence at all spatial scales. All relationships between characteristics and amphibian occurrence are positive.

		Species			
Spatial scale	**Habitat characteristic associated with high probability of occurrence**	**Tailed frog tadpole**	**Pacific giant salamander**	**Tailed frog adult**	**Torrent salamander**
Sample unit	Large amount of large substrate	■	■	■	■
	Large stream width	■	■		
	High elevation			■	■
	High % pool	■			
Patch	>150-ft forested band[a]		■	■	■
	Southwesterly aspect	■	■		
	Old forest on ≥1 side of stream[b]	■			
	Low stream gradient		■		
	High stream gradient				■
Sub-drainage	>150-ft forested band[a]	■	■	■	■
	Northeasterly aspect	■	■		
	High % old forest in sub-drainage			■	■
	Large area with slope <60%	■	■		
	Large sub-drainage area	■			
	Small sub-drainage area				■
	Low stream gradient		■		
Drainage	>150-ft forested band[a]	■	■	■	
	No characteristics[c]				■

[a] "Forested band" variables represent the presence of (patch scale) or high % of stream length (sub-drainage and drainage scales) with a band of >55 year-old forest >150 ft in width on each side of the stream.

[b] Represents presence of old forest on ≥1 side of stream.

[c] A null model including only an intercept term was ranked as the best model for predicting torrent salamander occurrence at the drainage scale.

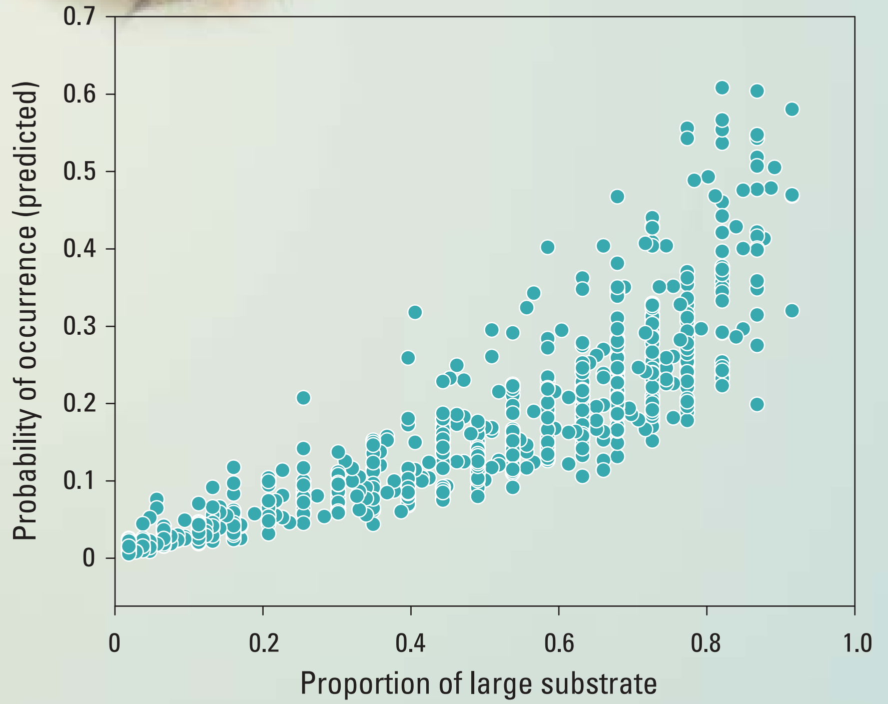

Suzanne L. Collins, CNAH

Because forested riparian habitat was a key characteristic for predicting occurrence of stream amphibians at broader spatial scales, the influence of forest band width was examined in greater depth. The researchers found that the relative odds of finding amphibians in streams generally increased with band width (Figure 4). For example, odds of finding a tailed frog tadpole in a stream surrounded by forested bands >150 ft wide were approximately 5 times greater than in a stream surrounded by forested habitat <50 ft wide (odds ratio = 0.205). The response of Pacific giant salamanders was similar to that of tailed frog tadpoles. For tailed frog adults and torrent salamanders, only a small number of observations were recorded, and this may have obscured biologically important differences among band widths for these taxa.

Case Study: Habitat Associations for Tailed Frog Tadpoles at Multiple Spatial Scales

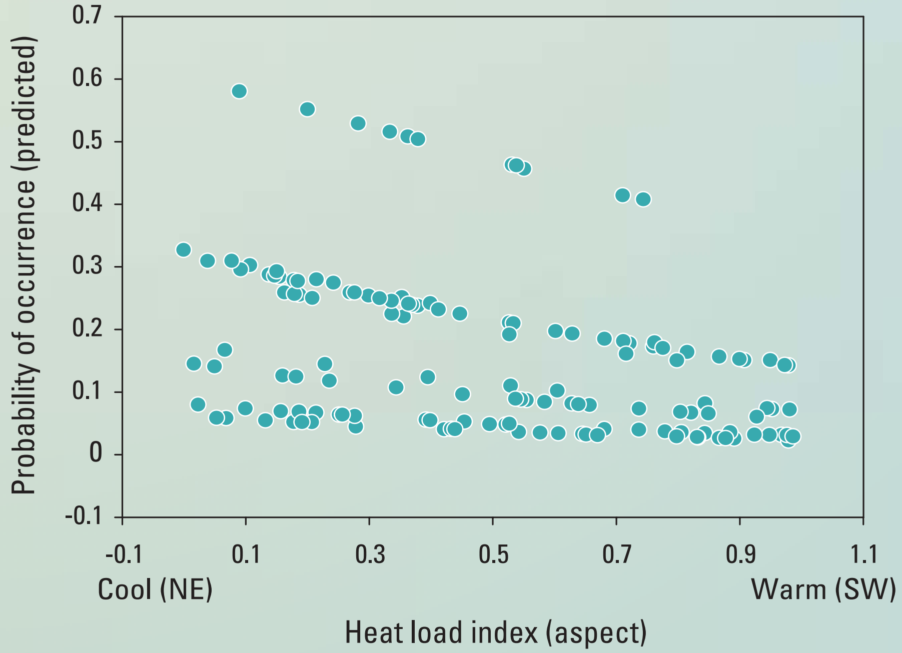
Sample-unit Scale

The top graph shows the probability of finding a tailed frog tadpole relative to a change in the amount of large substrate in a sample unit. As the proportion of large (pebble, cobble, boulder) substrate increased from 0.2 to 0.5, the predicted probability of occurrence increased from 0.06 to 0.14 and increased to 0.30 when the proportion of large substrates increased from 0.50 to 0.80. The graph was based on a best approximating model, which included stream width and % pool habitat.



Patch Scale

The bottom graph shows the probability of finding a tailed frog tadpole relative to a change in heat load index (aspect) in a patch. The odds of finding a tailed frog tadpole were 61% lower in a southwesterly facing stream than in a northeasterly facing stream. The graph was based on a best approximating model, which included a categorical variable representing the combination of stand ages (i.e., patch) on each side of the stream.



Sub-drainage and Drainage Scales

The proportion of stream length with forested bands >150 ft (46 m) wide was one of the most important variables in habitat models for tailed frog tadpoles (No graph provided).

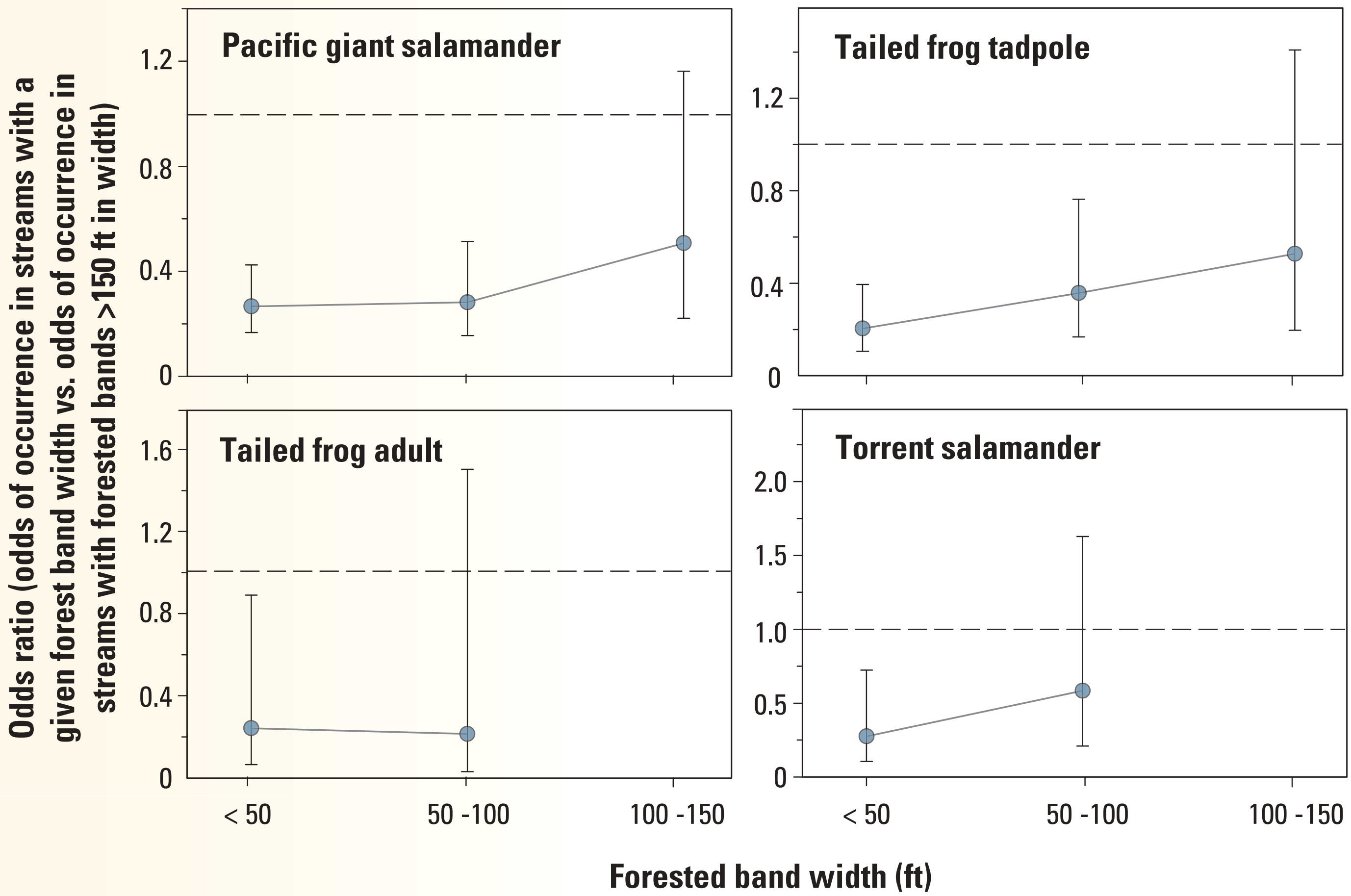


Figure 4. Relationships between forested band width and odds ratios for stream amphibians. Points represent ratio of odds of finding each species in a stream (patch scale) bordered by forested bands of various widths compared to odds of occurrence in streams with forested bands >150 ft in width (± 95% confidence intervals). The dashed line (odds ratio = 1) represents the point at which odds of occurrence do not differ. Odds ratios for torrent salamanders and tailed frog adults were calculated using data from only two band width age classes because these species did not occur in stands having 100-150 ft forested bands.

Management Implications

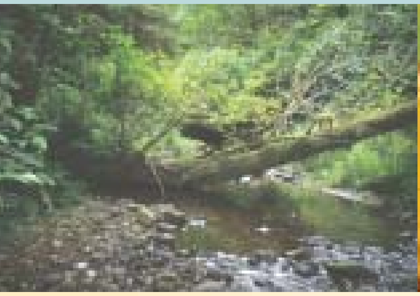
At all spatial scales, the combined influences of habitat structure and geophysical location were important in determining amphibian occurrence. In most cases, the importance of these variables could be related to known life-history requirements including cover, lack of sedimentation, cool temperatures, and habitat for foraging, movement, or dispersal. Because disturbance of riparian and upslope habitat at broad scales may affect amphibian habitat at finer scales by influencing stream temperature, microclimate, and sediment input, these life-history requirements and activities that affect these requirements should be considered when maintaining amphibian habitat is a management goal. For example, activities that increase sedimentation should be minimized adjacent to small, high elevation streams where tailed frog adults may congregate to breed and where torrent salamanders are likely to occur. One approach that has been suggested to achieve this is to retain blocks of land around small headwaters. Some consideration should be given to providing corridors in which metamorphosed amphibians, such as juvenile and tailed frog adults, may forage or disperse. Conservation priority for Pacific giant salamanders and tailed frog tadpoles should be given to maintaining forested habitat along streams in which these taxa are likely to occur (relatively wide headwater streams with northeasterly aspects). Results from this study suggest forested bands at least 150 ft in width will help maintain populations of some species. However, further research on the influence of forest band width is needed, particularly for tailed frog adults and torrent salamanders.

This study provides new insights into linkages between amphibian responses across spatial scales. Results also demonstrate that landscape-scale variables (e.g., the presence of forested bands or the percentage of forested stream length) can be used to assess management approaches for stream amphibian communities. These findings will facilitate determination of conservation-emphasis areas for species protection or less sensitive sites for forest resource management.

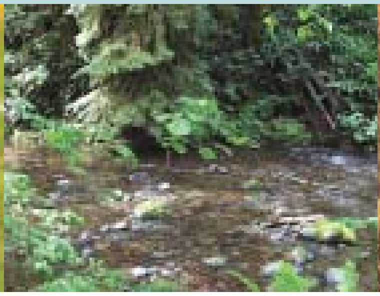
KEY RESULTS

- Large substrate in the streambed is a key habitat component for stream amphibians at fine spatial scales.
- Occurrence of stream amphibians was high in streams with northeasterly facing aspects, suggesting this variable can be useful in determining areas of conservation emphasis.
- The presence of forested habitat adjacent to streams and the amount of forested stream length in drainages were important in predicting occurrence of stream amphibians at broad spatial scales.
- Landscape-scale variables can be used to assess management approaches and habitat suitability for stream amphibians.

This factsheet is one in a series of information products developed by the Cooperative Forest Ecosystem Research (CFER) program on the influence of landscape pattern and composition on species in forested ecosystems of western Oregon. Funding for this research was provided to the CFER program by the Bureau of Land Management, USGS Forest and Rangeland Ecosystem Science Center, the Oregon Department of Forestry, and Oregon State University (OSU). Additional funding was provided by the OSU Forest Research Laboratory Fish and Wildlife Habitat in Managed Forests Program.



Suzanne L. Collins, CNAH



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Margo Stoddard received her Master of Science degree from Oregon State University in June of 2001. During her time with CFER, Margo served as a research assistant in wildlife ecology and was involved in research examining multi- and cross-scale relationships between stream amphibian occurrence and habitat structure in managed landscapes.

Dr. John P. Hayes is program coordinator and a wildlife ecologist for the CFER program. He also is a professor in the Department of Forest Science at Oregon State University. His research interests include the influence of forest management on wildlife populations, the influence of spatial scale on habitat selection, and the ecology and management of bats.

For Further Reading

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Off-Road to Ruin

How Motorized Recreation is Unraveling
California's Landscapes

The California Wilderness Coalition defends the pristine landscapes that make California unique, provide a home to our wildlife, and preserve a place for spiritual renewal. We protect wilderness for its own sake, for ourselves, and for generations yet to come. We identify and protect the habitat necessary for the long-term survival of California's plants and animals. Since 1976, through advocacy and public education, we have enlisted the support of citizens and policy-makers in our efforts to preserve California's wildlands.

For more information on the California Wilderness Coalition, visit our website at www.calwild.org.

Above: Fordyce Trail, Tahoe National Forest. Photo by Jim Rose.

Cover photo:
All-terrain vehicles, dirt bikes, and other off-road vehicles trespass into the North Algodones Wilderness on a regular basis, six years after the area was officially closed to motorized vehicles. Photo by Jim Rose.

Back cover:
Top: All-terrain vehicles at N. Algodones off-road vehicle area. Photo by Jim Rose.
Bottom left: Fordyce Trail. Photo by Jim Rose.
Bottom right: Jawbone Canyon. Photo by Howard Wilshire.

♻ Printed on recycled paper with soy-based inks.

Off-Road to Ruin

How Motorized Recreation is Unraveling California's Landscapes

March, 2001

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by Teri Shore

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- Public Employees for Environmental Responsibility
- Nordic Voice
- Center for Sierra Nevada Conservation
- Sierra Club Yahi Group
- National Audubon Society Tulare Chapter
- Sierra Club California
- Friends of the Earth
- American Lands Alliance
- Center for Biological Diversity
- Friends of Hope Valley

Additional copies of this report available for $15 each from the California Wilderness Coalition, or visit www.calwild.org to download report for free.

Table of Contents

Executive Summary

California contains a dazzling array of natural wonders. From snow-capped peaks to majestic forests, rocky shore lines to fragile desert lands, the Golden State is one of the most diverse and beautiful areas in the world. It is also among the most traveled.

Each year, millions of tourists visit California to enjoy its scenic treasures. They come expecting peace and quiet in the backcountry. Many leave, having never escaped the din of motors. California residents seeking scenery and quiet often have similar experiences—our wild places have been overrun.

Motorized vehicles are a fact of life in California. But a special class of these vehicles—called off-road vehicles—leave a unique mark on the landscape. These vehicles, which include dirt bikes, aggressive all-terrain vehicles, 4x4s, and snowmobiles, are made to be ridden in the backcountry. Their tracks are visible for generations, and their impacts often permanent.

Hundreds of thousands of these vehicles travel California's backcountry each year. When irresponsibly used, or improperly managed, they cause damage to sensitive soils, degrade critical wildlife habitat, trespass onto private property and closed areas, and shatter the quiet of the great outdoors.

The Algodones Dunes have become unsafe for family recreation activity due to the use of drugs and alcohol, and the problems of lawlessness that occur with such use.

Jim Rose

In southern California's Algodones Dunes, lawlessness prevails, sometimes bringing near-riot conditions. A recent BLM report stated, "The problem has advanced to the stage that the normal, law-abiding citizens are fearful of using the area."

Irresponsible off-road vehicle use also poses special problems for law enforcement. Some off-road vehicle riders disregard environmental laws and property boundaries. Some riders trespass onto private property, causing scars on the land and irritating landowners. These irresponsible riders are extremely difficult to catch.

The case studies contained within this report describe how California's majestic wonders continue to be tarnished by irresponsible off-road vehicle use. They also describe how California's residents are literally being "run out" by off-road vehicles. These accounts are by no means unique. They are representative of a much broader array of problems that exist throughout California's backcountry.

Examples of problems caused by off-road vehicles in California are enumerated below.

Algodones Dunes

Considered California's off-roading mecca, the Algodones Dunes suffer from out-of-control off-road vehicle use that has turned dangerous.

Hundreds of thousands of off-roaders visit the Algodones Dunes each year, creating a land management and public safety fiasco. On crowded holiday weekends, lawlessness escalates to the point where near-riot conditions prevail.

The dunes have become unsafe for family recreation activity due to the use of drugs and alcohol, and the problems of lawlessness that occur with such use, according to a Bureau of Land Management report. BLM rangers have been threatened and deliberately run over.

"The problem has advanced to the stage that the normal, law-abiding citizens are fearful of using the area," said a recent BLM report.

Lake Tahoe Basin

Illegal off-road riding is rampant in the Lake Tahoe Basin, and is contributing to the degradation of a national treasure.

In recent years, land managers in the basin have reported:

• increased complaints about riding close to or inside residential areas.

• trespass into closed areas and wilderness.

• destruction of wet meadows and other highly sensitive areas.

• protected areas that are being used as mini-moto-cross parks.

• new trails being cut into well-healed, rehabilitated trails and roads.

According to the Forest Service, off-road vehicle users have created new, unauthorized routes, creating erosion and sedimentation that may further diminish the clarity of California's most famous lake.

Jim Rose

Decades of unchecked off-road vehicle use has led to severe erosion at Dove Springs. Fragile soils that took thousands of years to develop are being quickly washed away.

Jawbone and Dove Springs

In the Jawbone and Dove Springs open riding areas, decades of overuse have entirely cleared topsoil from some areas, leaving behind bare bedrock.

Intense vehicle use at Jawbone Canyon and Dove Springs off-road vehicle areas near the town of Mojave have stripped all but the largest shrubs from the hillsides and carved deep ruts into the landscape. More than 1,000 acres have been heavily damaged and another 500 completely denuded.

The use is so heavy that, according to one report, after a rainfall, "the water formed a thick slurry of the loose soil, which flowed out over the canyon floor much like a lava flow, burying plants and trapping burrowing animals." Another study found that some upper slopes lost as much as one foot of surface soil over 13 years due to motorcycle use. Joshua trees have been uprooted by wind as soil washed rapidly away from their base.

Wildlife has largely disappeared as well: desert tortoises have been driven from the area, while the kangaroo rat and pocket mouse populations have almost entirely vanished.

Howard Wilshire

Howard Wilshire

Today, desert tortoises are barely surviving. Their home, impacted by thousands of miles of roads, has been carved into remnants of its original size. Now off-road vehicles race across the land, crushing tortoises and their burrows.

Knoxville

At the Knoxville Recreation Area, near Clear Lake, federal officials have abandoned management and law enforcement responsibilities, opening the door to widespread abuse and lawlessness.

For more than 15 years, the remote chaparral hillsides and serpentine barrens of Knoxville Recreation Area have been subjected to mostly unregulated motorcycle riding, four-wheeling, and random target shooting.

Due to its remoteness and lack of policing, Knoxville has grown practically lawless, and even dangerous at times. Appliances, TVs, computers and other large items have been found blown up and riddled with bullet holes. Long-time landowner Jim Erasmy warns his wife and women in general not to hike the area alone.

"Families come to the Knoxville area thinking that they have a safe place for their children to learn to ride. Instead they've entered an area with no rules, no road signs, and no real BLM presence....The risk of getting shot while riding off-road vehicles, hiking, horseback riding,...is a real and ever present danger," Erasmy says.

Susan Harrison

Turning hillsides to mud: uncontrolled dirt bike use has devastated portions of the Knoxville Recreation Area.

San Bernardino National Forest

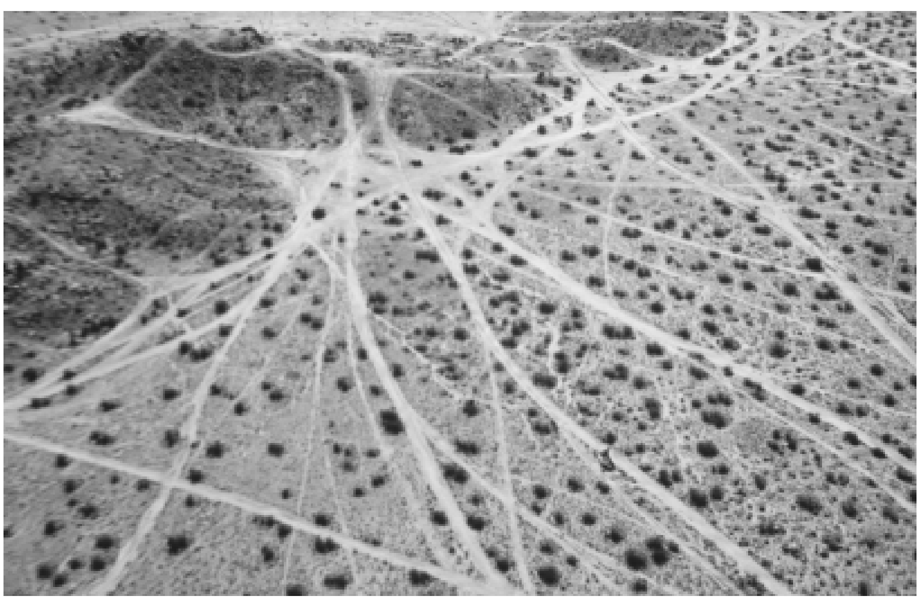
Off-road vehicle use has made many forest areas unsafe for people who live in and near the San Bernardino National Forest.

In the heart of the forest's upper elevations (the San Bernardino mountains), people like John Henderson must endure the constant roar of off-road motorcycles and off-road vehicles behind their homes.

"There is a hill they climb right behind us," Henderson explained. "The noise level is intolerable. They come by in packs. I have written letters to the Forest Service, but never got any answer."

Efforts to limit motorcycles and other off-road vehicles to designated roads and trails have met with only limited success. "I am more likely to encounter vehicles on locations prohibited to them than on a trail or road designated for them," said Tom Walsh of Blue Jay.

Sometimes, this use has disastrous effects. The Willow Fire, which burned 63,000 acres and destroyed several homes in 1999, "was strongly suspected to be from OHV riders leaving an unattended campfire," according to a government report.



Howard Wilshire

How much is too much? California has over 100,000 miles of off-road vehicle routes.

Bob Sollima, caretaker at Reds Meadow, describes the scene he witnesses in winter. "It seems that all the snowmobile pilots had 'moto-jump mania,' getting air off anything including buildings and vehicles. In one winter, I tallied damages to fences, hitch rails, a stone barbecue, a roof, my truck, windshield, two tree squirrels, a pine marten, and a fire hydrant [that] was sheared off.

"Some of the snowmobile abuses I've seen in the Reds Meadow Valley over the years, in times of low snowpack, are environmental damage to treetops, meadows, trails, creeks, lakes and the river. I've seen two snowmobiles stuck in the San Joaquin River, two submerged in Sotcher Lake and three overturned in creeks, spilling gas and oil into the water….

"This past holiday season, I saw some snowmobile abuses and a population density of 75 to 100 snowmobiles some days….My records show an average of one snowmobile per day in the wilderness area around here. That's about 200 to 250 wilderness ingresses per season. I've seen evidence of 14 snowmobilers in the wilderness in just two weeks."

Reds Meadow

In the Reds Meadow area of the Sierra Nevada mountains, illegal snowmobile trespass into wilderness areas happens quite frequently.

Each winter, the Reds Meadow area of the eastern Sierra Nevada becomes a popular snowmobile playground. Numerous snowmobilers ignore the rules and cross into campgrounds, wilderness areas and even Devil's Postpile National Monument. In most cases, the offenders are never cited or even confronted by law enforcement officers.

Snowmobiles regularly illegally enter Reds Meadow campground, often leaving behind beer cans and other trash. Dozens of other violations were reported to the Inyo National Forest, including:

- willful trespass into wilderness, Devil's Postpile, and closed campgrounds.
- destruction of property.
- damage to vegetation.
- physical threats to area residents.

Throughout the season, damage was inflicted on vegetation and soil, particularly when snowmobilers drove over patchy snow where ground was exposed. Tops of trees were snapped off. Campgrounds, signs and private property were routinely wrecked by rogue riders. Sometimes the polluting machines crashed and overturned into creeks and streams, spilling gas and oil into waterways. The snowmobiles also routinely drove around locked gates and past closure signs.

Bob Sollima

Sign damaged by reckless snowmobile use.

Jim Rose

Motorized madness: Conflicts are growing between off-roaders and hikers, equestrians, and other non-motorized recreationists. Sometimes, trails are difficult to share.

A plan for the future

These problems highlight the need to reform off-road vehicle management policies throughout California. We present below a fifteen-point plan for creating a more balanced and fair off-road vehicle policy in California. If implemented, this plan will:

• Minimize damage to California's landscapes,

• Reduce conflicts between motorized recreationists and other public land users,

• Prevent the harassment of landowners by motorized recreationists,

• Reduce illegal riding and trespass into wilderness areas, and

• Balance motorized recreation with other public land uses, such as hiking, horseback riding, mountain biking, hunting, fishing, and camping.

"There is a hill they climb right behind us. The noise level is intolerable. They come by in packs. I have written letters to the Forest Service, but never got any answer."
—John Henderson, Lake Arrowhead

The plan includes three elements: federal reform, state legislative reform, and state administrative reform. These elements are described below.

Frank Rauchschwalbe

A dirt bike churns up a creek in the Sierra Nevada mountains. Off-road vehicles degrade wildlife habitat, destroy vegetation and contribute to serious air and water pollution.

FEDERAL REFORM

Simply complying with the executive orders issued in 1972 by President Richard Nixon and 1977 by President Jimmy Carter and their own regulations will carry federal land managers a long way towards cleaning up off-road vehicle abuses of federal lands. To comply with current laws and alleviate the damage caused by off-road vehicles to public lands, federal land managers should:

Designate and map legal riding routes

Motorized vehicle use should be limited to vehicle routes designated, mapped, and posted by the appropriate land management agency as open to motor vehicle use, after the completion of environmental impact analyses. Motor vehicle use off designated routes should be prohibited.

"My home and refuge is no longer a sanctuary. This is my life and my home, and the homes of my neighbors.…The place where I would seek peace is in the woods. And that's where I'd get run over."
—Judith Spencer, Arnold

Determine where use is appropriate

All vehicle routes on federal land should be subject to environmental impact analysis, and motor vehicle use should be allowed only on those routes where the appropriate land management agency has documented that vehicle use will not cause adverse environmental impacts, and that impacts to the environment and other recreationists will be minimized.

Thresholds for unacceptable impacts must be established prior to beginning analysis.

Monitor the effects

The use of motorized vehicles should be allowed only in those areas where federal land managers are able to actively monitor the effects of motorized vehicles on the landscape. If monitoring determines that thresholds established for unacceptable impacts are reached in an area or trail, the area or trail must be closed until the impact is reduced to an acceptable level.

Protect undesignated wilderness

Across the state, wilderness-quality lands are being degraded by motorized vehicles. These areas should be declared off-limits to motorized vehicles.

Enforce the law

Some federal land managers are failing to prevent motorized vehicles from entering wilderness and other closed areas. Land managers should make trespass and closure violations a higher priority. Further, Congress should appropriate additional funds to assist land managers in enforcing federal laws and regulations.

STATE REFORM

In 1971, California enacted the Chappie-Z'Berg Off-Highway Motor Vehicle Act (OHV Act), which created the State of California off-road vehicle program. In the past three decades, this program has allocated over half a billion dollars to support off-road vehicle use on state, federal, and private land throughout California.

The OHV Act, as amended, found that "the indiscriminate and uncontrolled use of those vehicles may have a deleterious impact on the environment, wildlife habitats, native wildlife, and native flora," and that "effectively managed areas and adequate facilities for the use of off-highway vehicles and conservation and enforcement are essential for ecologically balanced recreation."

Without active involvement, the state runs the risk of being held responsible for the shortcomings of off-road vehicle management, while remaining unrecognized for the benefits of its grants program. The following legislative reform will help to bring balance to the state's off-road vehicle program, by ensuring that state funding is used to repair damaged areas, prevent future damage, and mitigate the effects of off-road recreation. State legislators should require:

Hiking, nature wildlife study, and camping are among the most popular outdoor recreation activities in the state. Off-road recreation, dirt biking and snowmobiling are among the least popular.

Increased funding for conservation and law enforcement

The state's off-road vehicle act urges California to control the impacts caused by the "indiscriminate and uncontrolled use" on the "environment, wildlife habitats, native wildlife, and native flora." This means that funding is needed to effectively enforce closed areas, protect soils and watersheds, carry out monitoring and remediation

Jim Rose

Jim Rose

Neighboring hillsides in the Jawbone Canyon and Dove Springs off-road vehicle areas, where decades of overuse has left denuded slopes, destroyed vegetation and serious erosion problems.

Laying waste to an American heritage: these geoglyphs in the California desert are thousands of years old. Hundreds, if not thousands, of sensitive archaeological sites like the one shown above have been damaged or destroyed by off-road vehicle use.

work, and keep riders on designated routes. Current funding is not adequate to fulfill these needs, and additional funding should be authorized.

Mitigation funding and non-motorized buffers

Off-road vehicles can cause extensive harm to the natural environment and wildlife habitat. Funding for mitigation of off-road vehicle damage is needed to ensure that critical habitat areas are protected. This mitigation may be possible at the site of the off-road vehicle use, or may be more appropriate elsewhere.

Uniform soil and habitat standards

Currently the state is utilizing highly technical soil protection standards that are difficult for non-geologists to apply. These standards should be updated and applied uniformly.

In no other state program do taxpayers reward illegal activity by subsidizing more opportunities for the same activities.

Polluter pays

Registration fees for off-road vehicles should be linked to emissions levels (higher emissions equals higher fees). This will create a positive incentive to reduce emissions from off-road vehicles.

Reducing off-road vehicle-related crimes

Currently, fines for riding a motor vehicle into closed areas are too low to effectively discourage use. Fines for vehicle trespass into closed areas should be dramatically increased to create a real deterrent to illegal riding.

OFF-ROAD VEHICLE GRANTS PROGRAM

Through its off-road vehicle grants program (which provides millions of dollars each year to support the acquisition, development, and operations of off-road vehicle facilities and areas on federally managed lands), the state is in a unique position to positively influence off-road vehicle management on public lands.

In the past, grants have been used by federal agencies to supplant federal funds. Grants should supplement, not replace, federal appropriations, and should not be used as a surrogate for federal funding to carry out land management responsibilities.

In order for the program to adequately mitigate the effects of off-road recreation and prevent excessive off-road vehicle-related damage, the state should adopt the following principles with regard to its off-road vehicle grants program:

Comply with the law

The top priority of the grants program should be to monitor and repair existing resource damage, prevent future damage, and ensure compliance with state and federal laws and regulations. Grants should not be given to districts that cannot ensure compliance with federal and state laws and regulations, except to

Throughout the desert, prehistoric remains have been "run over and ridden through, and tires have been spun on them, causing soil erosion, surface soil displacement, vegetation loss, and the loss of scientifically important stratigraphy associated with the sites."
—BLM archaeological site protection grant proposal

Jim Rose

Jim Rose

The Knoxville off-road vehicle area has long received state funding. The area is so unsafe that local residents advise against visiting there alone. "Families come to the Knoxville area thinking that they have a safe place for their children to learn to ride. Instead they've entered an area with no rules, no road signs, and no real BLM presence."
—Jim Erasmy

bring those areas into compliance with the law.

Protect sensitive areas

Grants to support projects that could adversely impact or jeopardize the ecological integrity or social values of wild areas or rivers should be eliminated.

Prevent future damage

Acquisition and development of new off-road vehicle areas and trails should cease until all lands within the program are in full compliance with all applicable state and federal laws, regulations, and policies, and current resource damage is adequately addressed.

Respect other land users

Grants should not be used to fund projects that create or expand conflicts with non-motorized recreationists. Projects submitted for grant awards should assure that residents and private property owners adjacent to the proposed project area are protected from noise, trespass, and property damage.

Do no harm

The state should not fund off-road vehicle activities in areas where off-road vehicle use has been shown to cause unacceptable environmental damage, where off-road vehicle use will lead to damage of sensitive lands, or where off-road vehicle use will lead to an increase in illegal riding or conflicts with other recreationists. In addition, the state should not fund areas that cannot demonstrate compliance with all federal and state laws and policies.

Through the enactment of the above reforms, California can head off an environmental disaster in the making.

Our fragile heritage is at risk, and immediate action is needed to ensure it is maintained, intact, for the benefit of future generations.

In an otherwise quiet forest, the sound of a loud motorcycle can be heard for over 11,500 feet—a distance of over two miles!

Introduction

California contains many of the world's most diverse landscapes. The state's borders enclose snow-capped mountains, fragile deserts, rugged coast lines, and oak savannas.

To escape the roar of the city, many of California's residents seek to enjoy the peace and quiet of the great outdoors. They are often surprised when they arrive: California's backcountry is being methodically overrun by irresponsible motorized vehicle use.

Motorized vehicles are a fact of life in California. But a special class of these vehicles—called off-road vehicles—leave a unique mark. These vehicles, which include dirt bikes, aggressive all-terrain vehicles, 4x4s, and snowmobiles, are made to be ridden in backcountry areas and across wildlands. The state currently contains from half a million to two million such vehicles.[1]

They leave tracks and scars that are visible for generations. Their impacts are significant and often permanent.

This report highlights some of the areas that have been affected by off-road vehicles, and makes recommendations for corrective actions. The case studies presented within this report are by no means unique: they are merely examples of the problems that exist across California's landscapes.

In the long run, whether or not California's natural heritage is preserved, or continues to be degraded, depends in part on whether the state can come to grips with the proliferation and damage caused by motorized vehicles.

Howard Wilshire

Hungry Valley State Vehicular Recreation Area, where uncontrolled off-road vehicle use has caused permanent damage. This area can likely never be restored.

Damage to land and people

More off-road vehicles (ORVs) make tracks across California's public lands than anywhere else in the U. S.[2] In fact, the amount of public land open to off-road vehicle use in California is at least double that of any other state.[3]

Off-road vehicles generate various types of environmental and public harm in California. The primary environmental damage is inflicted on soil, plants, and wildlife. Noise, air and water pollution are also problems. Some off-road vehicle users trespass onto private lands and into wilderness, drive illegally on closed roads and areas, create conflicts with other trail and backcountry users, and pose public safety hazards due to disregard for existing regulations.

Many off-road vehicle enthusiasts obey the rules. Yet problems are widespread, documented everywhere off-road vehicle riding occurs, and causing tremendous impacts on the environment as well as the experience of other outdoor recreationists. As one writer explained:

"St. Francis of Assisi himself while driving an ORV on wild land could not avoid diminishing the recreational experience of many non-ORVers in the same area. (Nor could he prevent much of the environmental degradation.)"[4]

User conflicts

One of the biggest source of conflict with motorized recreation is the


Jim Rose

Saxon Creek, Lake Tahoe Basin. According to one researcher, where off-road vehicle use is heavy, all life is eventually destroyed.

sound of the engines. In an otherwise quiet forest, the noise from an average motorcycle can be heard for 7,000 feet.[5] Some louder engines can be heard for over 11,500 feet—a distance of over two miles![6]

Off-road vehicle conflicts with hikers, backpackers, hunters and other outdoor enthusiasts are extremely common. Engine noise ruins the solitude of the backcountry. Dust kicked up by giant tires chokes hikers and backpackers exploring on foot. Other times, the conflicts are dangerous. Numerous hikers, bicyclists, and equestrians have been forced off trails in hazardous "near miss" situations by reckless off-road vehicle riders.

Recreational planner Robert Badaracco has long studied the phenomenon of off-roaders pushing out other users of outdoor areas. He observed this phenomenon taking place in the California desert, the Los Padres National Forest and the Sequoia region more than 20 years ago, and predicted that it would fast spread to other areas in the west.[7] History has proved him correct.

Landowner conflicts

For those living adjacent to popular off-road vehicle recreation areas, the clamor can make life miserable. A chronic off-road vehicle problem reported by the State Off-Highway Motor Vehicle Recreation Commission stems from "ORV recreationists straying off designated routes and causing resource damage, trespassing on private property and vehicular intrusion into wilderness areas."[8]

Unlawful riders, not content to stick to designated trails and areas, regularly forge their own paths across the desert and through forests, often showing blatant disregard for federal and state laws, the environment, and the rights of private property owners.

Air pollution

Many areas of the state are hot spots for smog,[9] and off-roaders contribute a significant share. The emissions of one two-stroke off-road motorcycle equals that of 118 passenger cars.[10] Newer, cleaner four-stroke engines still emit more than seven times the level of carbon monoxide as new cars.

To date, only ten models of off-road vehicles have been certified to meet the state's new air quality standards.[11] Yet the California Air Resources Board now allows the continued use of off-road vehicles that don't meet state emission standards.

Water pollution

Two-stroke engines, like those found on many motorcycles and snowmobiles, are notorious for spilling one-third of their gas-and-oil mixture of fuel unburned into the environment.[12] The effects are so severe that jet skis were recently banned from Lake Tahoe to prevent excess pollution. Yet, off-road vehicles have not been banned.

Numerous hikers, bicyclists, and equestrians have been forced off trails in hazardous "near miss" situations by reckless off-road vehicle riders.

Off-road vehicles can also leak fuel, oil, antifreeze and other toxic chemicals into streams and creeks, creating a potential source of ground water pollution.

Lawlessness and illegal riding

Trespass into wilderness and other closed areas is an ongoing problem.

Trespass into more than 40 wilderness and other protected areas across the state that are closed to vehicle use was documented for this report. Many more areas are likely suffering the same fate.

This widespread problem is largely ignored by public land managers. During 1998 and 1999, the U. S. Forest Service issued only 64 citations for closure violations.[13] The Bureau of Land Management did even worse, giving only ten wilderness trespass citations in 1999 *throughout the entire state*.[14] These numbers pale in comparison to the numbers of reported violations.[15]

Steve Tabor, Desert Survivors

Fresh wheel tracks 2.7 miles inside the Rice Valley Wilderness. This is part of an old BLM jeep trail within wilderness that was closed in 1994 by the California Desert Protection Act. It is still regularly used by groups of off-roaders today.

Destruction of soil and vegetation

Every time a tire crosses the earth, damage is done.[16] A motorcycle driven 20 miles on a flat desert surface impacts one acre of land and commonly displaces from 15 to 66 tons of soil in those 20 miles.[17] An average four-wheel drive vehicle disturbs an acre of land in just six miles of travel, and in that distance moves up to 300 tons of soil on steep slopes in just one pass.[18]

On hillsides, soils and rocks are ripped up by vehicles and sent hurtling downhill. This type of wasting leads to notches or grooves in the surface as deep as six feet in soft soils and loose rock and even down to three feet deep in hard rock.[19] In some heavily used off-road vehicle areas of California, erosion has occurred at rates 86 times higher than federal standards![20]

A heavily used section of the ridge lost 11,000 metric tons of soil in the months after a heavy fall rain.
—Erosion Off the Road

Wildlife disturbance

The state of California is only now beginning to monitor and research the effects of off-road vehicles on native wildlife. However, studies have already found that the “deleterious effect of off-road vehicles on native plants and animals is undeniable.”[21] In fact, the Geological Society of America reported more than 20 years ago that “where off-road vehicle use is heavy, virtually all existing life is ultimately destroyed.”[22]

Animals are run over both on roads that cut through their habitat and in open areas where they normally never see a vehicle. They are also molested, maimed, shot and carried off illegally.[23] Some species will not cross roadways; these roadways then serve to fragment and reduce the animals’ home range.[24] Studies in the Mojave Desert found that off-road vehicles severely reduced the numbers and types of reptiles.[25]

Ample opportunity

According to the State of California Off-Highway Motor Vehicle Recreation Division, the state features over 100,000 miles of roads and trails for off-road vehicle recreation.[26] California’s national forests alone contain over 44,000 miles of vehicle routes.

In addition, California contains hundreds of thousands of acres of open riding areas in the desert. The State of California also operates seven off-road vehicle recreation areas, totaling more than 90,000 acres. Numerous small, locally operated off-road vehicle areas are also spread throughout the state, providing even more opportunity for off-road vehicle enthusiasts.

With over a hundred thousand miles of roads and trails, hundreds of thousands of acres of open areas, and numerous state and local parks, off-road vehicle users have the opportunity to ride in every corner of California.

Still, some off-road vehicle users complain about lack of opportunity and access. With so much opportunity currently available, the question is, "How much is enough?"

The public speaks

Numerous public opinion polls have demonstrated California residents' preferences regarding outdoor recreation. Perhaps the most commanding survey was completed by the State of California Department of Parks and Recreation.[27]

Among the report's findings were:

• Hiking, nature study, and camping are among the most popular outdoor recreational activities in the state. Off-road 4x4 recreation, dirt biking, and snowmobiling are among the least popular. Of the 43 recreational activities surveyed, snowmobiling was the least popular of all.

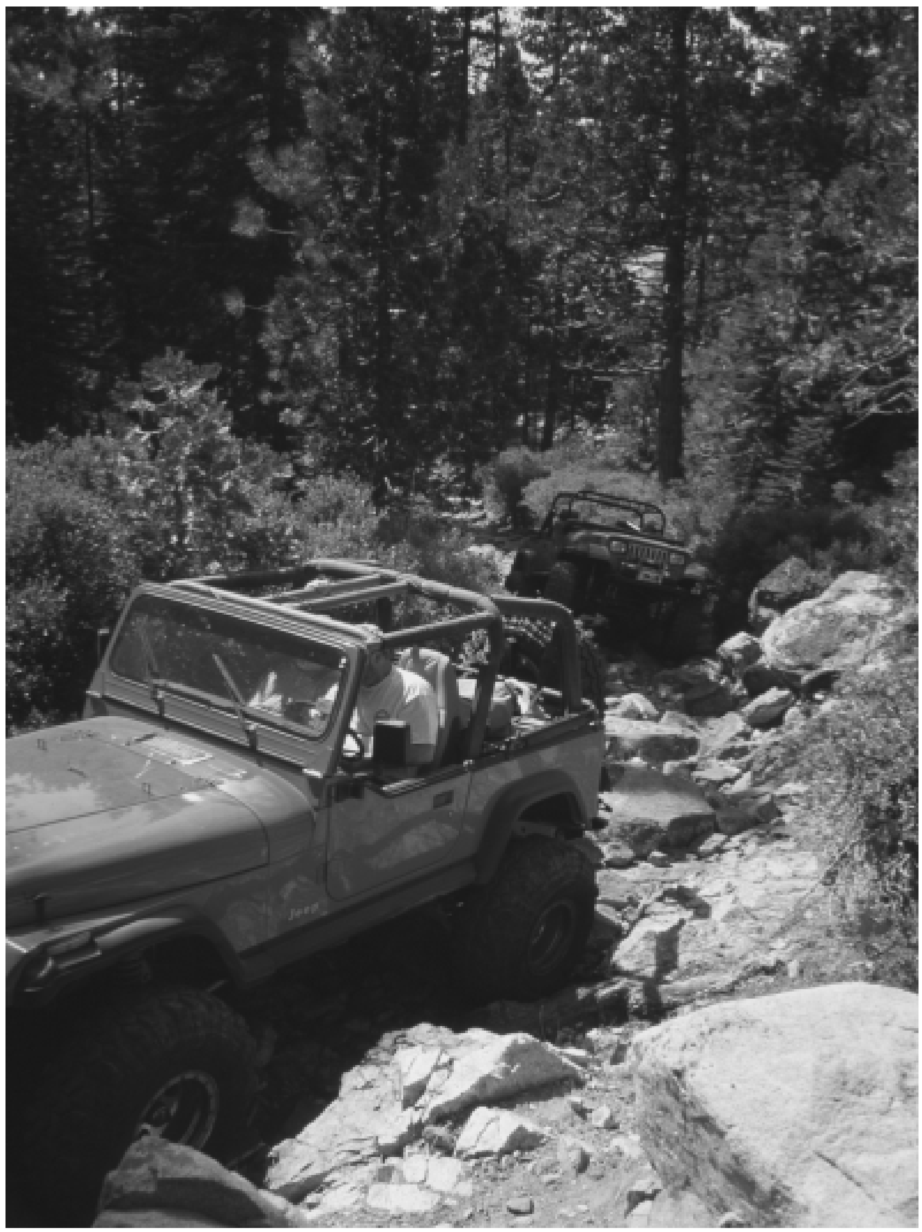
• Latent (unmet) demand was ranked "low" for snowmobiling, dirt biking, and off-road 4x4 driving. Demand was high for hiking, backpacking, and nature observation.

• Public support for providing public funding for snowmobiling, dirt biking, and off-road 4x4 driving was ranked "low" by respondents. Support for public funding of hiking, backpacking, and camping opportunities was high.

• Overwhelmingly, respondents agreed that protection of the natural environment is an important aspect of outdoor recreation areas.

Further, in a recent survey, California residents were asked, "How much is enough?" with regard to the numbers of roads and trails that are available to off-road vehicles. The consensus from that

In one area, large groups of off-roaders regularly dump trash onto desert floor, race across private property, and trample the sage and chaparral. In another, dust from off-roading sometimes gets so thick that brown clouds of dirt obstruct the view of passing drivers on Highway 395.



Jim Rose

A recent study showed that hiking, camping and wildlife viewing are among the most popular recreation activities in California. Off-road recreation, like that on the Fordyce Trail shown here, is among the least popular.

survey was that 100,000 miles of roads and trails for off-road vehicles is far too excessive. Survey respondents agreed that half of those road miles should be closed to off-road vehicles.[28]

Federal off-road vehicle management: Losing ground

On federal lands, off-road vehicle management is subject to a myriad of confusing, occasionally conflicting laws and policies. Regional land management plans aim to designate where off-road vehicles can and cannot be driven. Sadly, these plans are often written with incomplete environmental analysis, and are often unenforced or even unenforceable due to their imperfect drafting.

Perhaps the clearest direction regarding off-road vehicle management on public lands was given by President Richard Nixon in 1972. That year, the President signed an executive order that attempted to control the damage being caused by off-road vehicles.[29]

Among other things, the President directed each land management agency to issue clear directions as to which areas and trails were open to off-road vehicle use, and which areas and trails were closed. The President also gave guidance as to how those areas and trails should be designated and required monitoring to assess and minimize impacts.

Sadly, 29 years after President Nixon issued this order, its contents are still roundly disregarded by the federal land management agencies. The study and designation of vehicle routes—explicitly required by the order—have rarely been completed. When route designation has been completed, it is often without the analysis of the four criteria spelled out by the President's Order.

A 1995 report by the General Accounting Office (GAO) confirms these findings.[30] GAO auditors visited eight popular off-road vehicle locations (including three in California) to determine how well those areas had complied with the executive orders. The report found that compliance with the orders was extremely spotty. Inventories and maps were not complete. Signs were missing. Monitoring was "virtually non-existent."

The report's authors concluded that without additional funding, and a greater emphasis placed on compliance, the executive orders would never be fully implemented.[31]

Despite clear legal requirements to do so, federal land management agencies have not analyzed, nor have they designated, legal off-road vehicle routes throughout much of California.

Mimi Jennick

State of California Off-Road Vehicle Program: Struggling to Keep Up

Since 1971, the State of California has managed an off-road vehicle program with revenues from state taxes on gasoline. By 2000, the program had mushroomed into a $35 million undertaking, complete with responsibilities for managing seven off-road vehicle recreation areas, and providing millions of dollars to federal agencies for the management of off-road vehicle opportunities on federal lands.

While often touted as a "model program" by off-road vehicle enthusiasts, the state's program is actually fraught with problems. For example, over 50% of the income from the program is derived from tax on the fuel used by illegal, unregistered vehicles.[32] In no other state program do taxpayers reward illegal activity by subsidizing more opportunities for the same activities.

Under the administration of Governor Gray Davis, the state's program has made significant moves to address past problems created by off-road vehicle use and abuse. The state has completed new program regulations, and placed a higher emphasis on funding for conservation and law enforcement. The state has also undertaken a review of legal requirements, and is attempting to bring the program into compliance with state laws.

These are welcome changes that have helped to bring to light the numerous problems in state and federal off-road vehicle management.

However, numerous questions still remain, and the state program will require constant scrutiny to keep it on track.

The California Desert

Three distinct desert types form the 25 million-acre California desert: the Mojave, the Sonoran, and the Great Basin. Much of California's vast desert expanse—over 10.5 million acres—is managed by the Bureau of Land Management. The area supports a magnificent array of diverse, wild places.

More than nine million people visit the California desert each year.[33] Most use is low-impact, but off-road vehicle use alone has impacted more than one million acres.[34] The desert is now one of the most popular off-road vehicle riding areas in the entire world and hosts more than 70% of off-road vehicle use in the state.[35] The area features more than 45,000 miles of paved and dirt roads.[36] In fact, 95% of the desert is now within three miles of a road.[37]

The problems caused by off-road vehicles can be severe. Desert soils are extremely fragile and recover very slowly. The tracks made by tanks driven by General Patton's army during maneuvers in the 1940s are still easily seen today. And now the tires of motorcycles, dune-buggies, ATVs and four-wheel drive vehicles leave behind trails of destruction that may not recover for millennia.[38]

The BLM has largely failed to make and enforce off-road vehicle use designations throughout the desert, and regulation of off-road vehicle use is still based on land management plans from the 1970s and 1980s, when off-road vehicle use first boomed.[39]

Further, illegal riding continues to be a persistent problem in the desert. According to the BLM's Needles Field Office, "the most significant issues related to off-highway vehicles are travel off of existing roads and use of vehicles in designated wilderness areas." The Needles staff found that "the majority of damage in all areas was a result of vehicles traveling off the existing route...."[40]

The following excerpts highlight the problems facing the California desert.

A L G O D O N E S D U N E S

Lawlessness and Devastation

- Imperial County
- Bureau of Land Management, El Centro Office
- Off-Road Vehicle Designation: South Algodones Dunes-open (49,310 acres closed in November 2000 to protect milkvetch)
North Algodones Dunes Wilderness Area-closed
- Issues/Concerns: lawlessness, environmental damage, wilderness trespass
- Key species: Peirson's milkvetch (threatened), fringe-toed lizard (endangered), flat-tailed horned lizard (species of concern)

Jim Rose

The land behind the sign is legally designated wilderness, and has been closed to off-road vehicles since 1994. Despite the closure, off-road vehicle riders regularly trespass into the wilderness.

The ancient sands of Algodones Dunes form a shifting white band along the southernmost reaches of the Imperial Valley. At 40 miles long and five miles wide, this is the largest dune system in California. It is also the landscape most heavily traveled by off-road vehicles in the Golden State.

Over the past 20 years, hundreds of thousands of motorized vehicle users have driven the towering dunes, traveling freely up and down the sandy hills and washes. Large unruly crowds and inadequate law enforcement have caused the region to become dangerous for people as well as the region's native plants and wildlife.

Until recently, the legions of riders have been allowed to motor almost anywhere, anytime on more than 70 percent of this 150,000-acre public resource.[41] Visitation exceeds 1,000,000 each year, peaking on holiday weekends such as Thanksgiving, when as many as 100,000 visitors travel to the dunes for an off-road experience.[42] Millions of dollars of state monies fund this public safety and environmental fiasco.

Lawlessness has escalated to the point where near-riot conditions prevail.[43] The dunes have become unsafe for family recreation activity due to the use of drugs and alcohol, and the problems of lawlessness that occur with such use.[44]

A tragic example was Halloween 2000, when three people were killed in accidents and several others severely injured.[45] In addition, BLM rangers have been threatened and deliberately run over by dune buggies.[46]

"The problem has advanced to the stage that the normal, law-abiding citizens are fearful of using the area," said a Bureau of Land Management report.[47] However, the BLM has also admitted its "inability to bring in enough law enforcement rangers" at Thanksgiving and other times due to a chronic shortage of staff and a public safety workload "nowhere equaled in all of BLM."[48]

The BLM El Centro office requires a full-time staff of 12 rangers, but during most of 2000, the office was down to three rangers and one chief ranger.[49] The three were responsible for patrolling not just Algodones Dunes but the entire 1.2 million acres managed by the BLM El Centro office. On weekends, all available staff were pulled to the dunes, leaving the rest of the desert wide open to recklessness and resource devastation by other unlawful off-roaders.

The agency's problems in controlling the lawlessness remain unresolved, as evidenced by BLM special agent Roger Bruckner's recent statements in the *Yuma Sun:* "He...said last weekend 'was a really good weekend,' despite the deaths and serious injuries."[50]

Environmental damage

The Algodones Dunes constitute a unique habitat type in the Sonoran Desert with desert pools and several plant communities that provide habitat for rare plants, reptiles, beetles and other creatures.

Peirson's milk-vetch, a species federally listed as threatened, is a silvery, short-lived perennial plant and a member of the pea family. It is several feet tall and produces small purple flowers in the spring. Found only in the Algodones Dunes, it has been driven literally almost to extinction by rampant off-road travel.

The endangered fringe-toed lizard and rare flat-tailed horned lizard are also at risk from off-road travel in the dunes. Both burrow just beneath the surface of the sand and are vulnerable to being crushed under the pressure of a vehicle tire. The flat-tailed horned lizard is known to freeze when in danger, leaving it even more at risk of being flattened when on the surface.

Areas of the Algodones Dunes that are heavily used by off-roaders are devoid of wildlife and native plants.[51]

Trespass and citations

Wilderness trespass into the North Algodones Dune Wilderness Area is

Jim Rose

Visitation at Algodones Dunes exceeds 1,000,000 people each year, peaking on holidays such as Thanksgiving, when as many as 100,000 visitors travel to the dunes for an off-road experience.

FLAT-TAILED HORNED LIZARD

(*Phrynosoma mcallii*)

Status: BLM sensitive

As its name suggests, this lizard has a long, broad and flattened tail. And, yes, it has horns. Most people have never heard of it, but the lizard has made headlines in southern California.

Off-road user groups, with the help of state commissioners, fought to keep the rare reptile off the endangered species list—proposing voluntary measures instead.[55] Both state and federal wildlife officials wanted to give the lizard added protections due to several threats, including off-road vehicle use.[56] The state off-highway commission voted against them, and today the lizard must fight for survival in areas of heavy vehicle use.[57]

The tire-scarred acreage inside the Ocotillo Wells State Vehicular Area was made a horned lizard research area.[58] Organized racing events run right through prime horned lizard territory, including special areas set aside for lizard protection.[59]

This lizard is more likely to be run over than most of its cousins, because it freezes instead of running or taking cover when danger comes.[60] Like other lizards, it can't burrow well into sand or soil that has been driven over and compacted, so its natural cover is removed by motorized recreation. The wrecking of plants means less places to hide and forage for food.

The flat-tailed horned lizard may very well disappear if off-road use is not reduced or halted in key habitat areas.

George Barnes

The North Algodones Dunes Wilderness Area is home to several rare plants and animals. The area is off-limits to off-road vehicles.

cited in BLM reports and by wilderness travelers. The BLM lists these wilderness violations as "illegal wilderness entry from dunes, sign removal and defacing, illegal wilderness entry from private property."[52]

Natural resource protection was ranked as the second highest enforcement priority after public safety by BLM officials.[53] But citations for wilderness trespass and natural features destruction represented only 2% of the total number issued on key holiday weekends between Halloween 1999 and Easter 2000 (34 out of 1470).[54]

Status

As a result of a lawsuit by a coalition of environmental groups, the BLM recently agreed to temporarily close 49,310 acres of the dunes to motorized vehicles, until a permanent solution is developed to save the Peirson's milk-vetch from extinction. Seventy thousand acres remain open to off-road vehicle use. To date, the closure has not been effectively enforced.

Recommendations

• The existing closures should remain in place, be made permanent, and be expanded to include closure of the Mammoth Wash area north of the wilderness.

• If off-road vehicle trespass into the closure areas cannot be minimized, then larger closures should be implemented.

• The East Mesa Area of Critical Environmental Concern and class L and M lands east of the dunes should be closed to off-road vehicles and RV camping.

• Microphyll woodland washes east of the dunes should be closed to motor vehicle use.

• Law enforcement in the dunes should be dramatically increased. (Increased enforcement is the most requested service by the public in comments to the El Centro BLM office.)

• Increased monitoring on the impacts of off-road vehicles on fragile dune vegetation and wildlife should be implemented and management altered to reflect the results of this monitoring.

• The BLM should conduct a carrying capacity report on the dunes and restrict visitors to numbers within the area's capacity.

• State funding should be used to ensure that the area is properly managed, or state funding should be withdrawn if proper management cannot be achieved.

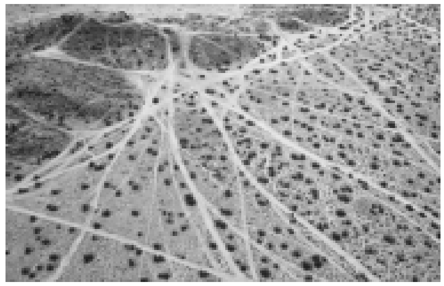
Controlling Routes through the Desert

- County: San Bernardino
- BLM, Barstow Field Office, Ord Mountain Planning Unit
- Off-Road Vehicle Designation: Limited Use-designated routes only
- Issues/Concerns: illegal route proliferation, erosion and trampling, wildlife damage, lack of enforcement and funding
- Key species: desert tortoise (threatened), Peninsular bighorn sheep (endangered)

The Ord Mountains region in the Mojave Desert southeast of Barstow encompasses rocky high desert, low washes, badlands and sensitive plant communities. Wildflowers bloom brightly in spring when the climate is right. The area attracts large numbers of recreationists from southern California due to its beauty and easy access from major highways.

This planning unit is located entirely within critical habitat for desert tortoise. It is also important to desert bighorn sheep, horned lizards, raptors, bats and other desert animals. Covering more than 100,000 acres of public land, it is an integral part of the California Desert Conservation Area.[61]

Nevertheless, for as long as 50 years, an uncontrolled network of off-road vehicle routes has criss-crossed this unique desert area.[62] Between 1977 and 1989, mileage jumped by 28%—even though none of these routes had ever been officially designated or approved.[63]



Howard Wilshire

By failing to designate legal riding routes, the BLM invites "route proliferation": an excess of vehicle routes, often with devastating effects.

The Ord Mountains are also threatened because they are sandwiched between the Stoddard and Johnson Valley areas, which are open to off-road vehicle use. Past races allowed by the BLM through the Ords have attracted off-roaders to the area. BLM has since ceased to allow racing through the Ords. This protection from races must continue to benefit tortoise recovery.

Illegal travel into the Ord Mountains is exacerbated by vandalism and removal of route signs and other traffic control mechanisms. The Cinnamon Hills, a rock-climbing area known also as Finger Rock, attracts off-road vehicle users who create new illegal trails.[64] Increased law enforcement has

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"I have lived on my private land in the Northern El Mirage Valley for the past five years. I have installed solar electricity and have a heavily insulated wooden house. We haul in our water from a private well some 12 miles distant.

"From the very first months that I lived here, I noticed a wanton disregard for safety and the law displayed by visitors to this area. Shooting machine guns, building bonfires in high winds and abusing the integrity of the land for recreational purposes (i. e., traveling cross-country at unsafe speeds) are just some of the behaviors I have observed.

"I began a journal of the calls I made to the San Bernardino Sheriff's Department and to BLM Emergency. BLM Ranger Richard Bayer, among others, has responded to the site known as the 'Edwards Bowls,' which is adjacent to my property, many times in the past five years. About two years ago, Bayer put up signs advising visitors of the shooting regulations as well as the fire, fireworks and motorcycle safety requirements. These signs have been repeatedly vandalized.

"We have put up 'No Trespassing' signs at the perimeters of private land. Motorcycles just drive by them as they blaze new trails into virgin property. I have repeatedly reported these activities, but no one has been apprehended."

—Douglas Parham, acting president,Western San Bernardino County Landowner's Association in Palmdale

DESERT TORTOISE
(*Gopherus agassizii*)

Status: Threatened


Howard Wilshire

California's state reptile, the rare desert tortoise, has burrowed into the hot sands of the California desert for millions of years. This slow-moving, long-lived reptile was once abundant throughout southeastern California.[71] Tortoise images, dating back thousands of years, have been found left behind by Native Americans.[72]

Today the tortoise's home, impacted by thousands of miles of roads, has been carved into remnants of its original size. Off-road vehicles race across the land, crushing tortoises and their burrows.

Off-road vehicles are among the most destructive and wide-spread threats to the tortoise's survival.[73] Not only do the vehicles run over these native crawlers, they also destroy vegetation, crush burrows, and ruin soils necessary for survival.

In one desert location, off-road vehicles cut in half the number of tortoises and burrows compared to a similar place without motorized recreation.[74] In Fremont Valley, 40% of tortoises found dead on a study plot were killed by gunshot or vehicles traveling cross-country or on trails.[75] As of 1980, off-road vehicles ran through one-quarter of all tortoise habitat and more than two-thirds of desert land with high densities of tortoises.[76]

Saving the ancient tortoise will require decisive action to reduce the out-of-control use of motorized vehicles in the desert.

apparently not been sufficient to prevent this illegal riding.

Off-road vehicle use has taken a toll on the soils, vegetation and wildlife in the area. Off-road vehicles crush desert tortoises on the surface and in their burrows. They chase away bighorn sheep from watering holes. And the soil compaction by the tires and weight of the machines makes it difficult for desert horned lizards to dig to safety underground.[65]

Many of the routes go directly through washes that serve as travel corridors for bighorn sheep and nesting areas for desert tortoises. Route mapping found that most of the routes in washes (70%) and almost all motorcycle routes (86%) went directly through zones known to house as many as 20 to 50 tortoises per square mile.[66]

Although BLM was required to designate legal off-road vehicle routes by the California Desert Conservation Area plan in 1980, it wasn't until 1995 that the agency finally acted. At that time, the BLM closed more than 300 miles of routes and trails due to "escalating regional route proliferation and concern for the survival of the desert tortoise."[67] More than 100 miles of routes were left open and 150 designated for use by private landowners with inholdings.[68] However, illegal route proliferation continues and the planning process for permanent designation remains stalled.

In 1996, using GIS (Geographic Information Systems) technology, BLM undertook the most extensive route inventory ever completed on BLM lands anywhere.[69] More than 600 miles of routes were identified. Desert tortoise habitat was analyzed and ranked. Areas important to bighorn sheep, special plants, birds of prey and other desert natural resources were reviewed. A plan was developed to designate routes and rehabilitate damaged areas.

Unfortunately, the plan proposed to increase open route mileage over the emergency plan by nearly 50%, growing from 100 to 153 miles of open routes.[70] Further, the designated open routes would continue to cross essential desert tortoise, bighorn sheep, and lizard habitat. Thus the plan was stalled.

The BLM has agreed to maintain the existing emergency route network for the Ords under a lawsuit settlement agreement with the Center for Biological Diversity. While less than perfect, the existing network provides more protection for species than the BLM's preferred alternative in the Ord Route Network Draft Environmental Impact Statement.

The Ord Mountain route designation process now awaits the completion of the West Mojave Desert Coordinated Management Plan, which has been proposed since 1992.

Recommendations

- The Stoddard to Johnson route should be canceled and any future off-road vehicle events through the Ords disallowed.
- The existing emergency route network should be maintained until a final decision has been been made through the West Mojave plan.
- The Ord Mountain route designation process should be completed and implemented. Open routes should be identified and signed, and closed routes should be rehabilitated.
- Road mileage should be dramatically reduced in desert tortoise and bighorn sheep habitat.
- Law enforcement should be increased, and higher penalties for illegal riding should be implemented.

Erosion at its Worst

• County: Kern

• Bureau of Land Management, Ridgecrest Office

• Off-Road Vehicle Designation: Open

• Issues/Concerns: Extreme erosion, trampling, impacts on adjacent lands

• Key species: Mojave ground squirrel, kangaroo rat, pocket mouse

Located in the northwestern Mojave Desert, Jawbone Canyon and Dove Springs off-road vehicle areas have been used by dirt bikes and other off-road vehicles for decades, causing tremendous impacts.

In some areas, intense use has stripped all but the largest shrubs from the hillsides and carved deep ruts into the landscape.[77] Unmanaged riding has caused the fragile desert soil to run downhill in huge quantities during rainstorms, burying what vegetation remains at the bottom of slopes.[78]

The damage at Jawbone Canyon is easily seen from the nearby Highway 14. Use by motorcycles, dune buggies, and 4-wheel drive vehicles has devastated extensive areas of the valley floor and canyon walls. The visible ruts and gouges may never disappear. Plant cover has been stripped and the soil churned up. After a rainfall, "the water formed a thick slurry of the loose soil, which flowed out over the canyon much like a lava flow, burying plants and trapping burrowing animals."[79]

Wildlife has largely disappeared from Jawbone Canyon and Dove Springs. Desert tortoise have been driven from the area, while the kangaroo rat and pocket mouse populations have just about disappeared.[80] The area was historically bighorn sheep habitat and also was home to the Mojave ground squirrel. Nesting birds of prey are often disturbed by visits from off-road vehicles at nearby Robber's Roost. Some drivers ignore the seasonal closure during times when prairie falcons and golden eagles nest.[81]

Although the level of disturbance to wildlife is high, there is great potential for ecological restoration here, starting with the protection of springs and riparian areas. Kelso Creek, Dove Springs and Butterbredt Springs are suitable habitat for the endangered Southwestern willow flycatcher and Least Bell's vireo. The flycatcher has been sighted in recent years in the area and is known to occur just west of the Jawbone-Dove Springs area on the south fork of the Kern River and at Lake Isabella.

More than 1,000 acres within Dove Springs Canyon have been heavily damaged and another 500 completely denuded.[82] The off-road vehicle use began in the lower part of the canyon, then moved up after the canyon floor was stripped of plant life and the walls rutted and mined with gullies.[83]

A 1983 study found that some upper slopes lost as much as one foot of surface soil over 13 years due to increased water and wind erosion on soils and vegetation caused by motorcycle use.[84] Joshua trees have been uprooted by wind as soil washed rapidly away from their base.[85]

The legally sanctioned off-road vehicle activities not only destroy the designated areas, but inflict damage on adjacent lands. Dove Springs is situated in the watershed above Red Rock Canyon State Park.

Jim Rose

Jim Rose

An unscarred hillslope within Jawbone Canyon provides a comparison to the scarred hillside within the Dove Springs off-road vehicle area. Note the deep gullies signifying extreme erosion, and the lack of vegetation.

Howard Wilshire

A ridge inside Red Rock Canyon State Park still bears the scars of off-road vehicle use, decades after the area was fenced off.

Sediments and pollution wash downstream, damaging the protected waterways and associated plant and animal life below.

A ridge inside the state park still bears the scars of off-road vehicle use after two decades. The ruts and gouges caused by hill climbing remain visible long after the area was closed in the 1970s. A heavily used section of the ridge lost 11,000 metric tons of soil in the months after a heavy fall rain.[86]

Travel between the two open areas is limited to designated trails, yet reports of illegal cross country riding and trespass into private property are common. As early as 1982, BLM reported that extensive riding outside the designated open areas has resulted in erosion of canyon hillsides, primarily in upper Jawbone, Bird Spring, Horse and Sage Canyons.[87]

Even then, the area between the two areas was considered a de-facto play area because of new illegal routes.[88] "Many trails and dirt roads have been made which are not necessary for general travel and access. These areas will remain void of vegetation for years due to compaction and erosion of topsoil," BLM managers wrote.[89]

Recommendations

• The BLM should fence all springs, leaving at least a .25 mile buffer around the springs, by the end of 2001.

• Off-road vehicle trespass onto the Pacific Crest Trail should be prevented.

• Illegal routes outside of open areas should be revegetated.

• Only a minimal route network (no more than 18 miles per township) outside open areas should be kept.

• Areas with excessive damage should be immediately closed and rehabilitated to prevent future damage.

• Greater law enforcement is needed to enforce regulations outside of open riding areas.

• Within the Jawbone-Butterbredt Springs Area of Critical Environmental Concern, riding should be limited to designated trails only.

Howard Wilshire

The visible ruts and gouges at Jawbone Canyon may never disappear.

PENINSULAR BIGHORN SHEEP
(*Ovis canadensis cremnobates*)

Status: Endangered

Mass hunting of Peninsular bighorn began in the 1800s and initiated the decline of the species. Disease, spread from domestic livestock, hastened the decline.[90] Development, roads, and mines reduced habitat. Off-road vehicle recreation created further disturbance in the sheep's limited range in California's Peninsular Ranges. Today, fewer than 500 bighorns cling to life in the arid terrain that stretches from Palm Springs to the Mexican border.[91]

The illegally created Dunn Road in the northern Santa Rosa Mountains near Palm Springs is a top threat to this species. Continual off-road vehicle use of this road stresses sheep during lambing and water stress seasons.

Off-road use by the U.S. border patrol and other off-road vehicle use close to the U.S.-Mexico border violates the Jacumba Wilderness and harms sheep.

Off-road vehicles continue to threaten the sheep's survival. Bighorn have been known to abandon areas after human disturbance began.[92] In one study, bighorn visits to a water source decreased nearly 50 percent on days when vehicles were used in the area.[93] Sheep can acclimate to only low numbers of lightly used routes and dispersed recreational impacts.[94] The closer the approach by humans or vehicles, the more severe the sheep's response.[95]

Protecting the bighorn will require strong intervention.But, just as we brought the species to the brink of extinction, so too can we bring it back.

Destroying an American Heritage

The California desert holds rare archaeological resources found nowhere else in the world. Prehistoric art forms known as geoglyphs were etched or tamped into the desert floor by indigenous people thousands of years ago. These giant-sized formations were made by aligning stones or turning over rocks to form elaborate designs.

The shapes of human figures, horses, serpents and other animals are best viewed from the air, since they are large images built by native peoples as messages to their spiritual world. One of the oldest geoglyphs dates back 2,600 years. A more recent example is an abstract of a horse that may date from the time of the Spanish Anza expedition of 1776.[96]

Sadly, hundreds if not thousands of such archaeological sites have been impacted by off-road vehicles on public lands in California.[97] Many of these sites are found in a two-by-three mile stretch of uplifted desert southwest of El Centro. This area alone contains 21 of these rare manifestations, most of which have no physical protection.[98]

Geoglyphs, petroglyphs and other remains of ancient people are spread throughout the desert. In fact, several off-road vehicle open areas are known to contain important archaeological sites, including Dove Springs, El Paso Mountains, Stoddard Valley and Johnson Valley.[99] None of these areas has ever been systematically inventoried for cultural resources, even though known sites exist that need stewardship.[100]

Instead, throughout the desert, prehistoric remains have been "run over and ridden through, and tires have been spun on them, causing soil erosion, surface soil displacement, vegetation loss, and the loss of scientifically important stratigraphy

National Geographic Society

Blythe archaeological site, 1932

Howard Wilshire

Blythe archaeological site, 1975

Reckless ORV use has severely damaged the ancient geoglyphs. Numerous other archaeological treasures have suffered a similar fate.

associated with the sites," according to the BLM.[101] The agency also reports that "fragile desert pavement surfaces have been cracked, causing the surface archaeology to be diminished or lost. In some instances, off-road vehicles have provided easy access for illegal artifact hunting, vandalism, theft of rock art panels, and even grave robbing."[102]

Aerial photos taken by BLM document off-road vehicle routes criss-crossing and circling the unique rock art. Fences and other barriers either have not been installed, or are destroyed or ignored by dirt bike and other off-road vehicle enthusiasts. For instance, the BLM noted that on one Memorial Day weekend, two motorcycles were lifted over a fence at the Yuha Maze site.[103]

And damage by off-road vehicle users has been exacerbated in recent years by higher overall visitation to the desert and increased border patrol activity.[104] Many new roads, trails and hill climbs near the geoglyph sites have increased.

Following is a sample of some of the sites that have been damaged:

South Panamint Valley Geoglyphs

Of more than 200 known geoglyphs in the southern Panamint Valley, more than 30 located near an access road suffered damage due to "driving on the pavement and spinning donuts." The road has cut through some of the geoglyphs.

Barstow Area Desert Tortoise Geoglyph

This interesting, small geoglyph is in danger of being overrun by vehicles due to increased visitation. The figure was created by removing desert pavement to form a circle with radiating spokes.

Throughout the desert, prehistoric remains have been "run over and ridden through, and tires have been spun on them, causing soil erosion, surface soil displacement, vegetation loss, and the loss of scientifically important stratigraphy associated with the sites."

Shoshone Area Amargosa Mystery Ring

The formation of 29 rings created by scraping the desert pavement is threatened by increased dirt bike and other off-road vehicle use, which has already impacted a portion of the site.

The Maze near Shoshone

This archaeological site is comprised of a series of 67 mound circles spiraling into a central maze. It is a fragile prehistoric resource, yet is encircled by vehicle tracks.

Mule Canyon Geoglyph

In 1984, a fence was constructed to keep the public from driving over the trail, geoglyphs and dance circle. But a portion of the site that was not fenced continues to be driven over. Eventually this portion of the site will be obliterated.[105]

The BLM is attempting to protect some rare sites with new monitoring, protection and planning programs. The agency is recruiting volunteers to help in these efforts. But lack of law enforcement and the high numbers of off-road vehicles make such intentions difficult, if not impossible to fulfill. It takes only one careless all-terrain vehicle to cause irreparable damage to a cultural site that has existed for thousands of years.

Recommendations

- All archaeological sites should be off-limits to access by off-road vehicles, and law enforcement should be increased in these sites.
- Fines for damaging archaeological sites should be increased, and imposed on violators.
- Additional fencing and bigger, more effective barriers should be installed.
- Sites should not be publicized.
- Cross-country riding should be eliminated, and riding should be restricted to designated routes where archaeological surveys have been completed.

"Fragile desert pavement surfaces have been cracked, causing the surface archaeology to be diminished or lost. In some instances, off-road vehicles have provided easy access for illegal artifact hunting, vandalism, theft of rock art panels, and even grave robbing."

Southern California's Urban Interface

Where Sprawl and Open Land Collide

Southern California's forests have both the highest concentrations of people and endangered species in the United States.[106] This unique mix of people and wildlife poses special management problems.

In some areas, endangered species are severely impacted by unmanaged off-road vehicle use. In others, conflicts between off-road users and other recreationists abound.

Conflicts between residents and off-road vehicle users occur on lower slopes of the mountains that face the Los Angeles Basin, in towns like Hesperia and Loma Linda that border open lands.

A recent survey of Hesperia residents found that off-road use was a top law enforcement issue.[107] The residents said their main concerns were noise, dust, destruction of the desert landscape, fire danger and trespassing.[108]

In one area, large groups of off-roaders regularly dump trash onto the desert floor, race across private property, and trample the sage and chaparral.[109] In another, dust from off-roading sometimes gets so thick that brown clouds of dirt obstruct the view of passing drivers on Highway 395.[110]

Further south, the town of Loma Linda manages 790 acres of undeveloped land between Riverside and Redlands. Here off-road motorcycles tear around on rugged, unroaded terrain with very little policing.[111] Some irresponsible off-road vehicle users threaten hikers and mountain bikers by driving out of control. Dirt bikes and other off-road vehicles also pose a very real fire danger due to dry grass and foliage.

Hesperia and Loma Linda are just two of the hundreds of communities across California that must deal, on a daily basis, with the effects and threats posed by irresponsible off-road vehicle users.

SAN BERNARDINO NATIONAL FOREST

Hello Dirt Bikes, Good-bye Hikers

- Counties: San Bernardino, Riverside
- United State Forest Service, San Bernardino
- Off-Road Vehicle Designation: Designated routes
- Issues/Concerns: Route proliferation, user conflicts, endangered species
- Key species: southwestern arroyo toad (endangered), desert tortoise (endangered), Southwestern willow flycatcher (endangered), Least Bell's vireo (endangered)

Located 60 miles east of Los Angeles, the San Bernardino National Forest rests at the urban fringe. Between 12,000 and 15,000 off-roaders take to this forest each year.[112] Irresponsible off-road vehicle users create conflict with homeowners and other trail users by traveling off designated routes and adding to the fire danger in summer months.

Community conflicts

In the heart of the forest's upper elevations (the San Bernardino mountains), people like John Henderson must endure the constant roar of off-road vehicles behind their homes. Henderson and his wife moved to a quiet residence near Lake Arrowhead 23 years ago. But ever since the Forest Service designated a trail at their back door as motorcycle route, the silence is no more.

"There is a hill they climb right behind us," Henderson explained. "The noise level is intolerable. They come by in packs. I have written letters to the Forest Service, but never got any answer." Not only is the sound shattering the Hendersons' peace, but they can no longer walk the trails because of rocks and deep ruts.

"We are being denied access. And that's what we moved here for. The trails go from three feet wide to 10 or 15 feet wide in many places," Henderson said. He has also witnessed off-roaders charging through the brush, crushing and ripping out plants. He's seen riders drop cigarette butts onto the dry forest floor, risking fire in an area that is still recovering from a devastating 1999 blaze.

Route proliferation throughout the forest

The Forest Service classifies off-road vehicle route proliferation as a continuing problem.[113] Dirt bikes and other off-road vehicles are limited to designated trails and roads, but this does not seem to deter drivers from entering many areas of the forest.

"I am more likely to encounter vehicles on locations prohibited to them than on a trail or road designated for them," said Tom Walsh of Blue Jay.

He described one section of trail as a "linear badlands." This stretch of trail between Deep Creek and Tent Peg campground "has many gullies and ditches; in some places it has been widened by use to well over 50 feet. There are many short undesignated routes which parallel and connect with it. There is considerable evidence of riding on road banks parallel to the road and of the use of high road banks as hill climbs."

On another trail near the North Shore campground at Lake Arrowhead, campers try to take walks on trails heavily used by motorcycles. Along one section the trail is so badly eroded that has been transformed into a trench with steep sides.

Dozens of other forest areas experience these and other problems. In Crowder Canyon, trespass into archaeological and paleontological sites is common, as is vandalism of locked gates and illegal driving in washes. Some irresponsible users travel illegally on state and county roads and create new routes in the Summit area. Along Cleghorn Ridge, non-street-legal vehicles travel on county roads en route to the Pilot Rock off-road vehicle area. At Marshall Peak, new trails are destroying habitat and causing erosion.[114]

The Willow Fire and off-road riding

The potential fire danger posed by off-road vehicles was made real in August, 1999 when a blaze broke out near Willow Creek, burning 63,000 acres.[115] Several homes and dozens of vehicles were burned. The Willow Fire "was strongly suspected to be from OHV riders leaving an unattended campfire."[116] The area had been open to off-road vehicles on designated trails only.

After the fire, the singed lands were closed to off-road vehicle use. But the burn exposed many areas that had been previously buried under thick chaparral, enticing riders to return.[117] The Forest Service proposed to spend $300,000 to keep irresponsible off-road users out of the area; no other group of recreational users required such an expenditure to control where they went.[118] *Not only did irrespnsible off-roaders likely cause the fire, they are doing more damage by riding illegally on the burned ground, and state funds are being requested to stop them.*

Recommendations

- The San Bernardino National Forest should close any routes or trails that negatively affect endangered or threatened species to off-road vehicles.
- The forest should close trails to motorized vehicles if the trails are near to homes disturbed by off-road vehicle noise, dust, gas fumes and other pollution.
- The fire danger of off-road vehicles must be considered in management plans for the forest.
- Trails and routes in poor condition should be closed and repaired.
- The designated route system must be enforced and penalties for illegal riding issued.

"Since the advent of off-highway vehicles and specifically motorcycles and ATVs on our mountain trails in the San Bernardino National Forest, our quality of life has been dramatically degraded. Our residence is located contiguous with the forest. We have lived at this location for nearly 23 years; long before the off-road activity became a reality.

"For years our family has enjoyed forest walks. However, this enjoyment has recently been altered because of the off-road vehicle activity. As seniors, we can no longer enjoy hiking here because of loose rocks and destroyed trail. Without question, our forest is being desecrated and scarred by this off-road vehicle activity."

—John Henderson, Lake Arrowhead

The Sierra Nevada: Backbone of California

The Sierra Nevada stretches 400 miles, from the California desert in the south to just below Lassen Peak in the north. The sequoias, Yosemite and Lake Tahoe immediately come to mind when picturing the Sierra. The "Range of Light," as John Muir called these great mountains, contains millions of acres of national forest and wildlands.

Hundreds of rare and unusual plants, animals and fish populate the Sierra. Black bear, bobcat and mountain lion thrive. Little-known carnivores like the American marten, Pacific fisher, Sierra Nevada red fox and even the scarce and reclusive wolverine are known to still hang on in remote corners of this magnificent range. And some, like the California spotted owl, mountain yellow-legged frog and Yosemite toad may be facing extinction.[119]

While the national parks and wilderness areas are off-limits to motorized recreation, much of the national forests are not. Motorcycles and all-terrain vehicles tear up alpine meadows in open riding areas and closed areas. Snowmobiles run cross-country skiers out of favorite spots and illegally venture far into remote, wild areas.

Following are examples of a few of the places in the Sierra Nevada where dirt bike and other off-road vehicle use is taking a toll.

LAKE TAHOE BASIN

Tracks across a National Treasure

• Counties: Placer, El Dorado, Alpine

• United States Forest Service, Lake Tahoe Basin Management Unit

• Off-Road Vehicle Designation: Limited Use-designated routes only

• Issues/Concerns: environmental damage, off-route riding, landowner and user conflicts

• Key species: California spotted owl, Northern goshawk, marten

The Lake Tahoe Basin is a popular and sensitive watershed that attracts 22 million visitors each year to its lakeshore and mountains. Situated at 6,225 feet, Lake Tahoe straddles the crest of the Sierra Nevada. Drastic declines in Lake Tahoe's water quality led President Clinton to make protecting the 300,000-acre watershed and the clarity of Lake Tahoe a national priority.

One of the many steps taken to reverse the lake's decline was the banning of polluting two-stroke jet skis. These thrill crafts dump as much as one-third of their fuel unburned into the waters where they are driven.[120]

At the same time, the Lake Tahoe Basin has experienced a tremendous surge in both off-road vehicle and snowmobile use.[121] Both dirt bikes and snowmobiles contain two-stroke engines that contribute to water and air pollution and cause environmental damage similar to the jet skis. Yet, use by dirt bikes, snowmobiles, and other off-road vehicles is allowed and even encouraged by the U. S. Forest Service with state funding.

In 1998-99, the Lake Tahoe Basin Management Unit documented 12,000 off-road vehicle visitors, not including snowmobiles. Due to the recent popularity of off-road motorcycles and four-wheel all terrain vehicles in the Lake Tahoe Basin, forest managers are reporting illegal off-road vehicle activities, including:

• widespread trespass beyond designated route systems,

• increases in the number of complaints about riding close to or inside residential areas,

• trespass into closed areas and wilderness,

• destruction of wet meadows and other highly sensitive areas,

• protected areas being used as mini-moto-cross parks, and

• new trails being cut into well-healed rehabilitated trails and roads.[122]

More than 400 miles of legally built and illegal, user-created routes and an additional 200 miles of off-road vehicle trails traverse the Lake Tahoe Basin. The unplanned, user-created routes often cross meadows and streams and other environmentally sensitive areas. These unsurfaced roads are considered to be a primary cause of water pollution, sending fine particles of soil and polluting elements such as phosphorus downstream toward Lake Tahoe.[123]

Damage by dirt bikes, snowmobiles, and other off-road vehicles to roads and forest land has been documented by the Forest Service, residents and visiting photographers along the McKinney-Rubicon Trail, Twin Peaks off-road vehicle staging area and trails, King's Beach off-road vehicle trails, and various Forest Service roads.

According to the Forest Service, in the Twin Peaks area, dirt bike and off-road vehicle users have expanded the network of trails beyond designated routes and have created soil disturbance on steep, highly erodable areas.[124] New routes have caused vegetation damage, soil displacement and degradation of the natural features.

Route widening, destruction of meadows and riding in closed areas have also been documented on the McKinney-Rubicon Trail.

Using snowmobiles, areas that were previously inaccessible due to terrain and topography are now within reach. An example is Freel Peak in the Lake Tahoe Basin. This 10,881-foot mountain on the lake's southeast shore is within a designated non-motorized area. In the past few years, snowmobile tracks have appeared higher and higher on the mountain where tracks had never been before.

Not only is this illegal, but it may be having an impact on a sensitive alpine plant community, the Freel Peak cushion plant community. The problem is magnified because of a lack of law enforcement in these areas.[125]

Since 1985, the Lake Tahoe Basin Management Unit has received $1 million in state funds to manage off-road vehicle use. The funding was used for law enforcement, public education and monitoring. The Forest Service cites decreases in funding since 1995 as the reason for recurring problems due to increased off-road vehicle use in the Lake Tahoe Basin. The agency has requested $144,000 in 2001 to conduct additional trail maintenance, law enforcement, monitoring and public outreach.

Recommendations

• The Forest should designate and enforce an off-road vehicle route network. Routes should be signed and citations issued for non-compliance.

• This network should be established through complete environmental analysis, and should take into account impacts on Lake Tahoe's clarity.

Jim Rose

This is the Saxon Creek area, in the Lake Tahoe Basin Management Unit. Like so many sites throughout California, Saxon Creek has been degraded by excess and indiscriminate dirt bike and other off-road vehicle use.

"On May 24, 2000, a dirt biker destroyed a pristine meadow on the Miller Lake Road, driving like a maniac in circles, ripping up a watershed area. The Forest Service did some major improvements a few years ago. Big stones were put down to stop vehicles from entering certain areas. This particular area was nothing but a dry dusty circle in the summer. But since it was blocked off, we had seen it slowly come back to life. This spring it was a restored beautiful meadow full of new life. We witnessed this area being deepened, carved into over and over again in a few minutes.

"This destruction, as well as the littering with beer cans by drunken 4-wheelers, is becoming a real problem in our area...."

—John and Birgitta McCarthy, Tahoma

Loving It to Death

• County: Placer

• United States Forest Service, Lake Tahoe Basin Management Unit, Lahontan Regional Water Quality Control Board, Placer County

• Off-Road Vehicle Designation: Designated Route

• Issues/Concerns: Soil erosion, route expansion, destruction of meadows

The McKinney-Rubicon Trail was once a Native American footpath, then later utilized by trappers, explorers and others as a route through the Sierra Nevada. Today, the trail draws thousands of jeep drivers and off-road vehicle enthusiasts from all over the world who want to maneuver the narrow, rocky trail.

The trail runs through an important watershed and sensitive forest land that cannot easily hold up to giant tires and endless trampling. All-terrain and other off-road vehicles are causing soil erosion and sedimentation by grinding up the surface soil, driving deep ruts into the trail and creating new, unauthorized vehicle routes into the forest.

More than 400 miles of legally built and illegal, user-created routes and an additional 200 miles of off-road vehicle trails traverse the Lake Tahoe Basin. These unsurfaced roads are considered to be a primary cause of water pollution, sending fine particles of soil downstream toward Lake Tahoe.

Riding starts early in the season while the ground is wet and muddy in many areas. As a result, stream crossings and low areas become muddy bogs. Some drivers just go around, widening the trails or intentionally creating new ones.

The problems date back 20 years or more. In the 1980s, water bars were installed along the trail, rocks and barriers put in place, and other preventive measures taken to reduce erosion.[126] But after seasons of heavy use and extreme winters, soil erosion became so bad that in 1994, the water quality agency responsible for the area required that Placer County (which owns most of the road itself) to immediately stop the discharge of tons of soil and sediment into McKinney Creek.[127]

In response, the county paved about two miles of trail along the creek. This greatly reduced the unlawful deposits, but also made the area more accessible to dirt bike and other off-road vehicle riders. So the problems continued.

After revisiting the trail in fall, 2000 to view more recent damage, regional water quality officials required the county to take immediate action, charging that it had violated previous, legally binding agreements to control and repair the environmental problems.[128]

The most recent damage documented includes:

• severe erosion of tons of sediment, which is washing into Lake Tahoe,

• new spurs being formed off the main road,

• jeeps and trucks being driven into small lakes for cleaning,

• sensitive meadows being destroyed,

• vehicle use on closed and obliterated roads,

• soil damage, encouraged by Placer County's early plowing of the access road,

• four-feet-deep chasms in the road due to soil loss and quarrying, and

• significant road widening.[129]

The water quality officials are now requiring that Placer County come up with plans (such as no snow removal on the paved areas, and seasonal road closures) to reduce off-road use during wet conditions. They have also mandated that the county develop plans to stop the erosion, run-off and damage to wetlands.

Placer County has requested $20,000 in state funding to begin to address some of the Rubicon Trail's problems. The application cited severe erosion near Lily Lake and rutting and soil erosion at several other locations along the trail. However, the grant also calls for early season snow removal, which leads to off-road vehicle travel in wet conditions. By plowing the road, the problems are only perpetuated.

Recommendations

The McKinney-Rubicon Trail should be closed seasonally during wet weather, and snow plowing should be delayed in order to prevent entrance into the area until conditions are drier.

Non-Management at its Worst

- County: El Dorado
- El Dorado National Forest, Georgetown District
- Off-Road Vehicle Designation: Limited Use-designated routes only-*Designated without analysis*
- Issues/Concerns: Soil erosion, wildlife habitat destruction, user conflicts
- Key species: Pacific mule deer, California spotted owl, Northern goshawk, native trout

The Eldorado National Forest, in the heart of the Sierra Nevada mountain range, is a popular area for both wildlife and people. The forest is home to numerous imperiled species, including the California spotted owl, goshawk, native trout, mule deer and marten. The area is also well-frequented by recreationists year-round.

Unfortunately, the Eldorado is also a case study in bureaucratic inability to deal with the impacts caused by off-road vehicle use.

In 1990, forest managers published an off-road vehicle travel plan, a 2-inch thick compilation of maps provided by dirt bike and other off-road vehicle users.[130] Unfortunately, the plan was completed without any of the required environmental analysis. In 1995, the Eldorado was ordered by its Washington headquarters to complete site-specific environmental review of its off-road vehicle plan by May of 1997.[131] As of January 2001, that review has yet to be initiated. The Eldorado blames a lack of funding, although it has received millions of dollars in state off-road vehicle grants since the mid-1980s.

The Georgetown Ranger District was ordered by a federal judge in 1988 to do an environmental impact statement (EIS) for the Rock Creek off-road vehicle area. The document was completed eleven years later, and was promptly appealed by local environmentalists. As with the forestwide off-road vehicle trail system, off-roaders defined the trails system at Rock Creek. The California Enduro Riders Association (CERA), which has sponsored 100-mile dirt bike races in the area since the early 1980s, demanded that the system maintain at least 100 miles of trail for their enduros. The former director of the state's OHV Division, Cliff Glidden, was a member of CERA and threatened the district with a loss of funding unless the concerns of CERA were met.[132]

The Rock Creek plan included criteria for closing trails when a storm produces over 1.5 inches of rain, despite evidence that watershed damage occurs at a much lower level of precipitation. In the spring of 2000, CERA's enduro began in a downpour, after an overnight storm dumped .62 inches of rain on the steep trails. By the end of the race, over an inch of rain had fallen.[133] Damage to the trails was so extensive that three trails were subsequently closed to all uses until repairs could be made.[134]

A few months later, over the Veterans Day weekend, the Rock Creek trails remained open despite well over 1.5 inches of precipitation because the district off-road vehicles manager didn't want to inconvenience dirt bikers who had come to spend the three-day weekend.

To provide the 100-plus miles of trail, the 1999 Rock Creek decision opens three-quarters of a 10,000-acre critical deer winter range to winter off-road vehicle activity, under the guise of a study. The study claims to


Jim Rose

Many motorized vehicle routes are touted as "multiple-use" routes, where hikers and other non-motorized recreationists can "share the trail" with ATVs and other off-road vehicles. Such sharing can often be quite difficult.

examine whether vegetation management can mitigate the impacts of off-road vehicles on wintering deer. The Forest Service's own senior biologist, asked by the EIS team to review the proposed study, responded that lifting the winter closure was "ill-advised" and that the proposed monitoring plan was "useless."[135] The district ranger ignored this recommendation.

In lifting the winter closure, the Forest Service also ended protection for the only reproducing spotted owl pair in the 23,000-acre off-road vehicle area. That owl pair could not

Mimi Jennick

The Mammoth Bar moto-cross track, just north of the Eldorado National Forest, lies directly adjacent to a potential Wild and Scenic River.

be detected in the spring of 2000, the first season after the closure was lifted. California spotted owls are declining at a rate of 7% per year and the species is a candidate for listing under the federal Endangered Species Act.[136]

According to studies, there are no amphibians found in the Rock Creek area, despite 23 named streams in the 23,000-acre watershed.

The Rock Creek trail system designates about 112 miles of off-road vehicle routes, with only 14 miles reserved for all non-motorized recreationists. The plan is expected to increase dirt bike and other off-road vehicle use considerably, at the expense of hikers, equestrians and wildlife.

Further, road/trail density is more than double the density recommended for wildlife by the California Department of Fish and Game. The Forest Service ignored that recommendation.[137]

Finally, despite high levels of illegal riding, law enforcement in the Rock Creek area is almost nonexistent. Residents and recreationists who observe illegal off-road vehicle activity are instructed by the Forest Service to call 911.

The result of this bureaucratic fiasco is that dirt bike and other off-road vehicle use continues to dominate the trails in the Eldorado, and at Rock Creek in particular, and other recreationists as well as wildlife continue to get short shrift.

Recommendations

• The Eldorado National Forest should complete a forest-wide environmental analysis of its trail system. Until that is completed, the state should provide grant funding only for the analysis, closure of damaged areas and trails, and law enforcement.

At Rock Creek:

• The critical deer winter range should be closed from November through May, and spotted owl activity centers should be closed February through August. Further, competitive events should be eliminated or restricted to areas and times of the year when such events will not conflict with wildlife needs.

• Trails should be closed when use on wet soils would cause damage to trails or streams. Use should not be allowed when rainfall exceeds .75 inches.

• Road/trail densities should be consistent with those recommended by wildlife models.

• The Forest Service should ensure adequate law enforcement to protect residents and resources.

"In June, 2000, a group of hikers from South Lake Tahoe hiked the Caples Creek Trail.…Well, imagine our complete dismay and outrage when we got to Government Meadow (looked forward to as the high point of our hike because we anticipated the beautiful meadow full of shooting star wildflowers) and found that dirt bikers had completely ruined the meadow by riding in circles all over it.

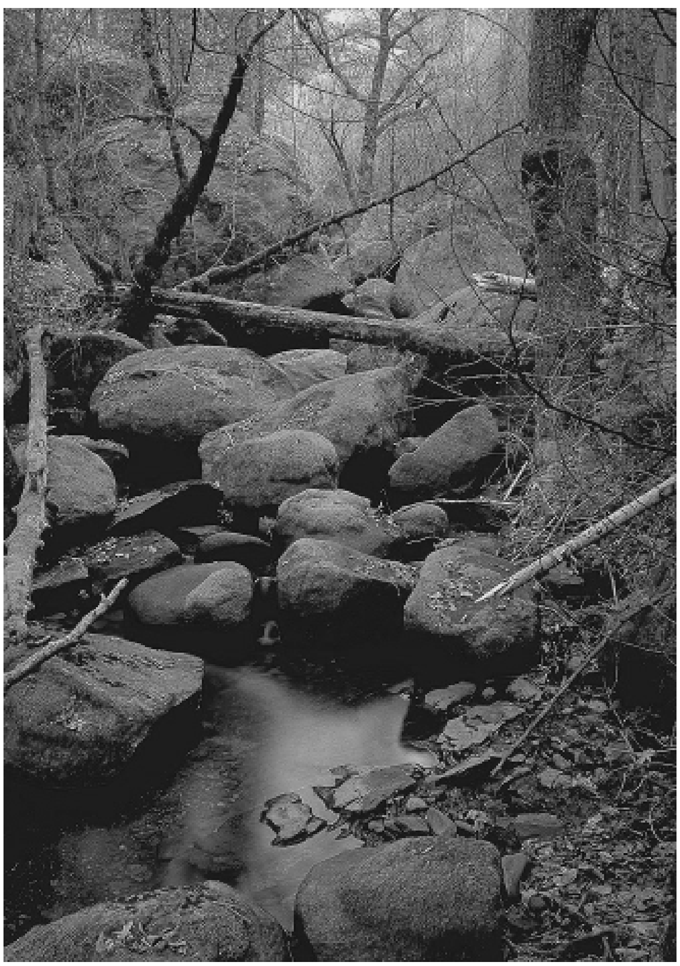
"The meadow was wet and muddy and the tracks left in it by the bikers will probably be there for decades. In fact, because of the large number of ruts, I believe that the meadow may never be the same again. We were all quite sickened by what we saw here."

—Gay Havens, South Lake Tahoe

Face Off in Stanislaus

Frank Rauchschwalbe

"My home and refuge is no longer a sanctuary," says local resident Judith Spencer. Dirt bikes regularly tear past Spencer's home, creating illegal routes, harassing wildlife, and irritating residents and visitors.



Chris Hall

The Interface

- County: Calaveras
- Stanislaus National Forest, Calaveras District
- Off-Road Vehicle Designation: Limited Use-designated routes only-None Designated
- Issues/Concerns: Homeowner conflicts, route proliferation, environmental damage
- Key species: California red-legged frog (endangered)

After three years and three different plans, homeowners who live on the edge of the Stanislaus National Forest near Arnold are ready for a final decision about off-road vehicle use in their community. Most want riding, at the very least, limited to the north section of the 8,000-acre finger of forest called the Interface. The two-by-eight mile stretch is surrounded by five subdivisions with hundreds of homes, a 500-unit private campground and three large ranches.

But the latest Forest Service trails proposal—the third in so many years— would allow dirt bikes and all-terrain vehicles to continue to use the entire length of the Interface, to the dismay of most residents. The Stanislaus has received $14 million in state off-road vehicle grants since 1988, and some residents fear that the reliance on state funding has impaired the Forest Service's judgment regarding the area's management.

"My home and refuge is no longer a sanctuary," says

Two-stroke engines, like those found on many motorcycles and snowmobiles, are notorious for spilling one-third of their gas-and-oil mixture of fuel unburned into the environment.

Dirt bikes and other off-road vehicles can also leak fuel, oil, anti-freeze and other toxic chemicals into the streams and creeks, creating a potential source of ground water pollution.

Judith Spencer, who responded to the Forest Service's pro-off-road vehicle policies by joining with others in the community to form Commitment to Our Recreational Environment (CORE). "This is my life and my home, and the homes of my neighbors. It's a very difficult situation. The place where I would seek peace is in the woods. And that's where I'd get run over."

When Spencer and other property owners first moved here, as long as 35 years ago, the area was relatively quiet and remote. Today the Interface buzzes with the noise of off-roaders roaring up hillsides and across creeks. Since 1991, the 18-mile route system has mushroomed to over 100 miles of user-created, unauthorized trails.[138] Out-of-control riding is exposing the neighborhood to constant noise, exhaust smoke, dust, and danger when motorcycles or ATVs rip down local streets. Law enforcement both on and off the dirt trails has been lacking.

The off-roaders also damage fragile plants such as the endangered Whipple's monkey flower and disturb wintering black-tailed deer herds. The area contains potential habitat for the nearly extinct red-legged frog (known as the Calaveras jumping frog). The proposed trails run through important habitat for the California spotted owl and across creeks that have already been identified as erosion and silt problem zones.

In a letter to the Forest Service, the California Department of Fish and Game expressed concern about the degradation of wildlife corridors and habitat. The Department of Fish and Game owns adjacent protected lands that could be impacted by heavy off-road vehicle use in the Interface. Potential re-introduction of the California red-legged frog in the Interface could also be hampered by off-road vehicle use.[139]

Spencer and CORE hope to spare the neighborhood and the native wildlife from increasing off-road vehicle use by relocating dirt bike and other off-road vehicle use out of the Interface, or, at a minimum, supporting designated routes in the northern section only.

Of the more than 300 public comments on the last proposed trails plan, only seven percent favored off-road vehicle use throughout the Interface. Another 35% favored limited use north of San Antonio Creek, but most (46%) called for complete elimination of off-road vehicles. However, the Forest Service never offered this alternative.

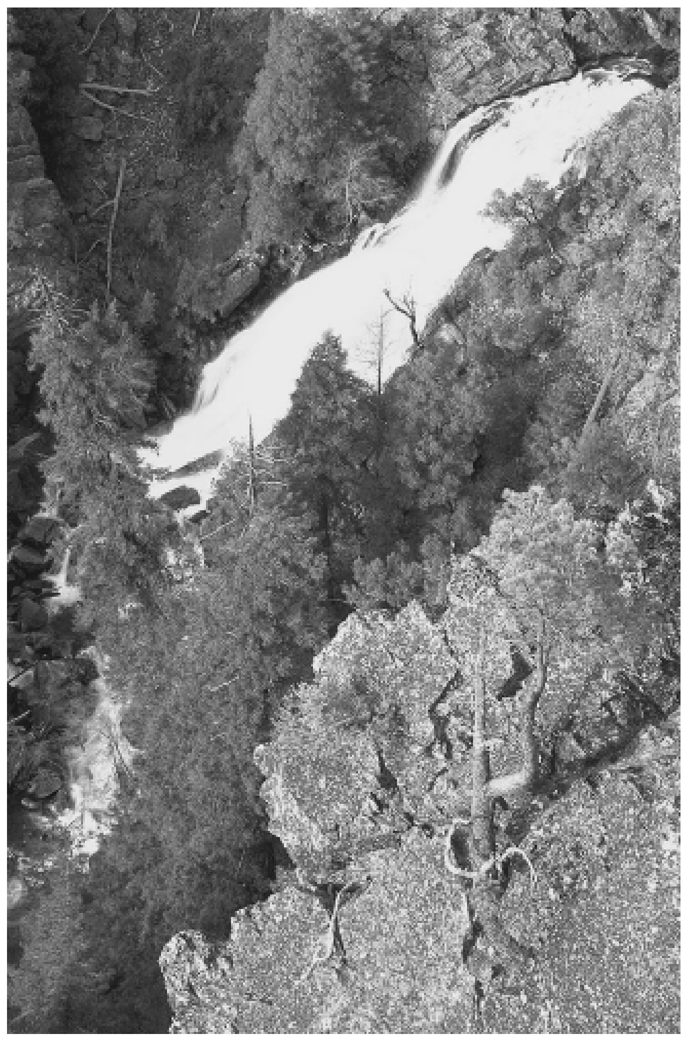
"If this plan is approved, there is nowhere in the Interface that a person could take a quiet walk, go bird-watching or fishing without being in earshot of motors," Spencer says.

The latest Interface Recreation Trail Plan calls for building 10 miles of multiple-use trails and 11 miles of non-motorized trails. It would designate 37 miles of existing routes for motorized use, and 17 for non-motorized. Another 26.5 miles would be closed. The U.S. Forest Service officials said they would make a decision by summer of 2001, though it could take five years to build new trails and close existing ones.[140]

Recommendations

• A trails plan for the Interface area should be finalized and completed.

• Off-road vehicle use in the area should be limited to the northern section of the Interface only, or eliminated entirely.



Chris Hall

The refuge known as the Interface is being ravaged by out-of-control use by all-terrain and other off-road vehicles.

California's Coast Ranges

The range of coastal mountains that runs down the state's ocean side covers a diverse landscape that varies from thick chaparral to cool redwood stands. The majority of California residents live in these coastal ranges, and recreational demands continue to grow every year. Population growth and demand for recreation are making every square mile of public land more and more important.

Many of the public lands along the California coast receive consistent use from damaging off-road vehicles. Few Californians take part in this activity, yet the toll taken on the landscape is far-reaching.

Popular off-road areas in the Coast Ranges include Knoxville and Cow Mountain Recreation Areas, Mendocino National Forest, Los Padres National Forest, Hollister Hills State Vehicular Recreation Area, Clear Creek Recreation Area, and Ocean Dunes State Vehicular Area.

KNOXVILLE RECREATION AREA

Shooting and Riding in Rocksville

- Counties: Napa and Lake
- Bureau of Land Management, Clear Lake Resource Area, Ukiah District Office
- Off-Road Vehicle Designation: Limited Use-designated routes only-None Designated
- Issues/Concerns: Lawlessness, environmental damage, trespass
- Key species: Foothill yellow-legged frog, southwestern pond turtle, serpentine plants and animals

Susan Harrison

Severe erosion is the rule within the Knoxville ORV area. With little to no law enforcement presence, off-roaders tear up streams and seeps, denude hillsides, and destroy vegetation. This hill is one of many that have been affected.

For more than 15 years, the remote chaparral hillsides and serpentine barrens of Knoxville Recreation Area have been subjected to mostly unregulated motorcycle riding and four-wheeling, random target shooting, and rampant lawlessness.[141] Steep, rocky terrain dominates the 18,000-acre outpost, situated at the corner of Lake and Napa counties—between Lake Berryessa and the town of Lower Lake.[142]

Most visitors arrive via off-road vehicles.[143] Due to its remoteness and lack of policing, Knoxville has grown lawless, and even dangerous at times. Appliances, TVs, computers and other large items have been found blown up and riddled with bullet holes. Revving motors and flying bullets frighten wildlife and chase away campers and hikers. Most complaints from neighboring landowners and visitors call for increased law enforcement.[144] Long-time landowner Jim Erasmy warns women not to hike the area alone.[145] "You just never know," he says.[146]

The routes used in Knoxville were never designed for off-road recreation; they were built for mining gold, mercury and silver.[147] Some are over 100 years old. Other roads were later constructed for ranching, firebreaks and hunting. Dirt bikers and other off-road vehicle riders have since worn many new miles into the chaotic system, which is neither designated nor mapped for off-road recreation.

Without proper drainage or construction, tire tracks have deepened trails into rills and ruts that quickly carry topsoil away. It is so difficult to maneuver these trails even with a motorcycle or four-wheel-drive that off-roaders derisively call the area "Rocksville." Yet, the BLM has encouraged off-road vehicle use with annual state grants and plans to build more trails.[148]

The types of off-road vehicle-related problems regularly documented by the Bureau of Land Management at Knoxville include:

• trespass onto private land,

• illegal shooting,

• trampling of meadows, riparian areas and rare serpentine wetlands,

• accelerated erosion and soil compaction, and

• destruction of habitat for the foothill yellow-legged frog and southwestern pond turtle, both designated by the BLM as sensitive species.[149]

While the damage is extreme and ongoing, not everyone sees Knoxville as a sacrifice area for motorized madness. Biologist Susan Harrison of the University of California at Davis sees a rich natural resource deserving of special protections. The serpentine soils in Knoxville attract plants and animals that live in very few places.[150]

Several plants and butterflies, as well as a little-known ant, are among the creatures that rely on Knoxville's rare serpentine soils. The public lands are next to private lands that form the McLaughlin Natural Reserve, a study area for serpentine ecosystems. Harrison has proposed that Knoxville Recreation Area be given special new environmental designation.[151]

However, uncontrolled dirt bike and other off-road vehicle use continues to degrade the rare soils and natural systems in the Knoxville area. Most roads cut across serpentine areas, and two heavily used off-road "play areas" in highly erosive serpentine barrens have caused deep ruts that destroyed sensitive vegetation.[152]

Off-roaders trespassing into the neighboring McLaughlin Preserve have ruined numerous study sites.

A university researcher on a spring field trip was surprised by a truck full of "guys driving over the plants, with beers in one hand and guns in the other. These guys drove through a barbed wire fence and over a huge tree to get into the research plot, where rare plants grew."[153]

Until recently, BLM planned to build as many as 32 miles of new off-road vehicle trails through the Knoxville lands. But objections from neighbors and naturalists backed by tougher off-road vehicle grant conservation and monitoring requirements have caused the BLM to slow its course, at least temporarily. Instead, the BLM plans to map the existing routes, designate some for off-roading and build loops to connect dead-end trails. But Harrison and others believe that the problems do not stem from a lack of loop systems, but the lack of planning and enforcement. The loop plan may just be an attempt to extend the trail system and get additional state funds.

However, soil and wildlife monitoring is beginning. New managers also hope to curtail

Jim Rose

An unauthorized hill climb at Knoxville. Off-roaders frequently create unauthorized and illegal trails, causing extreme resource damage. Riders also regularly trespass onto adjacent private lands, harassing and disturbing local landowners.

shooting by designating a shooting range. The agency has also promised to assign more rangers and law enforcement officers to the region. It is unknown what the effects of these new management promises will be.

Recommendations

- Area managers should designate routes as open, limited, or closed to off-road vehicle use. Open routes should be effectively signed; closed routes should be rehabilitated.
- Law enforcement should be dramatically increased, and citations issued for trespass, and closed area violations.
- No new vehicle routes should be constructed until the area can be properly managed.
- Knoxville should be designated as an Area of Critical Environmental Concern, and managed accordingly.
- A new management plan should replace the outdated 1973 plan and the status of Knoxville should be reconsidered.
- The area should be closed to shooting. Furthermore, Knoxville should be closed to dirt bikes and other off-road vehicles if the lawlessness continues.

"Our family has owned property in the Knoxville area since 1940. Prior to 1970, seasonal hunters and an occasional camper shared the winding, single lane road with property owners. Devil's Head road is a county road that was bladed once a year. Even during wet winters, the road was usually passable….Devil's Head road has now become impassable during most of the winter months due to four wheelers and OHV users thrashing it for fun or what's commonly known as 'mud running.'

"Families come to the Knoxville area thinking that they have a safe place for their children to learn to ride. Instead they've entered an area with no rules, no road signs, and no real BLM presence. Due to a lack of funding, law enforcement for the most part has been non-existent….

"During the last 25 years the area has been taken over by a lawless element. Landowners have experienced an increase in trespass issues. Some OHV riders have been caught destroying gates and cutting fences in order to take short cuts….

"Another serious safety issue is the unlawful shooting across Devil's Head road. The risk of getting shot while riding off-road vehicles, hiking, horseback riding, working as a field researcher for U.C. Davis or in general just being in the area is a real and ever present danger. There have also been incidents of individuals shooting in the direction of private dwellings. Numerous trees have been blasted in half, and the general area has become a dumping ground for bullet-riddled computers and other trash. Explosives were even used to destroy a large "vandal proof " steel sign set in concrete.

"Through the years, my family and I have personally observed an increasing disregard for the environment, the safety of law-abiding citizens, and the rights of property owners. Until this changes, BLM should not be allowed to proceed with promoting Knoxville as an OHV area."

—Jim Erasmy, Middletown

Pouring Soil Down the Drain

• County: San Benito

• Bureau of Land Management, Hollister Resource Area

• Off-Road Vehicle Designation: Mostly limited use on existing routes, with several open play areas

• Issues/Concerns: soil erosion, trampling of endangered plants, route proliferation, health risks and trespass

• Key species: San Benito evening primrose (threatened), rayless layia (sensitive), foothill yellow-legged frog (sensitive) and western pond turtle (sensitive)

Many problems remain to be addressed in this large off-roading area in the interior coastal range east of King City. Soil erosion, route proliferation, species impacts, trespass, and health risks due to naturally occurring asbestos, all pose serious threats to the region's environment and visitors.

The San Benito evening primrose was driven to the brink of extinction in the serpentine-rich Clear Creek soils region due to rampant off-highway vehicle recreation. Two-thirds of the known growing spots for the small yellow-flowered annual herb have been trampled or otherwise adversely affected by off-roaders.[154]

Recently, Clear Creek managers began to fence off and protect primrose growing areas from the tracks of motorcycles and ATVs. With a number of growing areas now under care, the primrose probably now has a chance to survive.[155] The sensitive rayless layia (a wildflower), California red-legged frog, and western pond turtle are also receiving better protections from off-road vehicles in some areas.

Clear Creek's 50,000 acres of mostly mountainous terrain attract 30,000 to 40,000 visitors per year. Nearly all ride off-road vehicles. The terrain consists of forest, meadow and unusual barren lands associated with serpentine soils. These barren lands are also the source of natural asbestos. More than 800 miles of roads, trails and user-created routes exist in the Clear Creek area.[156] BLM recently designated a route network, but allows riding over *all* routes in the area, not just those designated. Management activities, however, are focused solely on the designated routes.

The results are predictable. Many routes within the area do not meet the standards for soil erosion jointly developed by state and federal officials in 1991. The Clear Creek watershed is extremely prone to erosion, and the excessive dirt bike and other off-road vehicle use (and lack of management) only exacerbates the problem.

About 160,000 tons of soil is lost each year, a level 45 percent above natural erosion rates for the area.[157] This primitive system of unpaved roads dumps a shocking 38,000 cubic yards per square mile of sediment into Clear Creek every year.[158] On hill climbs, the soil loss rate is 4.9 tons per acre per year; for the route network, 80.2 tons per acre per year.

It's clear that closing roads to dirt bikes and other off-road vehicles would result in immediate improvement in terms of soil erosion. But enforcement of the new route designation system remains voluntary.[159] In addition, all routes will remain open until inventoried and analyzed, which is expected to take years.[160] Wet weather closures would be the exception to the "open-until-closed" policy.

As a result, off-road vehicle riding continues throughout the system and even onto neighboring private lands and into the mines that still operate within Clear Creek. Dirt bikers and other off-road vehicle riders threaten their own health and that of others when kicking up asbestos dust during the dry season. Riders are given warnings and ride the area knowing the risks. An asbestos Superfund site is located within Clear Creek. Adjacent landowners have threatened to sue over the potential public health hazard.[161]

With rare flowers, soils, sensitive animals and the public health at risk from motorized recreation in Clear Creek, it's time that the agency take more aggressive action in addressing the region's chronic problems.

Recommendations

• The route designation process should be completed, and enforcement of open and closed routes made a high priority.

• Monitoring should be increased, and routes and open areas where damage has occurred or is occurring should be immediately closed and rehabilitated.

Dirt bikers and other off-road vehicle riders threaten their own health and that of others when kicking up asbestos dust during the dry season.

SAN BENITO EVENING PRIMROSE
(*Camissonia benitensis* Raven)

Status: Threatened

You wouldn't expect a delicate flower to thrive in the dry barrens of the Clear Creek area of San Benito County. Yet the San Benito evening primrose grows only in the serpentine soils that dominate this landscape. That the rare plant still survives is a marvel, as this area has long been a haven for unrestricted off-roading. In fact, off-road vehicle recreation is the number one threat to the primrose—and has been for at least 30 years.[162]

As of 1998, the plant was found in only 27 places, and nearly two-thirds of those spots had been trampled or otherwise affected by dirt bikes and other off-road vehicles.[163] The tons of erosion that hundreds of off-roaders let loose every year pose a big threat. The primrose prefers alluvial terraces created in stream banks by water flow and organic debris. When huge torrents of soil pour down from the hill climbs and slopes traveled by off-roaders, the plant gets buried.

When the open slopes above creek canyons are undisturbed, they are protected from erosion by the silt-clay crust that forms on serpentine soils. When the crust is disturbed, mass wasting occurs. Soil loss in the Clear Creek area caused by off-road disturbance has been 160,000 tons per year, 45 percent higher than the natural rate.[164]

Fences have been erected around primrose sites, but riders still venture into these areas and cause damage.[165] The future of the primrose lies perilously in the hands of the land managers and off-roaders who continue to put the flower in jeopardy.

Rugged Terrain & Off-Road Trouble

- Counties: several
- United State Forest Service, Los Padres National Forest
- Off-Road Vehicle Designation: Designated Routes
- Issues/Concerns: Route proliferation, endangered species, soil erosion
- Key species: Southwestern arroyo toad (endangered), California red-legged frog (endangered), California condor (endangered)

The two-million-acre Los Padres National Forest covers much of the southern, outer Coast Range, extending 200 miles from the western boundary of Los Angeles to the middle of Monterey County.[166] The diverse and rugged terrain varies from the steep ocean cliffs of Big Sur to the dry desert of the Cuyama Badlands.

Many portions of the Los Padres have suffered from excessive off-road vehicle use. The Mt. Pinos Ranger District in the central Los Padres has been used by off-roaders since the 1950s.[167] Off-roaders can travel on 350 miles of routes that cover more than half the forest.[168] Two intensely used off-road areas are situated along the desert-like eastern boundaries of the Mt. Pinos Ranger District— Ballinger Canyon and Hungry Valley State Vehicular Recreation Area.

Ballinger Canyon covers 7,000 acres north of Ventura in the Cayuma Valley off Highway 166 and inside the national forest's boundaries.[169] Hungry Valley is on state-owned land outside the national forest boundary at Gorman and Highway 5.

These off-road hotspots have

Howard Wilshire

Hungry Valley State Vehicular Recreation Area, photo taken in 1998. Deep slots have been cut by motorcycles and ATVs. The hill is capped by an ancient river gravel, which is now being "mined" by dirt bikes and other off-road vehicles. These are permanent damages that cannot be repaired.

been heavily scarred by more than two decades of hill climbing, trail carving and loss of soil, plants and wildlife. The endangered blunt-nosed leopard lizard once lived in Ballinger Canyon, but has since disappeared from the area.[170]

A soil loss of 7,300 tons per square mile each year was estimated on slopes near the Ballinger Canyon campground area after less than ten years of off-road use.[171] This extreme erosion caused layers of dirt to bury plants and animals below. Water coursing down the slopes carved deep gullies that increased the erosion rate to 86 times what federal soil guidelines allowed.[172] Off-roaders simply avoid these tire traps and create new trails, beginning another destructive cycle.

Land managers here were among the first to produce a trail map for off-roaders and designate routes.[173] They have closed hill climbs and reduced the number of routes. Yet off-roaders continue to violate the rules and trespass onto neighboring private property where forest rangers have no jurisdiction.

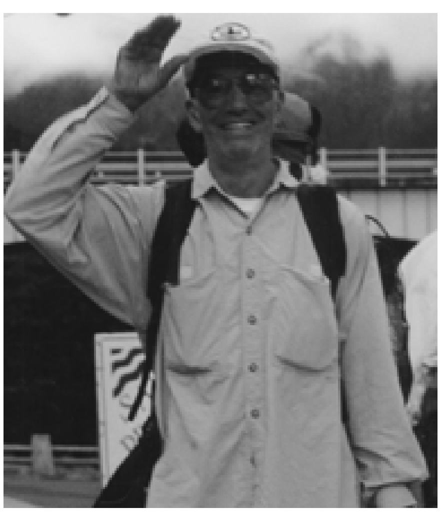
In Hungry Valley, more than 2,000 acres were stripped of grass and bushes, sliced with long, deep gullies and eroded by a maze of trails in about five years.[174] The severity of the soil erosion was demonstrated by the placement of benchmarks on the crest of a hill climb. Twenty-five inches of soil eroded from beneath the benchmarks over a period of eight years, freeing the once solidly buried metal stakes to roll down the hill.[175]

Between 1971 and 1978, the main channel draining Hungry Valley changed course as a result of off-road vehicle use and the resultant soil erosion. It was transformed from a shallow, braided, vegetated, somewhat insignificant channel into a 60-to-70 foot wide, six-foot deep culvert nearly devoid of plants.[176] In heavily used areas, major gullies have carried thousands of tons of soil downstream after one winter season.[177]

While restoration efforts have been made and some areas closed off, the damage continues. Hill climbing on closed areas remains a continuing problem at Ballinger

Howard Wilshire

Ballinger Canyon, photo taken in 1987, approximately 10 years after this hill was closed to dirt bikes and other off-road vehicles. Slots to 6 feet deep were cut by motorcycles and ensuing erosion.



"The worst experience I ever had with off-road vehicles was on the Pacific Crest Trail climbing out of the Mojave Desert and into the Tehachapis. I found that the trail was well used by motorcycles, even though it is closed to vehicles. In this section, there were extremely difficult trails, because the trail had split into six tracks. The motorcycles were coming up and making their own trails. You stood a very good chance of getting hit."

—***Tony de Bellis, Danville***

ARROYO TOAD
(*Bufo californicus*)

Status: Endangered

Only a few of California's streams still sing with the soft, high whistled trill of the arroyo toad.[182] Usually found in shallow pools and on sandy banks, about 75 percent of the buff-colored toad's favorite hang-outs are now gone.[183] Development, dams, water diversion, agriculture and recreation have taken over the many hidden places where the small toad once lived. The few that linger are protected by the Endangered Species Act. But they are not always safe from off-road vehicles.

Motorcycles and four-wheelers riding into the forests of central and southern California have wreaked havoc on the amphibians and their habitat, crushing toads and destroying breeding pools at sites where roads cross streams.[184] In 1991, a fence protecting a toad breeding pool was cut, giving off-road vehicles access to the creek. The vehicles destroyed a small sand bar that included a breeding pool used by at least 12,000 tadpoles.[185]

After a lawsuit forced action, the U. S. Forest Service prohibited off-road access in several toad-dwelling areas. In 1998, the Angeles National Forest closed off nearly five square miles of forest, including a campground and about 17 miles of off-road vehicle trails upstream from Littlerock Reservoir to help ensure the toad's survival.[186]

In June 2000, federal biologists proposed 500,000 acres as "critical habitat" for the toad. Things are looking up for the arroyo toad, though it remains very close to the brink of extinction.

Canyon and Hungry Valley and on nearby forest land. Riding off designated routes and blazing routes also remains a problem throughout the forest.

In the Frazier Mountain and Alamo Mountain areas near Hungry Valley, off-roaders routinely ignore road closure and damage gates, fences and natural features in the process.[178]

Near the communities of Frazier Park, Lake of the Woods and Pinyon Pines, there is a "high incidence of use on illegal routes and a proliferation of illegal, informal motorcross track." In the Badlands areas, off-roaders ride illegally up washes and often into the adjacent Chumash Wilderness Area.[179]

Further, conflicts between off-road vehicle users and local residents continue. One resident describes her experiences: "I have encountered dirt bikes on trails they were not supposed to be...On one instance, they were racing up and down a canyon. I was riding with a woman whose horse was very afraid of them. When they got to us, they stopped next to the uphill side of the road, engine running, and as my friend tried to talk to them and tell them to stop on the "cliff side" her horse tried to get the hell out of there, and went partly off the downhill side, but was able to get back up. It could've been worse.

"We just hope we can get well out of their way before they come and kill us. Luckily they are so damn loud, you can hear them way off, and get off the trail in time...."[180]

Fire danger

Off-roaders also present a fire danger. Los Padres National Forest is one of the most wildfire-prone forests in the entire national forest system, with an average annual acreage burned by wildfire in excess of 20,000 acres. The threat from off-roaders driving around the forest without approved spark arrestors is common.[181]

Recommendations

- The Los Padres National Forest should effectively monitor off-road vehicle damage and impacts on endangered species, close routes that are negatively impacting watersheds or species, and rehabilitate areas that have been degraded.
- The designated route system should be better enforced, especially in Ballinger Canyon.
- Law enforcement should be increased to prevent private property trespass.
- Environmental representatives should be included in any planning for new off-road routes.

Howard Wilshire

Forest Service officials recently noted extensive illegal riding "and a proliferation of an illegal, informal motorcross track" in the Los Padres National Forest.

The Great Basin: California's Northeast Corner

The Great Basin desert crosses into California north of the Sierra, in the northeast corner of the state. In this remote region, pronghorns still roam and bald eagles nest.[187] Sagebrush, grasses and juniper dominate the lowlands, while mahogany grows above 6,000 feet in the Fort Sage Mountains.

This diverse section of the state attracts birds, hikers, mountain bikers, hunters, anglers and off-road recreationists. Eagle Lake is a favorite destination. The Skedaddle, Five Springs, Tunnison Mountains, Twin Peaks and Buffalo Hills Wilderness Study Areas are also found in this region.

Most of the public land in the area north of Susanville near the California/Nevada border is managed by the Susanville District of the Bureau of Land Management.

For many years, off-roaders were allowed to ride cross-country throughout the district. Today, most of the land still remains largely open to cross-country riding off trails and roads, except in certain areas that are closed to or specifically designated for off-road vehicles. A few key areas are protected but most of the land continues to experience the environmental and social problems associated with off-road recreation.


Jim Rose

Two-track into the Skedaddle Wilderness Study Area. All-terrain vehicles and dirt bikes have chewed up hundreds of thousands of acres of California's pristine wildlands.

Eagle Lake

Eagle Lake is one of the most sensitive areas in the Susanville District. A closed basin with no natural outflow, it is California's second largest natural lake.[188] Bald eagles and ospreys nest along the lake. Anglers catch trophy-sized fish. These incredible natural assets are what make the lake very special to many Californians.

The mandate of the BLM is to "protect and enhance the environment of Eagle Lake for all generations." It is required to determine the "optimum" mix of uses to preserve the integrity of the lake.[189] The agency is also required to make sure that activities on the land within the lake basin do not pollute the lake.[190]

However, the only real change in off-road vehicle policy that has been made to help meet these guidelines was to shift off-road use in a few selected places from totally open to designated routes only. Dirt bikes and other off-road vehicles are still allowed to travel on designated routes throughout the Eagle Lake basin, causing erosion and pollutants to enter the lake.[191] The latest basin plan aims only to "reduce" the potential for increased surface runoff and associated soil erosion and nutrient transport into Eagle Lake from off-road vehicles.[192]

That approach falls short of the zero discharges required by the water quality agency. And it's unlikely that off-roaders will stay on the designated routes.

The Eagle Lake managers did take positive steps by closing several areas near Eagle Lake to off-road vehicle use, including sections of shoreline along Black Mountain, nearby Big and Little Troxel Points, Buck Point, and the roadless portion of Rocky Point.[193] Willow Canyon is also now off-limits to off-roaders. But the question remains whether they will stay out, and whether the BLM can enforce these policies.

Fort Sage Off-Road Vehicle Area

The Fort Sage off-road vehicle area is the region's most popular riding area. As many as 6,000 dirt bikes and other off-road vehicles visit the area each year to roar up hill climbs, spin donuts in sand, and skid across compacted desert. A designated route system was implemented in 1998. But the plan may have come too late for this battered terrain and, in particular, for the badlands area known as the "Land of David."

Pioneer tracks go off in every direction. One hill climb that has been left open in the Land of David faces another that does not appear on the approved route map. But that has not stopped anyone from criss-crossing the hill with tracks. Dirt bikers ride past closure signs and over the newly reseeded earth, defying the attempts of the land managers to restore this beautiful landscape.

Other problems that the off-roaders impose on the area include littering, trespass, vandalism, wildfire and the visual impacts of hillside scars.[194]

About 80 miles of motorcycles and trails were approved for riding on the nearly 23,000 acres of Fort Sage. It is difficult to estimate how many additional, illegal routes are there. The area is very rocky and subject to severe wind erosion, high run-off and gullying.[195] The landscape changes from dry brush and grass covering an ancient lake bed at the lower elevations of about 4,000 feet, to the towering ridges of the Fort Sage Mountains at 6,000 feet, which is the recharge area for the Honey Lake groundwater basin.[196]

And while the land may seem empty and harsh to some, it is the perfect place for mule deer herds to spend the winter. Endangered pronghorn roam through in the spring. Golden eagles and prairie falcons hunt throughout the year.[197] Annually, about 2,000 to 3,000 people hike, watch birds, and enjoy non-motorized recreation in the area. How many more might come if the roar of off-roaders wasn't so overwhelming?

The Byers and Pass wildfires

When Fort Sage land managers determined that wildfire associated with off-road use "has not been a significant problem," they must not have been counting the 1996 Byers and Pass Wildfires in the East Bald Mountain Area.[198] On July 2, 1996, an illegally modified off-road vehicle ignited the Byers Wildfire, which burned 1,034 acres of rangeland, both public and private property. Then on Aug. 4, 1996, an automatic weapon started the Pass Fire, which burned 103 acres at a site less than 1/2 mile from the edge of the Byers Fire. Both were in sight of Susanville.[199]

Before the fires occurred, off-roaders were supposedly restricted to designated routes only, but in reality had been traveling willy-nilly cross-country through the brittle, dry area.[200] In order to protect the land from further devastation by off-roaders tearing around the singed hillsides, BLM wisely closed all 3,160 acres of the East Bald Mountain area to off-road vehicle use until further notice.

Wilderness study areas

The Skedaddle Mountains, Five Springs, and Tunnison Mountain Wilderness Study Areas have few roads running through them and retain wild qualities that could eventually lead to wilderness designation. Unfortunately, in some places off-roaders are allowed to drive into the areas on "designated roads." And sometimes they drive in without roads. The damage has not ruined the areas' wilderness qualities, yet trespass and environmental damage threaten the integrity of some of California's last wild places.

During one hunting season, a ranger monitoring the Skedaddle Mountains discovered a camp with a cleared tent site. Garbage, shell casings, and an illegal fire ring were also discovered. In other parts of the Skedaddles, he found branches from surrounding bushes stacked next to a fire ring and a "road closed" sign smashed and on the ground.[201]

The same ranger found severe soil compaction and disturbance along a tributary to Rush Creek in the Five Springs area. He saw four-wheel drive vehicles driving down part of the stream bed, driving over one spring source and three sections of the stream. Camping and dumping were also causing problems.[202] This ranger recommended in his report that the route be blocked off immediately. To date, it remains open.

The constant battle to keep off-roaders out of where they don't belong is never-ending.

Recommendations

- The BLM should complete route designations for all lands within the region.
- The agency should implement off-road vehicle policies that meet its mandate of zero discharge into Eagle Lake.
- The Fort Sage off-road vehicle area should be more closely monitored for off-route riding and penalties should be imposed if offenses occur.
- Rehabilitation and restoration is essential at Fort Sage, particularly in the Land of David.
- The Eagle Field Office should consider vehicle closures during high fire season.
- Trespass into and degradation of wildlands must be stopped by better enforcement, signage and use of barriers.

Snowmobiles: The Forgotten Off-Road Vehicle

Snowmobiling was once viewed as a benign recreational activity, but the problems associatedwith it have received increased attention in recent years, as newer and more powerful machines allow access into previously unreachable wild places.

Snowmobile exhaust is a public nuisance that contaminates air and water. Snowmobiles disturb wildlife and damage plant life. Unscrupulous riders enter wilderness areas, and other closed areas. Skiers, sledders, and snowshoers are endangered by speeding snowmobiles.

Snowmobiles typically use two-stroke engines, which expel 25 to 30 percent of their fuel and oil unburned into the environment. These emissions are highly toxic to plants and animals.[203]

Snowmobilers represent a tiny fraction of outdoor recreationists, but have an impact on numerous skiers, sledders, and other winter enthusiasts. The use of these machines in close proximity to other recreationists makes serious accidents a virtual certainty and destroys the silence, tranquillity, and natural beauty of the winter backcountry.[204]

H O P E V A L L E Y

Running out of Hope in a Crown Jewel of the Sierra

Hope Valley is located along California State Highway 88 just east of the Sierra Nevada crest in Alpine County, California. It is surrounded by the Mokelumne Wilderness to the south and west and the Meiss area (Dardanelles Roadless Area) to the north. A mix of expansive, open grasslands is interspersed with stands of aspen and lodgepole pine. This broad, majestic valley is guarded on all sides by imposing cliffs, ridges, and peaks of dark volcanic rock with intermittent pockets of whitebark pine and mountain hemlock adding to its splendor.

This area is heavily used by both snowmobiles and cross-country skiers, generating considerable user conflicts. Additionally, the proximity of Hope Valley to the Mokelumne Wilderness and the Meiss Roadless Area has led to numerous illegal entries by snowmobiles.[205]

Much of the area is under the jurisdiction of the Carson Ranger District of the Humboldt-Toiyabe National Forest. The Carson Ranger District currently has no rule regarding minimal snow depth for snowmobile use.[206] This lack has led to considerable resource damage from early and late winter snowmobile recreation. Even areas that are off-limits to motorized vehicles during the summer are accessible to snowmobiles during times of minimal snow coverage.

"Recently, I witnessed snowmo-

Joshua Boldt

Damaged meadow: This trail was continually used until the majority of the section was mud. The damage occurred in an extremely fragile alpine area, and is one of hundreds of meadows that have been damaged by careless snowmobile use.

biles using areas of open meadows that were covered in 3 to 4 inches of snow," said area resident Joshua Bolt. "After one or two passes, snowmobiles exposed grass and dirt. They made no attempt to avoid exposed shrubs and vegetation. On the contrary, it seemed as if exposed plants were actually targeted."

Popular paths had 6-inch ruts forming in the soil from repeated passes by snowmobiles. This damage took place in a highly fragile alpine meadow, directly off of a major state highway, with Forest Service personnel present. No effort was made to avoid disturbed areas, and they were not signed as closed.

"Our group, which consisted of six adults, one 5-year-old, one 1 1/2-year-old, and two infants, had stopped to sled with the two older children. All of a sudden, three snowmobiles came off of the hill to the west of the area, from around Red Lake. They came right through our group, scattering us. They went to the road (Highway 88), then turned to come back through our group again. They ruined what little snow there was, and destroyed the small sledding path we had managed to make."

—Jane Starratt, Woodfords

Joshua Boldt

Marginally marked, minimally enforced: Despite numerous incidents of wilderness and closed area trespass, Forest Service officials have failed to provide an adequate law enforcement presence in Hope Valley. The inactivity has continued to frustrate local residents.

Damage to a meadow ecosystem like this is particularly disturbing in light of findings from the recent Sierra Nevada Forest Plan Amendment, prepared by the U.S. Forest Service. According to this report, aquatic, riparian, and meadow ecosystems are the most degraded of all habitats in the Sierra Nevada. These natural systems are critical to the protection of water quality.[20] It is bewildering that the Forest Service would allow and promote an activity that is detrimental to an ecosystem they consider so threatened.

On one occasion, a forest law enforcement officer failed to act when witnessing a snowmobiler grinding his sled over soil and shrubs for more than 100 feet to get to the snow.

"I asked the officer if that was not considered resource damage," said John Brissenden of Sorensen's Resort, who was standing alongside the officer. "He said, 'yes,' but that it wouldn't hold up in court and that they would just throw it out."[208]

It seems likely that the damage here will continue until the Forest Service makes resource protection a priority in Hope Valley and other forest areas subject to snowmobile use.

Recommendations

- Stronger enforcement of snowmobile regulations is critical, including more patrols and better signing.
- Greater penalties for trespass and resource damage should be set, including possible confiscation of the snowmobile and a significant fine. The fine is currently only about $50 to $100.
- Set a minimum snow depth for the operation of snowmobiles.

Trashing a Winter Wonderland

Reds Meadow outside Mammoth in the eastern Sierra is a jumping-off point for both summer and winter excursions into the forest and the wilderness. In summer, backpackers, hikers and mule-trains share trails that take them into the Ansel Adams Wilderness and points beyond. Devil's Postpile Monument attracts day hikers. No dirt bike, snowmobile or other off-road vehicle use is allowed in this area.

Come winter, the Inyo National Forest management plan permits snowmobile use only when there is sufficient snow cover and prohibits operation in developed recreational sites and wilderness areas. Yet large numbers of unlawful snowmobilers ignore the rules and cross into campgrounds, wilderness areas and even Devil's Postpile itself.[209] In most cases, the offenders are never cited or even confronted by Forest Service or other law enforcement officers.[210] The area is not even regularly monitored by anyone other than residents who happen to witness the damage and trespass.[211]

This past winter, several hundred snowmobiles illegally entered Reds Meadow campground to access the Hot Springs.[212] There, they left beer cans and other trash alongside dumpsters locked down for the winter.[213] Part of the problem is that signs in the area said: "No 4-Wheel, No ATVs, No Motorcycles, No Mountain Bikes." But not "No Snowmobiles."[214] Dozens of other violations were reported to the Inyo National Forest by the winter caretaker of one of the area resorts. The offenses included:

- willful trespass into wilderness,
- driving snowmobiles into Devil's Postpile,
- illegal driving in a closed campground,
- destruction of government property,
- damage to vegetation,
- abandoned snowmobiles, and
- physical threats.

Throughout the season, damage was inflicted on vegetation and soil, particularly when snowmobilers drove over patchy snow where ground is exposed. Tops of trees were snapped off. Campgrounds, signs and private property were routinely wrecked by the rogue riders. Sometimes the polluting machines crashed and overturned into creeks and streams, spilling gas and oil into pure waterways. The snowmobiles also routinely drove around locked gates and past closure signs.

On Saturday, March 18, 2000, a snowmobile fell 11 feet down into a creek at Pumice Flat Group Campground, creating a hazardous waste spill. But when the incident was reported, Forest Service rangers said no law enforcement officers were even on duty that day to investigate—and this was on a peak weekend winter day. [215] The sheriff and wildlife officials inspected the scene a few days later, long after the snowmobile had been pulled out and taken home.

Wilderness trespass does not seem to be a concern to some snowmobilers. On Saturday, April 1, four snowmobilers were photographed waving to the camera while heading out of the Ansel Adams Wilderness.[216] They had also been temporarily stuck in Boundary Creek.

Bob Sollima

Government sign damaged by careless snowmobile use.

Bob Sollima

"As early as 1973, I saw snowmobile tracks at Minaret Lake, 20 miles in the wilderness from Mammoth….In 1985 and 1986, shortly after Mammoth Pass was declared wilderness…snowmobiles began running rampant in the Reds Meadow valley, the surrounding wilderness, and the national monument areas….

"It seems that all the snowmobile pilots had 'moto-jump mania,' getting air off anything including buildings and vehicles. In one winter, I tallied damages to fences, hitch rails, a stone barbecue, a roof, my truck, windshield, two tree squirrels, a pine marten, and a fire hydrant [that] was sheared off. I don't get paid around here and I don't much fancy the extra repair work. I've got plenty of roofs to shovel to keep me busy.

"Finally, after many years, we were given permission to keep motor vehicles out of the resort and pack station for security reasons during the closed season. Unfortunately, several hundred snowmobiles have ignored this closure….

"Some of the snowmobile abuses I've seen in the Reds Meadow Valley over the years, in times of low snowpack, are environmental damage to treetops, meadows, trails, creeks, lakes and the river. I've seen two snowmobiles stuck in the San Joaquin River, two submerged in Sotcher Lake and three overturned in creeks, spilling gas and oil into the water. I've seen drunken snowmobile pilots upside down in creeks, spewing out something or other into the water….

"This past holiday season, I saw some snowmobile abuses and a population density of 75 to 100 snowmobiles some days….My records show an average of one snowmobile per day in the wilderness area around here. That's about 200 to 250 wilderness ingresses per season. I've seen evidence of 14 snowmobilers in the wilderness in just two weeks."

—Bob Sollima, winter caretaker, Reds Meadow Resort, Mammoth Lakes

Without increased enforcement, signage and restrictions, snowmobiles will continue to disrupt the quiet hibernation of the Sierra and ruin the winter solitude for skiers, snowshoers and the wildlife that try to survive the harsh conditions.

Recommendations

- The Middle Fork San Joaquin River drainage, including the Minaret's Summit to Reds Meadow road, should be closed to snowmobile entry, except for administrative purposes.
- "No snowmobiles" signs should be posted at all critical access points on the boundary of the San Joaquin River drainage. Violators should be fined and penalized for trespasses and other violations.
- Protection of the wilderness and adjoining areas from snowmobile damage must be made a high priority for the Inyo National Forest.
- The Inyo National Forest should work with local law enforcement authorities to ensure that the latter have a strong presence in the area and respond quickly to snowmobile violations.

Snowmobiles Chasing Out Skiers

Snowmobiles are overrunning Brockway Summit above King's Beach at Lake Tahoe. Snow grooming (evenly dispersing the snow cover) on the west side of Highway 267 enables both casual snowmobilers and commercial outfitters to speed through the woods. Snow-covered roads on the east side draw snowmobilers to the summit of Martis Peak overlooking Lake Tahoe. What was once a favorite cross-country ski and snowshoeing area has become a snowmobile free-for-all, forcing out the traditional recreationists.[217]

Commercial snowmobiling causes a concentration of snowmobiles in the Brockway Summit area and significantly increases the conflict between non-motorized and snowmobile users. Casual snowmobilers have increased their use of the non-groomed road to Martis Peak because the commercial snowmobiling on the other side of the highway is very high. The only ones left out of this equation are the human-powered recreationists who venture out to enjoy the solitude of the mountains.

The snow grooming is funded by the state, but has never been evaluated for environmental impacts as state and federal laws require.[218] Instead, the U. S. Forest Service simply accepts state money, excludes grooming from environmental review and pushes snowmobile highways into the woods. Now skiers are questioning the legality of the ongoing snow grooming, not only at Brockway Summit but throughout California, in hopes of restoring some winter peace and quiet for both people and wildlife.[219]

Due to public pressure, the Forest Service put on hold a plan to build a parking lot near Brockway Summit. Officials from the forest and the state off-highway vehicle program are now conducting talks with winter recreationists to determine a course of action to provide areas for both muscle-powered and snowmobile activities.

Recommendations

The Forest Service should immediately initiate an environmental analysis that discloses the social, biological and physical impacts of the snowmobile use at Brockway Summit. An effort should also be made to mitigate the effects of the continued and expanded use at the summit.

Snowmobilers represent a tiny fraction of outdoor recreationists, but have an impact on numerous skiers, sledders, and other winter enthusiasts. The use of these machines in close proximity to other recreationists makes serious accidents a virtual certainty and destroys the silence, tranquillity, and natural beauty of the winter backcountry.

Joshua Boldt

Damage to streams crossing the Rubicon Trail: Snowmobiles typically use two-stroke engines, which expel 25 to 30 percent of their fuel and oil unburned into the environment. These emissions are highly toxic to plants and animals.

Unprotected Wildlands:

Damage Done by Off-Road Vehicles

Jack Wilburn

Fish Slough Wilderness Study Area, where all-terrain vehicles, dirt bikes, and other off-road vehicles continually stray from legal routes, causing extensive damage and detracting from the area's wild character.

Across California, off-road vehicles have degraded hundreds of thousands of acres of once-pristine wildlands. In southern California, the impact of dirt bikes, snowmobiles and other off-road vehicles on unprotected wildlands has been quite severe, and has been cited as a leading cause of wild area destruction.[220] By ignoring illegal activity, or sanctioning and actively encouraging off-road vehicle use, federal land managers are willfully enabling dozens of areas to lose their wild character.

For example, the Caples Creek area, located in the Tahoe Sierra, contains over seventeen miles of authorized off-road vehicle routes in its northern portion. Ironically, the Forest Service recommended to Congress that the southern and central portions of the roadless area be designated as wilderness.[221]

The off-road vehicle routes in the Caples Creek Roadless Area pass through many sensitive meadows and are located near streams in some places. The noise from dirt bikes in particular travels for miles, even into the portion of the roadless area recommended for wilderness status, and several archaeological sites have been damaged.[222]

The 94,000-acre Sespe-Frazier Roadless Area in the Los Padres National Forest contains several miles of off-road vehicle routes. Piru Creek, a stream that provides habitat for a variety of sensitive species such as the arroyo toad, is followed for many miles by off-road vehicle routes and is crossed by them at several points. The Forest Service has acknowledged that off-road vehicle use in and near the creek has damaged habitat for the toad, and yet it has approved new routes in the roadless area despite administrative appeals from conservationists. The net effect of these routes has been to reduce the ecological and aesthetic character of the area to such an extent that over 6,000 acres no longer qualify for wilderness designation.[223]

Turning firebreaks into off-road vehicle routes

A common way in which dirt bikes and other off-road vehicles damage roadless areas is by using firebreaks and turning them into de facto roads over time. These routes are built hastily during fires under emergency conditions and are therefore rarely designed to minimize erosion or accommodate vehicle use of any kind. As a result, federal and state agencies usually attempt to close firebreaks and rehabilitate them after fire-suppression efforts are completed.

However, it is often a fairly simple matter for all-terrain vehicles, dirt bikes, and other off-road vehicles to bypass attempts to close firebreaks and to turn them into motorized vehicle routes over time. By continuing to use the supposedly closed routes, off-road vehicle riders exacerbate the erosion and other impacts already occurring on firebreaks due to their steep and hastily constructed nature. Literally dozens of unroaded areas have been essentially roaded in this manner.

For example, the Reister Canyon

Roadless Area in the Mendocino National Forest had firebreaks constructed on many of its major ridges during fires in the 1980s. Today, all of these firebreaks have failed to recover because of continued dirt bike and other off-road vehicle use. Several new routes have also been illegally pioneered between the firebreaks to create loop "trails."

The Forest Service admits that, "a significant amount of unauthorized OHV use occurs along with some unauthorized maintenance of OHV trails."[224] This extensive system of motorized vehicle routes has destroyed the wild qualities that the Reister Canyon Roadless Area once had. The visual scars resulting from the off-road vehicle use can be seen for miles.

Steve Tabor, Desert Survivors

Rice Valley Wilderness: These tracks were carved miles into the wilderness, despite the fact that the area was closed to motorized vehicles in 1994.

Ruining the desert

Of all of California's public lands, those in the desert are the most vulnerable to off-road vehicle damage. In 1976, the Federal Land Policy and Management Act (FLPMA) required the Bureau of Land Management (BLM) to assess the wilderness potential of its lands. As a result of this assessment, the BLM identified millions of acres of wild country in California, primarily in the desert, as "wilderness study areas" (WSAs) suitable for designation as wilderness. The agency provided interim protection for WSAs until Congress could decide whether or not to designate them as wilderness.

Many wild areas in the desert were not identified as WSAs because all-terrain vehicle, dirt bike, and other off-road vehicle use had destroyed their wilderness character prior to the surveys being conducted For example, portions of the Algodones Dunes, Johnson Valley, and Stoddard Valley were declared "open areas" for off-road vehicles by the BLM in 1980.[225]

An open area is a region where dirt bikes and other off-road vehicles are allowed to travel anywhere without any restrictions. Areas designated as open are easily devastated by the creation of routes virtually everywhere it is physically possible for an off-road vehicle to make one.

The Stoddard Valley region in particular had areas with exceptional value, including outstanding habitat for the desert tortoise, prior to being opened up to off-road vehicles.

Areas designated as WSAs in the 1980s have mostly retained their wild character since that time. However, off-road vehicle use has degraded some portions of a few WSAs. For example, in the Soda Mountains WSA in the California desert, illegal off-road vehicle routes through the Cronese Basin have degraded a wash that flows through the area.[226] The illegal routes through the Cronese Basin are shortcuts between existing, legal routes outside of the WSA.

The creation of illegal shortcuts between existing routes is a common problem in the desert and other areas where the land is open in character and thus vulnerable to off-road vehicle trespass. In the Cronese Basin and areas like it, habitat for the desert tortoise, archaeological sites, and the ability of the area to qualify for wilderness designation are all threatened by the creation of illegal routes.

Recommendations

- Areas currently being considered for wilderness designation should be off-limits to off-road vehicles.
- Greater law enforcement is needed to ensure that off-road vehicles do not cause damage to wilderness-quality lands.
- Past damage, whether created by legal or illegal use, should be immediately addressed.
- Firebreaks and other illegitimate routes should be immediately closed to dirt bikes and other off-road vehicles and rehabilitated where appropriate.

Positive Steps

Where can examples of positive off-road management be found? Are there places where land managers have taken steps to repair, reduce and restrict the inevitable damage inflicted wherever dirt bikes, snowmobiles, and other off-road vehicles tread?

Based on public records and on-the-ground inspections, few, if any, off-road areas are managed well enough to prevent long-lasting environmental damage. However, in several areas land managers have taken positive steps to improve off-road vehicle management.

In some cases, closures have been the only solution. Black Sands Beach, along California's Lost Coast, was closed in 1998 due to environmental damage, closure violations, and user conflicts. Similar closures have been put in place in southern California to protect endangered species such as the desert tortoise and Peirson's milkvetch.

In other areas, limiting riding to designated routes or regions is an acceptable solution. The cases below demonstrate that positive gains can be made in off-road vehicle management, but they require commitment and dedication from land managers.

Red Hills

Located south of Chinese Camp, the Red Hills were permanently closed to off-road vehicles beginning in 1991 after years of uncontrolled driving and hill climbing, random shooting, destruction of property and trespass.[227] Dust clouds and denuding of the rocky hillsides caused by off-roaders had turned the area into an eyesore.

An "ever increasing" user-created network of paths and trails resulted in widespread trampling of flowers and plants that grow only in the reddish hued and highly erosive serpentine soils of Red Hills.[228] Few trees grow on this rocky island of land surrounded by oak woodland, but every spring, dazzling wildflowers displayed their colors. Attracted to the wide open spaces, the vehicles stripped the vegetation that clung to the tops of hilltops, often denuding hills entirely. Trails became deeply gullied. Erosion was a "very serious" problem.

Trash was dumped. Other visitors were scared off by the noise and reckless driving. Off roaders routinely drove around after dark "as fast as one can go."[229] Neighboring landowners complained more and more about the problems.[230]

Because of the unruliness and environmental damage, and following BLM policy to protect natural resources, Folsom Resource Manager Deane Swickard declared an emergency closure to off-road vehicles that was made permanent in 1993. Swickard had never used state funding to manage the area and was therefore free to make a tough decision without concern for lost monies.

The closure was designed to "reduce activities that damage and scar the land and allow damaged areas to recover their natural appearance."[231] Since the closure was put into place, "soil erosion and related problems have been virtually

Jim Rose

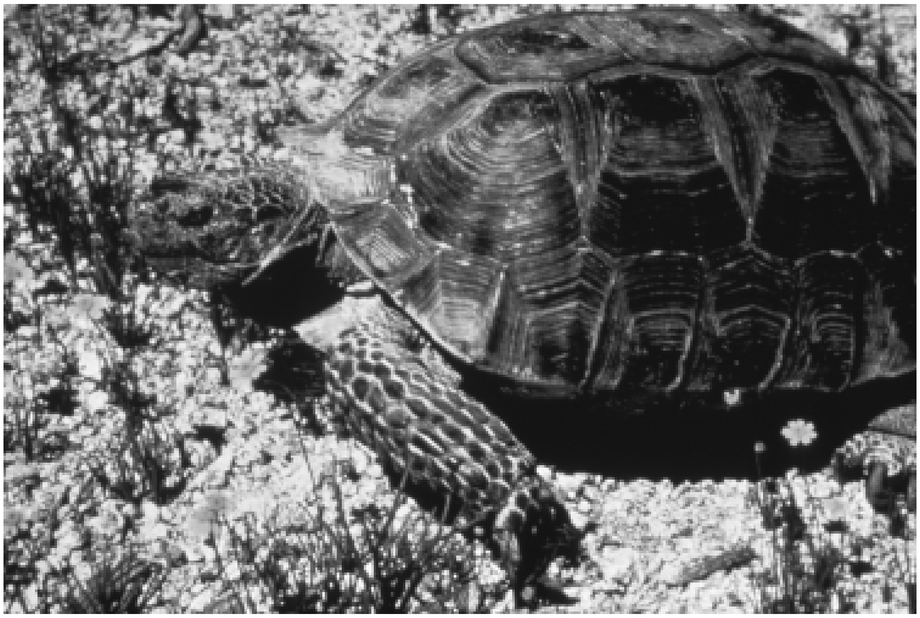
In some regions of the state, federal land managers are attempting to minimize the damage and conflicts presented by motorized vehicles. In some areas, like the Red Hills, the only solution is to close the area to dirt bikes and other off-road vehicles. In other areas, increased planning and law enforcement is needed.

halted"[232] and the Red Hills have come back to life.

Rand Mountains and Fremont Valley

The Rand Mountains and Fremont Valley are essential to the survival of the desert tortoise. In fact, more tortoises lived in this particular stretch of high desert than anywhere else—as many as 250 tortoises in one square mile—before a drastic decline in the 1980s.[233]

Prior to 1990, the area had been utilized by off-road vehicles for decades, and was open to cross-country travel for much of that time.[234] More than 200 organized motorcycle events blazed through tortoise territory each year.[235] In 1980 when new desert protections were passed, a designated route system was implemented and organized events were reduced. Nevertheless, between 1981 and 1987, nearly half of the desert tortoises found dead in a scientifically monitored plot in Fremont Valley were killed by vehicles, vandalism and gunshot wounds.[236] Clearly, the route system and management were not protecting the declining desert tortoise.



Howard Wilshire

Much progress has been made in southern California's Rand Mountains. Legal riding routes have been identified, and enforcement has begun. While numerous problems still exist, the area is managed far better than two decades ago, when rampant, unregulated riding took a tremendous toll on desert resources.

The area was closed to off-road vehicle use for a short time in 1989 after the tortoise was listed as a threatened species by the state of California.[237] The temporary closure soon ended, however.

By 1993, a new plan for the area was written that required an overall 83 percent decrease in the number of miles of trails and routes for off-road travel. Organized off-road events were banned. More "open" and "closed" signs were planted, and fences built to protect closed areas. Old trails were blocked with boulders and bales of straw. The BLM began obliterating parallel trails. The slow, but positive, change from an open area to a limited use area was underway.

However, big problems remain. The Bureau of Land Management, ignoring its own management plan, approved a 500-motorcycle event over Thanksgiving 2000.[238] Desert activists stopped this illegal race just days before the well-publicized event.

Riding on non-designated routes continues. A closed road that crosses an important desert tortoise area is shown on off-road maps as open.[239] Managers are aware that illegal off-system riding occurs, but say that the situation is improving.

"We don't have the resources to implement all 88 specific management decisions made in the new plan," said Steve Smith, Recreation/Wilderness Supervisor at the Ridgecrest BLM office, "but we are getting 80 to 90% compliance with the route system and that's a major improvement."

A new plan…was written that required an overall 83 percent decrease in the number of miles of trails and routes for off-road travel. Organized off-road events were banned. More "open" and "closed" signs were planted, and fences built to protect closed areas. Old trails were blocked with boulders and bales of straw. The BLM began obliterating parallel trails.

Reform: Preventing Off-Road Vehicle Damage

As documented throughout this report, off-road vehicle damage to California's landscapes is extensive, widespread, and often irreversible. Numerous sensitive watersheds, streams, wildlife habitats, archaeological sites, and scenic treasures have been degraded by indiscriminate dirt bike, snowmobile and other off-road vehicle use.

Fortunately, while past use has, in many places, caused permanent scars, future damage can still be prevented. Preserving California's natural heritage, however, will require decisive action by both state and federal policy-makers and land managers. In order to prevent continued damage caused by dirt bikes and other off-road vehicles to California's landscapes, immediate changes in state and federal policy are needed.

FEDERAL REFORM

Simply complying with the President's executive orders issued in the 1970s will carry federal land managers a long way toward cleaning up off-road vehicle abuse of federal lands. To comply with current laws and alleviate the damage caused by off-road vehicles to public lands, federal land managers should:

Designate and map legal riding routes

Motorized vehicle use should be limited to vehicle routes designated, mapped, and posted by the appropriate land management agency as open to motor vehicle use, after the completion of environmental impact analysis. Motor vehicle use off designated routes should be prohibited.

Determine where use is appropriate

All vehicle routes on federal land should be subject to environmental impact analysis, and motor vehicle use should be allowed only on those routes where the appropriate land management agency has documented that vehicle use will not cause adverse environmental impacts, and that impacts to the environment and other recreationists will be minimized.

Susan Harrison

Designating and enforcing legal riding routes will help to prevent illegal off-trail riding, such as that shown here at the Knoxville Recreation Area. Despite clear legal mandates, federal land managers have generally failed to make and enforce off-road vehicle route designations.

Federal agencies have completed only minimal monitoring of the effects of off-road vehicles. Additional funding is needed to ensure that land managers are adequately monitoring impacts caused by off-road vehicles, and that they are restoring damaged areas.

Inga Spence

Thresholds for unacceptable impacts must be established prior to beginning analysis.

Monitor the effects

The use of motorized vehicles should be allowed only in those areas where federal land managers are able to actively monitor the effects of motorized vehicles on the landscape. If monitoring determines that thresholds established for unacceptable impacts are reached in an area or on a trail, the area or trail will be closed until the impact is reduced to an acceptable level.

Mimi Jennick

The State of California should not be funding off-road vehicle use in areas where such use is damaging sensitive resources like undesignated wilderness or potential Wild and Scenic rivers.

Protect undesignated wilderness

Across the state, wilderness-quality lands are being degraded by motorized vehicles. These areas should be declared off-limits to motorized vehicles.

Enforce the law

Some federal land managers are failing to prevent motorized vehicles from entering wilderness and other closed areas. Land managers should make trespass and closure violations a higher priority. Congress should appropriate additional funds to assist land managers in enforcing off-road vehicle laws and regulations.

STATE REFORM

In 1971, California enacted the Chappie-Z'Berg Off-Highway Motor Vehicle Act (OHV Act), which created the state of California off-road vehicle program. In the past three decades, this program has allocated over half a billion dollars to support off-road vehicle use on state, federal, and private land throughout California.

The OHV Act, as amended, found that "the indiscriminate and uncontrolled use of those vehicles may have a deleterious impact on the environment, wildlife habitats, native wildlife, and native flora," and that "effectively managed areas and adequate facilities for the use of off-highway vehicles and conservation and enforcement are essential for ecologically balanced recreation."

Without active involvement, the state runs the risk of being held responsible for the shortcomings of off-road vehicle management, while remaining unrecognized for the benefits of its grants program. The following legislative reform will help to bring balance to the state's off-road vehicle program, by ensuring that state funding is used to repair damaged areas, prevent future damage, and mitigate the effects of off-road recreation. State legislators should require:

Jim Rose

Increased fines for trespass into wilderness and other closed areas will deter future dirt bikers and other off-road vehicle users from breaking the law. Current fines are as low as $50.

Increased funding for conservation and law enforcement

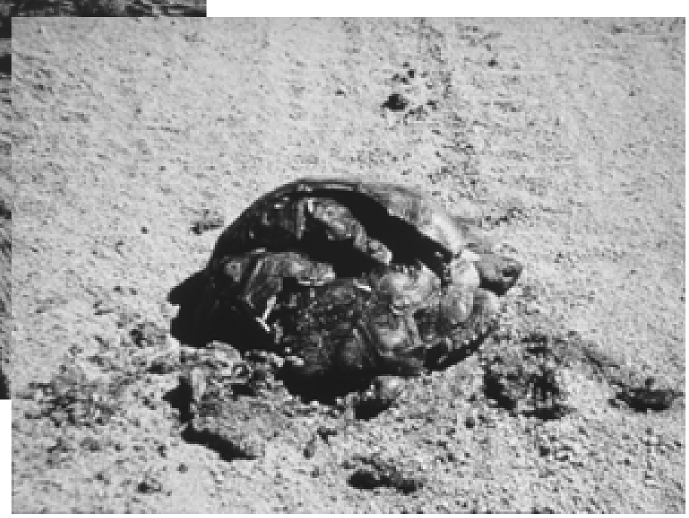
The state's off-road vehicle act urges California to control the impacts caused by the "indiscriminate and uncontrolled use" on the "environment, wildlife habitats, native wildlife, and native flora." This means that funding is needed to effectively enforce closed areas, protect soils and watersheds, carry out monitoring and remediation work, and keep riders on designated routes. Current funding is not adequate to fulfill these needs, and additional funding should be authorized.

Mitigation funding and non-motorized buffers

Off-road vehicles can cause extensive harm to the natural environment and wildlife habitat. Funding for mitigation of off-road vehicle damage is needed to ensure that critical habitat areas are protected. This mitigation may be possible at

George Barnes

Current state standards for soil and wildlife preservation are highly technical. The standards should be updated and uniformly applied.



Howard Wilshire

the site of the off-road vehicle use, or may be more appropriate elsewhere.

Uniform soil and habitat standards

Currently the state is utilizing highly technical soil protection standards that are difficult for non-geologists to apply. These standards should be updated and applied uniformly.

Polluter pays

Registration fees for off-road vehicles should be linked to emissions levels (higher emissions equals higher fees). This will create a positive incentive to reduce emissions from off-road vehicles.

Reducing off-road vehicle-related crimes

Currently, fines for riding a motor vehicle into closed areas are too low to effectively discourage use. Fines for vehicle trespass into closed areas should be dramatically increased to create a real deterrent to illegal riding.

OFF-ROAD VEHICLE GRANTS PROGRAM

Further, through its off-road vehicle grants program (which provides millions of dollars each year to support the acquisition, development, and operation of off-road vehicle facilities and areas on federally managed lands), the state is in a unique position to positively influence off-road vehicle management on public lands.

In order for the program to adequately mitigate the effects of off-road recreation and prevent excessive off-road vehicle-related damage, the state should adopt the following principles with regard to its off-road vehicle grants program:

Howard Wilshire

California contains hundreds of thousands of acres of "open" riding areas, where dirt bikes and other off-road vehicles are causing excessive damage to soils, plants, and watersheds. Riding should be limited to designated routes.

Comply with the law

The top priority of the grants program should be to monitor and repair existing resource damage, prevent future damage, and ensure compliance with state and federal laws and regulations. Grants should not be given to districts that cannot ensure compliance with federal and state laws and regulations, except to bring those areas into compliance with the law.

continued on the next page

Jim Rose

Acquisition and development of new off-road vehicle trails and facilities should cease until all lands within the program are in full compliance with all applicable state and federal laws, regulations, and policies, and current resource damage is adequately addressed.

Protect sensitive areas

Grants to support projects that could adversely impact or jeopardize the ecological integrity or social values of wild areas or rivers should be eliminated.

Prevent future damage

Acquisition and development of new off-road vehicle areas and trails should cease until all lands within the program are in full compliance with all applicable state and federal laws, regulations, and policies, and current resource damage is adequately addressed.

Respect other land users

Grants should not be used to fund projects that create or expand conflicts with non-motorized recreationists. Projects submitted for grant awards should ensure that residents and private property owners adjacent to proposed project areas are protected from noise, trespass, and property damage.

Do no harm

The state should not fund off-road vehicle activities in areas where such use has been shown to cause unacceptable environmental damage, or will lead to damage of sensitive lands, an increase in illegal riding or conflicts with other recreationists. Furthermore, the state should not fund areas that cannot demonstrate compliance with all federal and state laws and policies.

The enactment of these changes will help to ensure that California's landscapes are no longer scarred by indiscriminate off-road vehicle use, and are instead preserved in their grandeur, in perpetuity.

Jim Rose

The top priority of the state's grants program should be to repair existing damage, prevent future damage, and ensure compliance with laws and regulations.

Appendix A: Footnotes

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Appendix B
Trespass into Closed Areas

California desert

Big Maria Mountains Wilderness
Black Mountain Wilderness
Butterbredt Springs Area of Critical Environmental Concern
Carrizo Gorge Wilderness
Coachella Valley Preserve
Coso Wilderness
Coyote Mountains Wilderness
Death Valley National Park
Devil's Postpile National Monument
Fish Creek Wilderness
Golden Valley Wilderness
Grass Valley Wilderness
Indian Pass Wilderness
Jacumba Mountains Wilderness
Kingston Range Wilderness Study Area
Mojave National Preserve
Newberry Mountains Wilderness
Nopah Range Wilderness
North Algodones Wilderness
Orocopia Wilderness
Owens Peak Wilderness (Pacific Crest Trail)
Pacacho Wilderness
Palen McCoy Wilderness
Rice Valley Wilderness
Rodman Mountains Wilderness
Salt Creek Wilderness
Sawtooth Wilderness
Table Mountain Area of Critical Environmental Concern
West Mesa Area of Critical Environmental Concern
Yuha Desert Area of Critical Environmental Concern
Yuha Desert Flat-Tailed horned Lizard Management Area

Southern California

Bighorn Wilderness

Northeast California

Five Springs Wilderness Study Area
Skedaddle Wilderness Study Area
Tunnison Wilderness Study Area

Sierra Nevada

Ansel Adams Wilderness
Carson-Iceberg Wilderness
Desolation Wilderness
Dome Land Wilderness
Freel Peak Roadless Area
Inyo Mountain Wilderness
John Muir Wilderness
Lassen National Park
Meiss Meadows Roadless Area
Mokelumne Wilderness
Mt. Rose Wilderness

George Barnes

North Algodones Wilderness Area

George Barnes

To the left of the road (foreground of the picture) is the North Algodones Wilderness Area. The wilderness is closed to motorized vehicles. To the right side of the road, the side stripped of vegetation, is the North Algodones off-road vehicle area.

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NATIONAL COUNCIL FOR AIR AND STREAM IMPROVEMENT

RIPARIAN ZONE FOREST MANAGEMENT AND THE PROTECTION OF BIODIVERSITY: A PROBLEM ANALYSIS

TECHNICAL BULLETIN NO. 908

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PRESIDENT'S NOTE

Forestry practices have complex effects on the biological diversity of forest ecosystems. Forest management activities in riparian (streamside) zones are thought to have especially important effects. Riparian forests provide habitat for many aquatic and terrestrial species and have important ecological functions such as regulating stream temperatures and supplying organic matter to stream channels. It is not surprising that forestry BMPs and regulations often place special emphasis on riparian zone management.

Riparian forest management and its effects on biological diversity are important and controversial topics in the Pacific Northwest. Federal and state regulations affecting riparian zone management are major factors affecting wood supplies and economic returns to public and private forest owners in the region.

NCASI has been an active participant in research on riparian forest management since the late 1970s in partnership with agencies, universities, and member companies. Excellent progress has been made in documenting the ecological functions of riparian zones and in developing riparian BMPs that greatly reduce the impacts of forestry operations on water quality. Today, the scientific and policy debate about riparian forest management in the Pacific Northwest is focused on the degree to which BMPs and alternative prescriptions are effective in conserving important aspects of biological diversity including species that are imperiled and/or highly sensitive to disturbance.

This report provides an overview of the latest scientific information and expert opinion on riparian forest management and its effects on biological diversity in the Pacific Northwest. NCASI contracted with Dr. Daniel Sarr of the National Park Service to organize a multidisciplinary writing team and provide overall leadership for the project. Dr. Larry Irwin, Manager of NCASI's Western Wildlife Program, worked with Dr. Sarr on designing the project and editing this report.

In Part 1, the report's lead authors discuss what is meant by the often misused concept of biodiversity, and describe how natural processes of disturbance create the habitat heterogeneity that, at a range of scales, provides for biodiversity. Although forestry disturbances and natural disturbances can be distinguished, the authors describe how both can be evaluated in terms of their effects on biodiversity by assessing how they influence habitat heterogeneity, as well as legacy retention, physiological stress, and resource availability. These can all be assessed together in the graphical model developed by the authors.

In Part 2, the report addresses forest management effects on specific elements of biodiversity in sections prepared by experts on several taxonomic groups of aquatic and terrestrial organisms. In Part 3, the lead authors provide a synthesis of the taxa-specific information and propose some general principles and approaches to the protection of riparian biodiversity that may be employed at stand or landscape scales. The authors note that aligning forestry treatments to attempt to emulate natural disturbance regimes in landscapes has received much recent attention as the best approach in this regard. However, doing so may not be cost-effective or feasible in some landscapes. The review also points out that there are both merits and limitations to the riparian buffer approach to protecting biodiversity. The authors identify major gaps in ecological information on these topics, and the report concludes with a research agenda and a framework for critically evaluating past research.

The information, interpretation, and recommendations presented by the authors of this report have important implications for NCASI priorities. For example, it seems clear that perceptions of forestry impacts on biodiversity are influenced to a considerable extent by historical practices (e.g., harvesting without BMPs) and stand-level studies that may have limited relevance to current practices and landscape-level effects.

Ronald A. Yeske

October 2005

MOT DU PRÉSIDENT

Les pratiques forestières produisent des effets complexes sur la diversité biologique des écosystèmes forestiers. On estime que les activités d'aménagement forestier dans les bandes riveraines produisent des effets particulièrement importants. Les forêts riveraines fournissent un habitat pour plusieurs espèces aquatiques et terrestres. Elles possèdent d'importantes fonctions écologiques telles que le contrôle de la température des cours d'eau ainsi que l'acheminement de la matière organique dans les lits des cours d'eau. Il va de soi que l'on retrouve dans les meilleures pratiques d'aménagement forestier (MPAF) et la réglementation, une attention particulière à l'aménagement de la bande riveraine.

Sur la côte nord ouest du Pacifique, l'aménagement de la forêt riveraine et ses effets sur la diversité biologique constituent un domaine d'activité de premier plan, quoique controversé. Les réglementations fédérale et étatiques s'appliquant à l'aménagement de la bande riveraine sont des facteurs majeurs affectant l'approvisionnement en fibres et les bénéfices économiques pour le public et les propriétaires de forêts privées dans la région.

NCASI a activement participé aux recherches portant sur l'aménagement de la forêt riveraine depuis la fin des années 1970, en partenariat avec les agences gouvernementales, les universités et les compagnies membres. D'excellents progrès ont été réalisés en matière de documentation des fonctions égologiques des bandes riveraines et en matière de développement de BPF qui réduisent considérablement les impacts des opérations forestières sur la qualité de l'eau. Aujourd'hui, le débat scientifique et réglementaire au sujet de l'aménagement de la forêt riveraine sur la côte nord ouest du Pacifique se concentre sur la capacité des MPAF et des méthodes de rechange à être efficaces pour la conservation d'aspects importants de la diversité biologique, incluant les espèces menacées d'extinction et/ou hautement sensibles aux perturbations.

Ce rapport brosse un tableau des dernières informations scientifiques et opinions d'experts sur l'aménagement de la forêt riveraine et ses effets sur la diversité biologique de la côte nord ouest du Pacifique. NCASI a donné le mandat au docteur Daniel Sarr du service des Parcs Nationaux d'organiser une équipe de rédaction multidisciplinaire et d'assumer la direction générale du projet. Le docteur Larry Irwin, gestionnaire du Programme de la faune de l'Ouest de NCASI, a travaillé en collaboration avec le docteur Sarr sur la conception du projet et la rédaction de ce rapport.

Dans la partie 1 du rapport, les auteurs principaux discutent de la signification du concept parfois mal utilisé de biodiversité. Ils décrivent comment les processus naturels de perturbations créent l'hétérogénéité de l'habitat qui, à plusieurs niveaux, assure la biodiversité. Même s'il est possible de faire la distinction entre les perturbations forestières et les perturbations naturelles, les auteurs décrivent comment ces deux éléments peuvent être évalués en termes de leurs effets sur la biodiversité, en établissant comment ils influencent l'hétérogénéité de l'habitat, de même que les legs biologiques, les contraintes physiologiques et la disponibilité des ressources. Ces facteurs peuvent être évalués ensembles par l'intermédiaire d'un modèle graphique développé par les auteurs.

Dans la partie 2, le rapport couvre les effets de l'aménagement forestier sur des éléments spécifiques de la biodiversité. Cette information est présentée sous forme de sections préparées par des experts et porte sur plusieurs groupes taxonomiques d'organismes aquatiques et terrestres.

Dans la partie 3, les auteurs principaux présentent une synthèse de l'information spécifique aux taxons et ils proposent quelques principes généraux et approches pour protéger la biodiversité riveraine. Ces éléments peuvent être utilisés à l'échelle du peuplement ou du paysage. À cet effet, les auteurs notent qu'on a récemment porté beaucoup d'attention sur l'approche qui consiste à faire en sorte que les traitements forestiers émulent, autant que faire se peut, les régimes de perturbation naturelle dans les paysages. Cette approche est considérée comme étant la meilleure jusqu'à maintenant, toutefois, elle peut s'avérer non rentable ou encore non réalisable dans certains paysages. La revue fait également ressortir qu'il existe des avantages et des limites à l'approche de bande tampon riveraine en ce qui concerne la protection de la biodiversité. Les auteurs identifient les principales lacunes dans l'information écologique traitant de ce sujet et le rapport conclut en proposant un programme de recherche et un cadre d'évaluation critique des travaux de recherche passés.

L'information, l'interprétation et les recommandations émises par les auteurs de ce rapport entraînent d'importantes implications sur l'établissement des priorités de NCASI. Par exemple, il semble clair que la perception des impacts de la foresterie sur la biodiversité est influencée, en bonne partie, par les pratiques historiques (par exemple, la récolte sans MPAF) de même que par les études spécifiques au niveau du peuplement dont l'applicabilité aux pratiques actuelles et les effets au niveau du paysage sont limités.

Ronald A. Yeske

Octobre 2005

RIPARIAN ZONE FOREST MANAGEMENT AND THE PROTECTION OF BIODIVERSITY: A PROBLEM ANALYSIS

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ABSTRACT

This report evaluates the general effects of forestry practices on biodiversity along streams in the Pacific Northwest and northern California. There are four parts to the report. In Part I, we present concepts of biodiversity and the processes underlying it. Biodiversity is expressed as a general concept for species, habitat, and genetic diversity of all groups of organisms. We describe the interacting processes that govern riparian biodiversity by integrating those operating over large spatial extents, such as climate, with interrelated ones that have more localized influences, such as disturbance and habitat heterogeneity. The effects of forestry on biodiversity are then analyzed in the context of these controls, and how they are influenced by disturbances. We predict that habitat heterogeneity and retention of pre-disturbance biological legacies (trees, snags, logs, seed and spore banks that can be important to growth of populations of organisms after disturbance) are two of four key determinants of biodiversity because they may act as mechanisms that promote species coexistence. Habitat heterogeneity is especially scale-dependent. Physiological stress and related resource availability are the other two primary controllers of biodiversity because they may limit the number of species that coexist. These limiting factors are strongly influenced by geography. All four factors are combined into a simple graphical model for predicting how disturbance regimes in general, and forestry practices in particular, will affect biodiversity. Disturbance regimes that are intermediate in influence are predicted to best maintain biodiversity. Geographic variation, as described in Appendices A and B, must be considered when implementing the conceptual model, and we illustrate this by contrasting how a variety of forestry practices are predicted to affect biodiversity in relative extremes in the Pacific Northwest: wet forests west of the Cascades vs. dry forests on the east slope of the range.

The primary controllers of species diversity will have different effects on organisms depending on their life histories. Therefore, in Part II, we provide separate chapters by selected authors summarizing information about the effects of forestry practices on biodiversity along streams in the study area for specific taxonomic groups. These summaries contain the most current information on the ecology of the taxonomic groups, and how they and their habitats may be affected by forestry practices. Each section also suggests forestry practices that may sustain the selected taxonomic group. Finally, research needed to improve understanding of these taxa-specific topics is described.

Synthesizing this information in Part III, we stress that there may be tradeoffs in managing for different elements of biodiversity, which leads to complications in managing for overall biodiversity. This highlights the need for clear articulation of management goals. For improving overall biodiversity maintenance, the principles outlined in Part I lead to potentially cost-effective stand-level management actions. In terms of enhancing habitat heterogeneity, planting multiple crop species, leaving some native trees unharvested to remain through a second rotation, lengthening rotations and earlier thinning schedules may all be effective, depending on the circumstances. Woody debris and snags are critical habitat features for many species that can be maintained or created to improve legacy retention. Site preparation following harvesting that creates biological legacies that occur with natural disturbances and that conserves coarse woody debris can help maintain many non-crop species. Controlling exotic species that act as artificial keystones/pest plants can reduce physiological stress and maintain more natural resource availability for native species. We also describe strategies for

maintaining biodiversity at the landscape scale. Specifically, we discuss some advantages and limitations of disturbance regime-based management, riparian buffers, and conservation reserves as means to protect biodiversity.

The report concludes in Part IV with a draft research agenda to complement taxon-specific research recommended in Part II. This research agenda is based on reviews of existing literature and ongoing research, which exhibits geographic and taxonomic biases. The goal of the research proposed is to improve understanding of how to protect biodiversity in managed forests. There is a need for much basic ecological information about both the ecology of lesser known riparian taxa, as well as applied research determining their sensitivity to forestry related disturbance.

KEYWORDS

aquatic invertebrates, biodiversity, birds, buffer, disturbance regime, endangered species, fish, forest zones, fungi, keystone species, mammals, plants, riparian, stream amphibians, vegetation

RELATED NCASI PUBLICATIONS

Technical Bulletin No. 885 (August 2004). *Managing elements of biodiversity in sustainable forestry programs: Status and utility of NatureServe's information resources to forest managers.*

Technical Bulletin No. 857 (January 2003). *Wildlife and biodiversity metrics in forest certification system*s.

Technical Bulletin No. 799 (January 2000). *Riparian vegetation effectiveness.*

Technical Bulletin No. 775 (January 1999). *Assessing effects of timber harvest on riparian zone features and functions for aquatic and wildlife habitat.*

AMÉNAGEMENT FORESTIER DES BANDES RIVERAINES ET PROTECTION DE LA BIODIVERSITÉ : ANALYSE DE LA PROBLÉMATIQUE

BULLETIN TECHNIQUE NO. 908
OCTOBRE 2005

RÉSUMÉ

Ce rapport évalue les effets généraux des pratiques forestières sur la biodiversité le long des cours d'eau de la côte nord ouest du Pacifique et de la Californie du Nord. On retrouve quatre parties dans ce rapport. Dans la partie 1, nous présentons les concepts de biodiversité et ses processus sous jacents. La biodiversité se définit comme un concept général pour les espèces, l'habitat et la diversité génétique de tous les groupes d'organismes. Nous décrivons les processus interactionnels qui gouvernent la biodiversité riveraine en intégrant ceux qui opèrent sur de grandes étendues spatiales telles que le climat, en incluant ceux qui sont interreliés et qui produisent des influences plus locales telles que les perturbations et l'hétérogénéité de l'habitat. Les effets de la foresterie sur la biodiversité sont ensuite analysés selon ces contrôles et selon la façon dont les perturbations les influencent. Nous prévoyons que l'hétérogénéité de l'habitat et la rétention des legs biologiques avant perturbation (arbres, chicots, billes, amas de graines et de spores qui peuvent s'avérer importants pour la croissance des populations d'organismes après perturbation) constituent deux des quatre facteurs déterminants de la biodiversité parce qu'ils peuvent agir comme des mécanismes qui favorisent la coexistence d'espèces. L'hétérogénéité de l'habitat est particulièrement dépendante de l'échelle. La contrainte physiologique et la disponibilité des ressources associées constituent les deux autres agents primaires de contrôle de la biodiversité car ils peuvent limiter le nombre d'espèces qui coexistent. La géographie influence fortement ces facteurs limitants. On a combiné ces quatre facteurs dans un modèle graphique simple pour prédire comment les régimes de perturbation en général et les pratiques forestières en particulier, affecteront la biodiversité. On prévoit que les régimes de perturbations dont l'influence demeure intermédiaire seront les meilleurs pour maintenir la biodiversité. Il est nécessaire de considérer la variation géographique, décrite dans les annexes A et B, lors de l'implantation d'un modèle conceptuel et nous illustrons ceci en comparant comment une variété de pratiques forestières affecteront, on suppose, la biodiversité dans les extrémités relatives de la côte nord ouest du Pacifique : les forêts humides de l'ouest des Cascades vs les forêts sèches le long de la pente est de la chaîne.

Les agents de contrôle primaires de la diversité des espèces produisent des effets différents sur les organismes, dépendant de leurs cycles biologiques. Par conséquent, nous présentons, dans la partie II, des chapitres préparés par des auteurs sélectionnés, qui font la synthèse de l'information sur les effets des pratiques forestières sur la biodiversité le long de cours d'eau situés dans la zone d'étude pour des groupes taxonomiques spécifiques. Ces synthèses contiennent l'information la plus récente sur l'écologie des groupes taxonomiques ainsi que sur la façon dont ces groupes et leurs habitats peuvent être affectés par les pratiques forestières. Chaque section contient également des suggestions sur les pratiques forestières qui sont susceptibles de maintenir le groupe taxonomique retenu. Enfin, on retrouve une description des besoins de recherche pour améliorer notre compréhension des taxons spécifiques.

Nous avons fait la synthèse de cette information dans la partie III et nous mettons l'accent sur le fait qu'il pourrait être nécessaire de faire des compromis dans la gestion des différents éléments de la biodiversité, ce qui amène des complications dans la gestion de la biodiversité globale. Cette situation démontre qu'il est nécessaire de bien formuler les objectifs d'aménagement. Afin d'améliorer le maintient de la biodiversité globale, nous croyons que les principes avancés dans la

partie I sont précurseurs d'actions d'aménagement, au niveau du peuplement, potentiellement rentables financièrement. En ce qui a trait à l'amélioration de l'hétérogénéité de l'habitat, planter de multiples espèces cultivées, laisser quelques arbres indigènes en place jusqu'à une seconde révolution des peuplements, rallonger les périodes de révolution et raccourcir les programmes d'éclaircies représentent toutes des pratiques potentiellement efficaces, selon les circonstances. Les débris ligneux et les chicots demeurent des composantes critiques de l'habitat pour plusieurs espèces et il est possible de les maintenir ou les créer afin d'améliorer les legs biologiques. La préparation des sites après la récolte, qui crée des legs biologiques survenant de concert avec les perturbations naturelles et permettant de conserver les débris ligneux grossiers, est susceptible d'aider à maintenir les espèces non cultivées. Le contrôle des espèces exotiques qui agissent comme plantes essentielles et/ou nuisibles peut réduire la contrainte physiologique et maintenir plus de ressources naturelles pour les espèces indigènes. Nous décrivons également les stratégies pour maintenir la biodiversité à l'échelle du paysage. En particulier, nous examinons certains avantages et certaines limites de l'aménagement basé sur les régimes de perturbations, les bandes riveraines tampons et les réserves de conservation en tant que moyens pour protéger la biodiversité.

Dans la partie IV, le rapport conclut en présentant un programme de recherche préliminaire pour compléter les recherches sur les taxons spécifiques recommandées à la partie II. Ce programme de recherche se fonde sur les revues de la littérature existante et sur les recherches en cours (qui comportent des biais géographiques et taxonomiques). L'objectif de la recherche proposée est d'améliorer la compréhension des moyens pour protéger la biodiversité dans les forêts aménagées. Il existe un besoin d'obtenir plus d'information écologique de base sur l'écologie des taxons riverains et il existe également un besoin en matière de recherche appliquée pour déterminer leur sensibilité envers les perturbations reliées à l'aménagement forestier.

MOTS CLÉS

Invertébrés aquatiques, biodiversité, oiseaux, tampon, régime de perturbation, espèces menacées d'extinction, poissons, zones forestières, champignons, espèces pivots ou essentielles, mammifères, plantes, bande riveraine, amphibiens aquatiques, végétation

AUTRES PUBLICATIONS DE NCASI DANS CE DOMAINE

Bulletin technique no. 885 (août 2004). *Managing elements of biodiversity in sustainable forestry programs: Status and utility of NatureServe's information resources to forest managers.*

Bulletin technique no. 857 (janvier 2003). *Wildlife and biodiversity metrics in forest certification system*s.

Bulletin technique no. 799 (janvier 2000). *Riparian vegetation effectiveness.*

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PART I PRINCIPLES AND CONCEPTS

1.0 INTRODUCTION

1.1 Purpose

In this report, we outline broad principles, information needs, and research directions to support biological diversity protection in riparian forests on private lands of the Pacific Northwest. We addressed the complex topic of riparian biodiversity by recognizing that a small set of key processes primarily structure ecosystems and their biological organization (Holling 1992). Primary controls on the biological diversity that can exist in a given area have been identified through literature review and critical analysis by the authors for their relevance to Pacific Northwest riparian forests and their potential application across taxa. These controls are habitat heterogeneity, legacy retention, physiological stress, and resource availability. We incorporate these into a simple graphical model for evaluating effects of disturbance regimes, including forestry practices, on biodiversity in general. Use of the model requires an understanding of how a particular disturbance affects these four variables. This will be influenced by regional climate and other factors that vary spatially; therefore, we describe important aspects of geographic variation in the Pacific Northwest (summarized in Appendices A and B). We illustrate how this geographic variation influences disturbance effects by comparing how several standard forestry disturbances would be predicted to affect biodiversity in wet temperate forests west of the Cascades vs. those forests found on the xeric eastside of these mountains. Because there are tradeoffs in managing for different elements of biodiversity, we also provide an analysis of the effects of forestry practices on different life history groups. Individual authors present short chapters summarizing biology for each group, followed by a discussion of documented or expected responses of each group to riparian forest management.

The question of how best to maintain biodiversity is complex, involving social as well as ecological concerns. We will not address these broader societal questions. Instead, our intent is to provide a sufficient framework for evaluating the effects of various forestry approaches on overall biodiversity, and to recognize where differential responses among elements of biodiversity will occur. In addition, we conclude the report by providing a research agenda to better inform riparian biodiversity conservation on private lands in the Pacific Northwest. Although we articulate general principles that will be applicable in most forests, our particular study area is the Pacific Northwest portion of Washington, Oregon, and California, extending eastward to the east slope of the Cascades.

1.2 The Concept of Biodiversity

Biodiversity is a general concept for species, habitat, and genetic diversity of all forms of life (Westman 1990; Hunter 1999). Diversity embodies the amount of all three per unit area (e.g., species richness), their equitability (evenness in relative abundance) (Whittaker 1975; Westman 1990), and maintenance of viable populations of a complete array of native species. Compositional, structural, and functional biodiversity have also been recognized (Roberts and Gilliam 1995). Much of the focus, in the literature and here, is on compositional biodiversity (species assemblages). Biodiversity protection may or may not be consistent with maximizing species richness at a given, particularly local, scale. Instead, it is about preventing biological impoverishment at multiple levels of organization.

Unfortunately, there may be no metric that tracks trends in biodiversity and works well for the wide range of settings or life forms occurring in natural landscapes (reviewed by Layton, Guynn, and Guynn 2003; NCSSF 2005). The occurrence or absence of certain species, groups of species, or other biophysical elements may all be indicators of the status of biodiversity. Loss of any species may be an important, simple indication of degradation of overall biodiversity, particularly if the species is one that regulates the abundance of others (e.g., keystone, symbiotic, or mutualistic species). Conversely, return of a species extirpated in the past may be an important, simple indication of recovering biodiversity. However, both extirpation of species or addition of species (via normal dispersal) in a given area can be unrelated to biodiversity trends or to management actions. Such contrasts are inherent in the dynamic nature of populations, as well as processes underlying biodiversity in riparian areas, complicating the analysis of forest management effects on biodiversity.

Non-equilibrium processes and population dynamics over variable spatial and temporal scales must be considered when addressing questions about biodiversity protection (Spies and Turner 1999; NCSSF 2005). Changes in climate, human land use, and management lead to non-equilibrium dynamics. For example, vegetation at a given location changes over time and so does the nature of its biodiversity. At certain scales, a landscape is composed of few to many units that may be in the same (large disturbance events) or different (small disturbance events) developmental stages. Biodiversity at scales larger than the stand is determined by how disturbance events are synchronized or otherwise juxtaposed at these larger scales. This is particularly complicated in riparian forests.

1.3 Complexity of Riparian Areas

Riparian areas are among the biosphere's most complex environments (Naiman, Bilby, and Bisson 2000). They possess distinct ecological characteristics resulting from the interaction between terrestrial and aquatic ecosystems (Gregory et al. 1991; Naiman, Bilby, and Bisson 2000). From a functional perspective, riparian areas are considered to extend outward from the stream channel to the limits of flooding or beyond (Gregory 1997) and upward to include the canopy of streamside vegetation (Swanson et al. 1982). The steep environmental gradients, dynamic nature, and geomorphic complexity in riparian areas combine to support a great abundance and variety of life (Naiman et al. 1992). Riparian areas can have a disproportionate effect on ecosystem processes through their influence on water quality, terrestrial wildlife, primary productivity, and aquatic food webs (Gregory et al. 1991; Naiman et al. 1992; Minore and Weatherly 1994). Forestry practices can modify the biophysical dynamics and thus biodiversity in riparian areas (Brinson and Verhoeven 1999).

1.3.1 *Riparian Forests of the Pacific Northwest*

Appendix A describes general riparian and associated upland vegetation in forestlands and how it varies across the study area. Major vegetation types recognized by Franklin and Dyrness (1988) for Washington and Oregon, and Barbour and Major (1977) for California, are summarized and dominant species listed. The zones recognized by Franklin and Dyrness (1988) apply northward into British Columbia and southeast Alaska. Only forests from the northwest corner of California, the Klamath region, are included; the remainder of the state is not considered to be in the Pacific Northwest.

A general trend in riparian and associated upland vegetation is an increasing contrast between the two along the gradient from wet coastal forests to the drier inland forest in the Klamath region and the eastside of the Cascades (Appendix A). This east-west geographic variation will be discussed throughout this report, particularly in terms of how it affects physiological stress and resource availability following disturbance.

2.0 FACTORS CONTROLLING RIPARIAN BIODIVERSITY

In large and complex regions such as the Pacific Northwest, a multi-scale hierarchy of controls governs the distributions of organisms in time and space (Bestelmeyer, Miller, and Wiens 2003; Sarr, Hibbs, and Huston 2005). Climate, topography and other abiotic factors impose the broadest controls on resource availability and physiological stress, which in turn are influenced by biological interactions. Resource availability is defined as the presence of growth resources such as mineral nutrients, light, and growing space that allow species establishment and growth. Stress has been defined generally as the physiological response of an individual, or the functional response of a system caused by disturbance or other ecological process relative to a reference condition. It is characterized by direction, magnitude, and persistence (Rykiel 1985). Understanding the effects of disturbance on physiological stress and resource availability is key to predicting the responses of biodiversity to forestry practices. Habitat heterogeneity and biological legacies left in the wake of disturbance are two additional keys to understanding this response.

2.1 Abiotic Processes and Physiological Controls

Temperature and moisture conditions provide the most fundamental constraints on organisms through their direct effects on photosynthesis, metabolism, net primary productivity, and other physiological processes whose limits govern species' distributions. These processes underlie vegetation formations at the broadest scales (Holdridge 1947; Sarr, Hibbs, and Huston 2005) and affect the distributions of animals as well (Currie 1991; Hansen and Rotella 1999). Variation and interaction among temperature, humidity, and radiation in both space and time are also important factors driving the developmental or reproductive biology of many species.

Climatic gradients in the Pacific Northwest are the steepest in North America (Franklin and Dyrness 1988), and field and modeling studies have demonstrated that many elements of forest ecosystems vary across the gradients, including vegetation composition (Ohmann and Spies 1998, Appendix A), canopy density (Grier and Running 1977), primary productivity and ratios of above to below ground biomass (Runyon et al. 1994). Riparian plant diversity and composition show corresponding variation across climate gradients in the Pacific Northwest (Pabst and Spies 1998, 1999) as well as predictable, directional species responses to climate-driven variation in disturbance (Sarr 2005). Climate and its interaction with other large-scale landscape characteristics such as environmental or evolutionary history may therefore be viewed as a primary set of controls on spatial patterns of riparian biodiversity. Such geographic controls are fundamental to understanding effects of forest management at different locales (see Sections 4.2 and 4.3).

At more local scales, variation in geology, topography, and other watershed characteristics form a secondary set of abiotic controls on species distributions. For example, Whittaker (1960) and Harrison, Viers, and Quinn (2000) observed that vascular plant species turnover with change in elevation (Beta diversity) occurred more rapidly on serpentine vs. granitic or other substrata. At the basin scale, variation in geomorphology, hydrology, and soils create a varied template for riparian and aquatic biodiversity (Leopold, Wolman, and Miller 1964; Swanson et al. 1988). Gradients in climate, stream power, and channel gradient from headwaters downstream provide habitat heterogeneity that differentiates the riparian vegetation in any given watershed (Hupp 1986; Gregory et al. 1991; Tabacchi et al. 1996). At the stream reach scale (10's to 100's of meters), such abiotic factors result in steep gradients in factors influencing physiological stress, such as microclimate and soil oxygenation, and resources such as soil moisture. These gradients further differentiate riparian areas. Physical variability in the environment fosters complexity in vegetation composition and structure, yielding distinctive habitat for other life forms.

2.2 Biotic Processes and Resource Availability

At the scale of a riparian forest stand or stream reach, biotic processes become key controls on biodiversity. Biotic processes that result in pulses of nutrients and/or available light have an important influence on resource availability (Pollock, Naiman, and Hanley 1998). Biotic processes structuring vegetation increase habitat heterogeneity by creating more vertically, horizontally, and compositionally complex vegetation (MacArthur and MacArthur 1961; Moran 1980). Interactions among organisms, such as disease-mediated interactions and competition for limited resources, are key biotic processes affecting local diversity. Diseases caused by exotic pathogens may cause atypically high mortality, leading to local extinction, for example, the root rot (*Phytophthora lateralis*) to which Port Orford cedar has no natural resistance (Hansen et al. 2000; Jules et al. 2002). Non-native diseases can act as artificial keystone species (see Section 8.1) not only by having such strong pathogenic effects, but also by their competitive ability to displace native species (Elton 1958). Similarly, large vertebrate consumers with low population sizes are especially vulnerable to displacement and have been lost from many environments even where suitable habitat may exist (Duffy 2003). Such vulnerability appears to be trophically mediated, with important implications for conservation.

Competition among native species is perhaps the most important biological regulator of local diversity. It is the process that leads to dominance/equitability relationships that in turn affect the number of species that can coexist in an area (Whittaker 1975). In many cases, competition among similar functional groups of species is strongly linked to resource availability, with resource-rich environments fostering intense interspecific competition (Huston 1994). The factors that increase or decrease the rates of competitive exclusion, especially disturbance, are integral to models of diversity described in Section 3.0.

2.3 Disturbance Processes

Disturbance may be defined as any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment (Pickett and White 1985). Riparian forests are influenced by the most complex disturbance regimes in the Pacific Northwest, juxtaposing mass failure, fluvial, ice, fire, gap, herbivore, and pathogen-mediated disturbances (Gregory et al. 1991; Pollock, Naiman, and Hanley 1998; Naiman, Bilby, and Bisson 2000). Disturbance affects resource availability and physiological stress directly. Disturbances of variable area, frequency, and intensity also enhance habitat heterogeneity and regulate the dominance of highly competitive species, reducing competitive exclusion. Both effects will tend to increase diversity (Huston 1994; Spies and Turner 1999, Section 3.0). The two effects also partially explain why riparian areas usually have higher diversity than adjacent uplands. Finally, the maintenance of many species across cycles of disturbance requires persistence of biological legacies such as regeneration propagules (Odion and Davis 2000), or resources such as woody debris (Lindenmayer and Franklin 2002).

There is a developing conceptual framework for riparian ecosystems in the Pacific Northwest whereby geophysical disturbances, driven by direct and indirect abiotic effects, establish a dynamic mosaic of surfaces (e.g., channels, channel units, floodplains, terraces, alluvial fans) that may act as a template for riparian biodiversity (Grant and Swanson 1995; Fetherston, Naiman, and Bilby1995; Swanson et al. 1988; Pollock, Naiman, and Hanley 1998; Johnson, Swanston, and McGee 2000). The importance of different geophysical disturbances changes with stream order and channel gradient. Along larger streams flooding is more important; at headwater streams, landslides and debris flows dominate (Montgomery 1999). These processes operate at time scales ranging from months to years for chronic disturbances such as flooding to decades or centuries for episodic events such as debris flows. Likewise, spatial scales range from local to landscape-wide.

The fluvial/mass movement processes that structure riparian areas are controlled by basin geology, hydrology, and inputs of inorganic and organic material from adjacent slopes (Gregory et al. 1991). In the Pacific Northwest, large, infrequent landslides may play a dominant role in distributing woody debris from upland areas to streams (Reeves et al. 1995). The importance of large wood as a habitat feature supporting a number of elements of biodiversity has been well established in this region (Harmon et al. Franklin 1986; Naiman, Bilby, and Bisson 2000; Johnson et al. 2000). The template for biodiversity created by these geophysical disturbances is heterogeneous in geomorphology, soils, and vegetation composition and structure (Gregory et al. 1991; Pollock, Naiman, and Hanley 1998).

Fire will also directly and indirectly influence the riparian biodiversity template. Runoff and sedimentation processes increase after fire as a function of fire severity, the proportion of a watershed that burns, and post-burn rainfall patterns (Swanson 1981). Fire removes litter and vegetation in direct proportion to its severity. In addition, high severity fire can increase water repellency and decrease root strength. Increases in sedimentation with fire are caused by not only mass soil movements, but also by surface erosion processes (McNabb and Swanson 1990). Fire-induced tree mortality creates woody debris that is readily transported downslope by fluvial and mass movement events that occur more frequently following fires (Swanson et al. 1987; McNabb and Swanson 1990). These processes, and increased summer streamflows that often follow fire, both have a variety of effects on aquatic species.

Within riparian areas in the Pacific Northwest, considerable uncertainty exists about more direct effects of fire due to a lack of fire history information specific to streamside areas (reviewed by Dwire and Kauffman 2003). Recent studies in the Klamath region and eastern Cascades have found that fire return intervals in riparian reserves are more variable than in adjacent uplands and tend to be longer (Poage 1994; Everett et al. 2001; Skinner 2003). By acting as occasional barriers to fire spread, riparian areas may enhance the spatial and temporal diversity at the watershed scale. However, under favorable conditions for combustion, fire may readily spread through riparian areas as shown by physical evidence of continuity in fire disturbance among riparian areas and uplands on both sides (Poage 1994; Everett et al. 2001). Weather, fuel moisture, width of stream, topography, orientation of riparian areas relative to prevailing wind, fire intensity in upslope areas and other factors will affect the probability of fire crossing over riparian areas (Agee 1993).

Local disturbances that open gaps in the forest canopy are also important in Pacific Northwest forests in terms of affected area over time. Wind, disease, and insects are among the agents that create gaps. Gap creation rates for upslope forests range between 0.2% and 2% of a stand each year, which is equivalent to a rotation period of 50-500 years (Runkle 1985; Spies, Franklin, and Klopsch 1990). Gaps may cover 5-30% of a forest and affect 50% of the area at any given time. These gap-forming disturbances, though believed to be less common in riparian areas than upslope, are nonetheless widespread and are important for local plant diversity and tree regeneration (Sarr 2005). Many animals are also believed to respond to this fine scale heterogeneity in forests, including birds (Section 7.5), mammals (Section 7.4) and invertebrates (Section 7.6).

Wind disturbances also open large patches in forests (Hansen and Rotella 1999; Stinton et al. 2000). For example, in the western Cascades, Stinton et al. (2000) found that 10% of a landscape was affected by windthrow from 1890 through the late 1990s, and that less area was affected per year prior to onset of timber harvest. Climate, landform, stand conditions, and other disturbances, including timber harvest, will increase the frequency of windthrow events.

Figure 2.1 shows the relative frequency of both episodic (fire) and chronic (gap, small floods) disturbances in a hypothetical riparian forest over a 200-year period (episodic debris flows occur over longer time scales in an area this size, Swanson et al. 1987). The general pattern of relatively continuous disturbance of different types illustrates the dynamic nature of riparian areas over relatively short time scales. In this example, the rotation intervals (time it takes for the whole 500 ha to be affected) for the different disturbances are 80.7, 85.5, 82.8, and 602.4 years for gap, fire, flood, and windthrow disturbances, respectively.

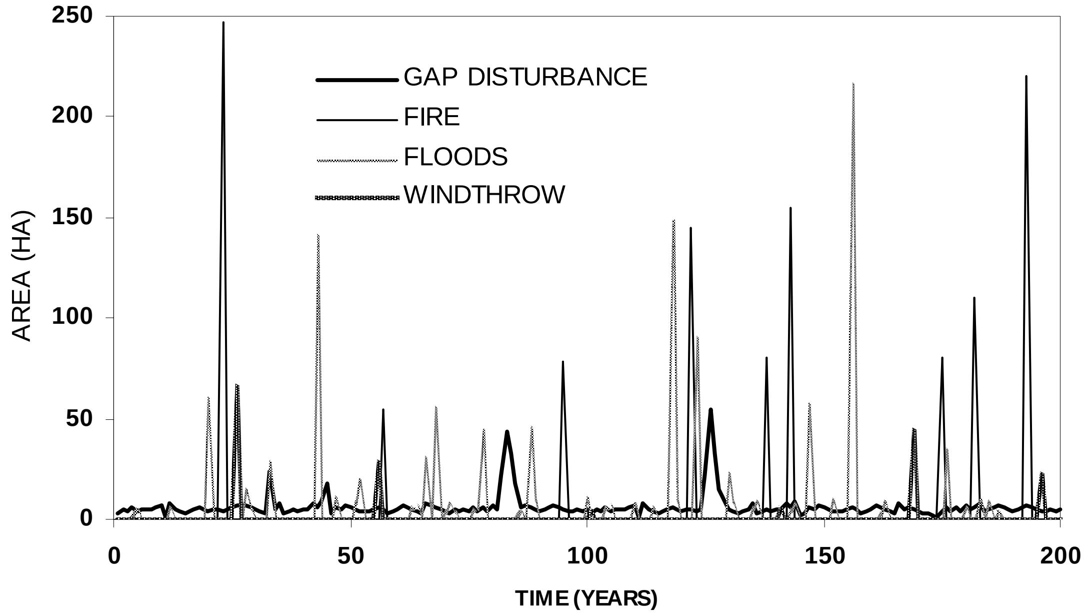

Figure 2.1 Hypothesized Disturbance Regime for a 500-Hectare Riparian Forest Based on Typical Mid-Elevation Disturbance Frequencies in the Study Area (see also Appendix B)

Although a similar degree of *relative* importance in these disturbances is likely to occur in riparian areas over much of the Pacific Northwest, the frequency, severity, and size of disturbances shown in Figure 2.1 all vary considerably across the study area. Understanding variations in natural disturbance regimes of an area can help to predict how forestry practices will interact with and affect biodiversity. In addition, aligning timber harvest disturbances to more closely emulate natural disturbances is an approach for improving biodiversity protection (see Section 11.1). Appendix B summarizes variation in large disturbances in the Pacific Northwest. Managers will need more specific information on their local areas and to keep in mind the points described below.

Fluvial and mass movement processes vary greatly throughout the Pacific Northwest in response to geologic and climatic factors (Swanson et al. 1987). The episodic nature of these disturbances makes their regimes difficult to characterize. A key factor is the potential for significant rain-on-snow events in different areas (Appendix B), which produce especially pronounced peak flows and associated effects.

Figures reported in Appendix B are not from riparian areas, but rather from uplands, which often burn more frequently. Reported figures are based on sampling fire scars on trees, methods whose precision is not well described and which are biased toward more frequent fire and toward surface fire (Baker and Ehle 2001; Whitlock 2004). Between the crown fire systems of higher elevations and the northwest coast, and the open forests of the east side of the Cascades there are complex mixed severity landscapes (Agee 1993), structured by patchy crown fires. The dynamics of these landscapes are poorly understood (Odion et al. 2004). This is made difficult by the variable nature of crown fire occurrences (Turner and Romme 1994).

3.0 CONCEPTUAL MODELS OF DISTURBANCE AND SPECIES DIVERSITY

It is impossible to describe succinctly the effects of disturbances on all aspects of biological diversity simultaneously. Species will show disparate responses to any disturbance given differences in sensitivity to disturbance frequency or intensity, specific requirements for habitat structure, and variation in dispersal and competitive ability. Even these fundamental elements of disturbance response are poorly understood for the majority of species occurring in riparian forests.

Nonetheless, extant conceptual models provide a theoretical foundation for evaluating landscape level species diversity in relation to disturbance. These models have broad empirical and theoretical support. They not only provide a starting point for posing hypotheses of disturbance responses for groups of species, but we use them as a basis for developing our conceptual model for predicting effects of forestry disturbances (Section 4).

According to Connell (1978), ecological communities of sessile organisms are composed of early seral species that colonize quickly after disturbance and late seral species that increase in abundance and dominance with time since disturbance. Maximum diversity occurs, therefore, when disturbance is sufficiently frequent to limit dominance, while allowing ample time for colonization by all species (Connell 1978; Sousa 1979; Petraitis, Latham, and Niesenbaum 1989) (Figure 3.1). Coexistence mechanisms based on the Intermediate Disturbance Hypothesis have been advanced in recent publications (reviewed by Roxburgh, Shea, and Wilson 2004).

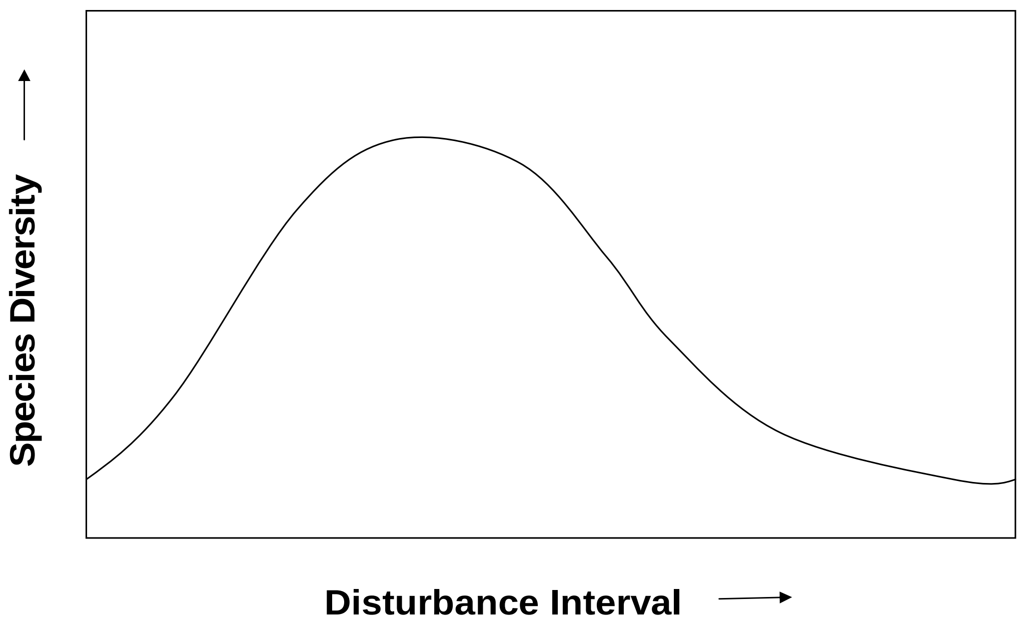

Figure 3.1 Relationship between Disturbance Frequency and Species Diversity (after Connell 1978)

The closely related *Dynamic-Equilibrium Hypothesis* (DEH) of species diversity (Huston 1979) linked the Intermediate Disturbance Hypothesis with observations that species richness is often highest at intermediate productivities (Grime 1973). Huston (1979) proposed that the relationships between disturbance and local diversity depend upon site quality, because competitive exclusion depends upon *both* the disturbance regime and the rate at which dominance develops. The result is a response surface that predicts richness along axes of productivity and disturbance frequency and or intensity showing why diversity varies at the landscape scale (Figure 3.2). The Dynamic-Equilibrium Hypothesis has received empirical support in grassland ecosystems, tropical rainforests, and Pacific Northwest wetlands (Sarr, Hibbs, and Huston 2005)

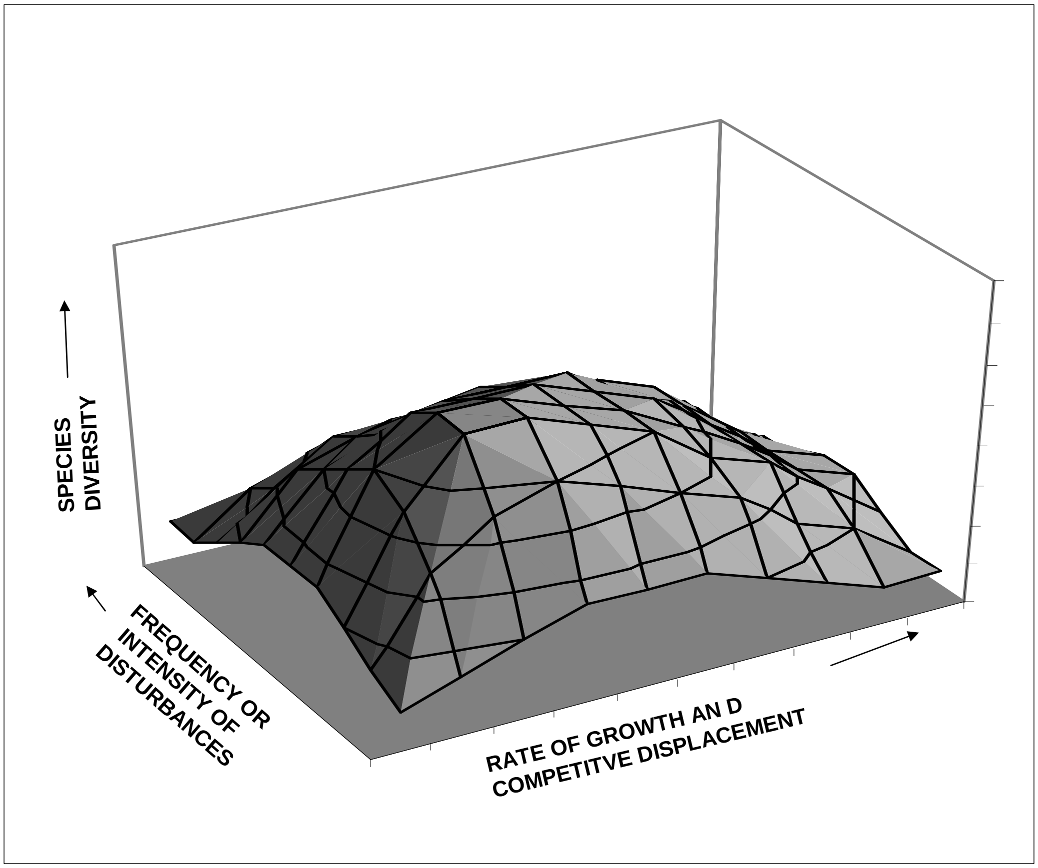

Figure 3.2 Species Diversity as a Function of Disturbance Frequency or Intensity and Competitive Displacement (after Huston 1979) (Rates of growth and competitive displacement are hypothesized to increase with rising site productivity.)

Existing conceptual models of local diversity echo a common theme: there is a relationship between disturbance and diversity, where intermediate levels of disturbance yield highest richness (Sarr, Hibbs, and Huston 2005). The Dynamic Equilibrium Hypothesis is perhaps the most robust model in that it explicitly links productivity to disturbance dynamics. Huston (1999) recognized that the models predicting highest diversity at intermediate levels of disturbance and productivity apply primarily to species within similar functional groups competing for limiting resources (e.g., vascular plants, intertidal organisms, planktonic algae). He speculated that different functional groups would respond distinctively to gradients of productivity and disturbance, with primary producers peaking in species richness at relatively low levels of productivity, because of the importance of competitive displacement at higher productivity levels. Conversely, species richness in upper trophic levels would be greatest at relatively high levels of ecosystem productivity.

Huston (1994, 1999) further developed these ideas to categorize landscapes by ecosystem productivity and disturbance regime and to recognize inherent differences in the patterns of diversity of distinct life forms. Using the DEH, Huston (1999) distinguished four landscape types involving combinations of high or low productivity and disturbance. In the high productivity, low disturbance scenario, Huston (1999) predicted low landscape heterogeneity due to rapid revegetation following disturbance. He predicted such landscapes would have low vascular plant richness, but possibly high richness of higher life forms, especially sensitive groups of species. An example of this type in the Pacific Northwest might include mesic old growth upland forests. High productivity, high disturbance landscapes should have higher levels of plant richness and relatively high richness of other life forms due to the high heterogeneity maintained by disturbance regimes. Riparian forests would likely fit into this landscape category. In low productivity, low disturbance landscapes, Huston predicted diverse and distinctive plant communities, but relatively low diversity of higher trophic groups. Huston cited tropical rainforests on infertile soils, or semi-arid woodlands as examples of this group. Other examples might include exceptionally infertile sites, such as the distinctive woodlands occurring on serpentine soils. The final landscape type, low productivity, high disturbance is predicted to be a low diversity type for all groups.

Although the DEH provides a powerful general model for predicting species diversity, the model and its derivatives must still be used with caution. It probably applies best for vascular plants (but see Currie 1991); other groups may respond differently than implied by Figures 3.1 and 3.2 (see Section 9), and there is less known about diversity patterns for other groups, particularly fungi, invertebrates, and non-vascular plants. The model also implies that ecosystem productivity and disturbance are the primary factors governing spatial patterns of biodiversity. There is a need to incorporate habitat heterogeneity and biological legacies left after disturbance into biodiversity predictions.

3.1 Habitat Heterogeneity, Biological Legacies, and Biodiversity Predictions

Heterogeneity is a multiscale concept encompassing a tremendous range of physical and biological complexity. Not only does diversity show a positive relationship with habitat heterogeneity (Kerr and Packer 1997; Huston 1994; Roxburgh, Shea, and Wilson 2004), but also viewed across taxa this relationship holds at the full range of scales, making the value of habitat heterogeneity for biodiversity one of the most reliable generalizations in conservation biology. Therefore, we incorporate heterogeneity along with the DEH as an organizing factor in the model we present in Section 4.

The habitat heterogeneity of a riparian forest varies across a range of spatial and temporal scales. The relative importance of such environmental variation is dependent upon the size, mobility, trophic status, and life span of the organism. Where the distribution and abundance of forest bryophytes might respond to fine scale variation in the chemistry or texture of tree bark, an elk herd may be influenced more directly by the much coarser scale patchiness of forest openings at the landscape scale.

Natural disturbance regimes juxtapose complex spatial and temporal patterns of disturbance upon landscapes and create not only an array of competitive environments, but also sharply contrasting abiotic conditions and habitat structures. Thus, physical complexity in environments (see Section 2.1) interacts with temporal variability created by disturbances (Section 2.3) to produce much of the habitat heterogeneity underlying biodiversity. Within forest stands, complexity in vegetation and soils form additional heterogeneity. Generally, old-growth riparian stands, with ongoing tree mortality and establishment, contain the most complex suites of living and dead structures. These include live boles and branches of different forms, snags, downed logs, deep organic soils with occasional tip-up mounds, densely wooded areas and hardwood-, shrub-, or herb-filled gaps. Although the juxtaposition of compositional and structural characteristic of old forests is believed to be an important template for biological diversity, complexity can also arise from large but incomplete disturbances, such as low to moderate severity fire, pathogen outbreaks, or windthrow that leave patches of living and dead trees, interspersed with disturbed areas of various sizes and degrees of mortality. Many forms of disturbance can positively affect heterogeneity if they are non-uniform in spatial extent, asynchronous in return interval, and variable in intensity.

Indigenous disturbance processes over evolutionary time have given rise to ecosystem resiliency, which is essential for the maintenance of biological diversity. This resilience is caused by natural succession processes that are in turn strongly dependent upon the biological legacies such as green seed trees, large wood, seed and spore banks, and resprouting organs that remain through disturbances (Perry 1994; White and Jentsch 2001; Franklin et al. 2002). Ecosystem recovery and ecological “memory” (the ability to return to a former condition through endogenous properties (Peterson 2002)) can depend entirely on legacies such as seed banks (Odion and Davis 2000). Legacies include resources stored by organisms during favorable conditions, which can be used for growth following disturbance. This “storage effect” has been identified as an important disturbance mediated coexistence mechanism (Warner and Chesson 1985; Chesson 2000; Roxburgh, Shea, and Wilson 2004). These essential functions of stored resources that can survive disturbance make retention of pre-disturbance biological legacies a fundamental consideration for biodiversity maintenance in managed riparian landscapes. Thus, we incorporate legacy retention with habitat heterogeneity, competitive exclusion, and physiological stress as primary factors to consider in evaluating disturbance and diversity relationships.

3.2 Intensity, Frequency, and Size of Disturbances

The Intermediate Disturbance and Dynamic Equilibrium Hypotheses do not provide a framework for evaluating differences in the interrelated nature, scale, and timing of disturbances, and how these may affect legacy retention, habitat heterogeneity, competitive exclusion and stress. Properties of natural disturbance regimes and their interrelated effects have been discussed by Miller (1982) and Malanson (1984). There are gradients in the following disturbance properties that affect the primary controls on biodiversity.

Intensity. Amount and kinds of vegetation killed, growing space made available, and biological legacies that remain. For forestry-related disturbance, this will be a reflection of the amount of existing forest, understory vegetation, forest floor and soil removed by harvests. In terms of forestry disturbances in riparian areas, proximity to stream may be an important component of disturbance intensity because natural disturbance levels are likely to be greater nearest the stream. Disturbance intensity is generally inversely correlated with frequency (Malanson 1984).

Frequency. Amount of time between disturbances. This is often expressed as an average (e.g., mean fire return interval); however, the time between disturbances is often highly variable, and so the range can be more meaningful.

Size. Geographic extent of a disturbance. This will also depend on frequency. Both are needed to calculate the amount of area that will be disturbed over a given time period.

Based on conceptual models of species diversity, disturbance regimes that are intermediate in characteristics may maximize diverse assemblages of species. Therefore, one metric for analyzing biodiversity protection is the extent to which disturbance effects are within the range of intermediacy. Because “intermediate” may imply central tendencies, it is important to emphasize that intermediate disturbance regime management does not require that intensity, size, and frequency each be maintained separately at intermediate or average levels. Relative extremes in disturbances and stochasticity are often important to coexistence and diversity (Christensen 1991; Gaines and Denny 1993; Clark et al. 2003). Instead, if any one dimension of disturbance, e.g., intensity, is high, the other two would need to compensate (e.g., frequency and size cannot also be high). To maintain intermediacy, intense disturbances can be infrequent and/or small, high frequency disturbances can be small and/or of low intensity, and large disturbances can be infrequent and/or low in intensity (Miller 1982; Malanson 1984). Maintaining biodiversity will be most difficult where two or especially all three disturbance regime properties are outside the bounds of intermediacy.

While the concept of intermediacy in disturbance regimes is a primary nonequilibrium explanation for high levels of diversity, the dimensions of intermediate regimes are qualitative (Shea, Roxburgh, and Rauschert 2004). Intermediate disturbance regimes may approximate those that have historically occurred in terms of balancing intensity, size, and frequency. Thus, explicitly managing based on historic disturbance regimes can be an approach for maintaining an intermediate disturbance regime (Section 11.1). However, historic range of variability is hard to measure, and is dependent on the time frame chosen as a reference. A recent review on biodiversity and sustainable forestry suggests that the historic range of variability concept must be updated and adapted with new information to be useful (NCSSF 2005). Data about climate change, invasive species, fragmentation, and other modern circumstances should be considered. In addition, there is great variability in historic disturbance regimes across the Pacific Northwest (Appendix B). Human impacts have both suppressed and increased disturbance. The entire regime of disturbances, including those of human origin, must be evaluated to determine how biodiversity will be affected. The model presented in Section 4 was prepared for this purpose.

The amount of legacy removal may be one measure of disturbance intensity. It is important to consider relatively apparent disturbance effects to aboveground legacies as well as less apparent effects to litter, soil, and belowground legacies. Ground-based harvesting and yarding systems can cause soil disturbances and compaction. The intensity of such disturbances will depend on the moisture content of the soil, size of material to be removed, equipment characteristics, number of passes of the equipment over the same area, depth of litter, soil type, and slope. The effects of timber harvest on legacies in soil and litter have been described elsewhere (Isaac, Hopkins, and Howard 1937; Graecen and Sands 1980; Wert and Thomas 1981; Geppert, Lorenz, and Larson 1985; Perry et al. 1989; Poff 1996; Pilz and Perry 1984; Perry and Amaranthus 1997; Hagerman et al. 1999; Neary, DeBano, and Ffolliott 2000; Byrd et al. 2000; Beschta et al. 2004; Karr et al. 2004; Sections 7.3 and 7.6). These effects do not always result in significant disturbances because they may be prevented through the use of aerial yarding and other low impact methods.

Where forestry practices include slash burning, there will also be effects on legacies in soils. The prolonged soil heating produced by smoldering combustion has been found to kill soil-stored seed and soil organisms, and to consume soil organic matter that supports soil biota. A wide variety of literature also addresses how soil legacies may be affected by slash or other surface fuel burns (Isaac, Hopkins, and Howard 1937; Pilz and Perry 1984; Frandsen and Ryan 1986; Albini et al. 1996; Poff 1996; Neary et al. 1999; Neary, DeBano, and Ffolliott 2000; Odion and Davis 2000; Brown, Reinhardt, and Kramer 2003; Korb, Johnson, and Covington 2004; Sections 7.3 and 7.6). Effects will vary considerably depending on fuel load, moisture, particle sizes, and whether broadcast or pile burning methods are used. Disturbances from forestry roads are another consideration. Roads increase the frequency and regularity of sediment inputs into streams (Section 6), and can lead to landslides (Jones et al. 2000, reviewed by Forman et al. 2003). Road-related disturbances vary considerably depending on characteristics of roads and the environment in which they are placed (Forman et al. 2003).

In sum, technology and techniques for harvesting and transporting trees, and treating harvested sites, vary and have changed over time. There are many options, leading to a wide range in net disturbance intensity. There are also effects that are particular to certain life history groups (Sections 6 and 7), geography, and site-specific circumstances. These and other factors need to be considered when evaluating the intensity of forestry disturbances, making a case-by-case approach necessary.

4.0 A CONCEPTUAL MODEL FOR EVALUATING DISTURBANCE EFFECTS ON BIODIVERSITY

When the primary controls on local and regional species diversity are considered in concert, biodiversity may be best protected where intermediate disturbance regime effects prevent uniformly stressful circumstances and/or dominance leading to competitive exclusion, habitat heterogeneity, and retention of functional legacies (Figure 4.1). The conceptual model shown in Figure 4.1 can be used to evaluate disturbance effects, and further how they may be manipulated to influence biodiversity. Note that the importance of both additional habitat heterogeneity and biological legacy retention is assumed to decrease with increasing levels of these (i.e., the relationship is assumed to be asymptotic). This is a key information gap that research needs to address (Section 14.1).

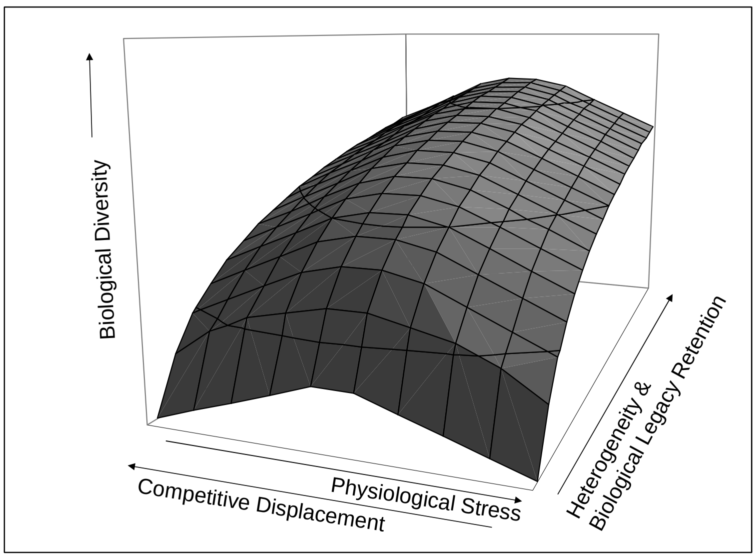

Figure 4.1 Conceptual Model for Evaluating How Forestry Disturbance May Influence Biodiversity (based on species diversity concepts)

The x-axis in Figure 4.1 is complicated because the effect of disturbance on resource availability or physiological stress depends upon the sensitivity of the taxon group of interest as well as geographic setting. However, it is possible to make predictions of biodiversity responses to disturbance in different settings (Section 4.2), but it is vital to first understand aspects of geographic variation in abiotic and biotic factors.

4.1 Linking Effects of Geographic Variation

Across large spatial extents, variation in climate causes not only regionalization in vegetation composition and structure, but also gradients in moisture, temperature, canopy architecture, woody debris recruitment, and understory light. Climate also manifests sharp contrasts in riparian forest productivity across the study area, with corresponding effects on species diversity. From west to east, local woody plant richness shows unimodal relationship with site productivity, with low values in both the highly productive Oregon Coast Range, where biotic control is greatest, and in the least productive eastern Cascades shrub-steppe where abiotic control is strongest (Waring et al. 2002). Highest local woody plant richness occurs in the somewhat intermediate southern Cascades and Siskiyou Mountains, where both controls are moderate (Ohmann and Spies 1998; Waring et al. 2002) (Figure 4.2). Local scale studies in western Oregon support the hypothesis that competition is a strong control on local plant richness in productive forests. For example, old-growth forests with active gap processes, silviculturally thinned stands, and early seral communities all harbor higher plant species richness than dense, even-aged conifer stands (Schoonmaker and McKee 1988; Stewart 1988; Bailey et al. 1998), especially fertilized plantations (Thomas et al. 1999).

The potential for positive effects of natural disturbances (and possibly silvicultural treatments) on richness will be greatest in productive riparian forests, such as occur in western Cascades and Coast Ranges of the Pacific Northwest, where competitive pressures are believed to be strongly governing local plant diversity (Pabst and Spies 1998, 1999; Hibbs and Bower 2001; Waring et al. 2002). In relatively unproductive forests of the eastern Cascades or Siskiyous, in contrast, a similar disturbance might add to physiological stress and lead to lower local diversity. There, plant diversity is locally high in undisturbed forests where local drought or infertility limits overstory density (Whittaker 1960; Stewart 1988).

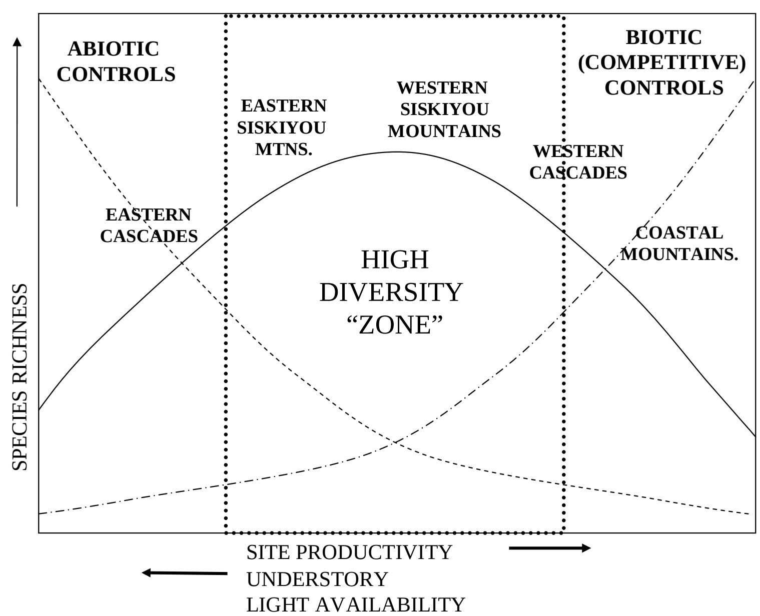

Figure 4.2 Conceptual Relationship between Local Riparian Vascular Plant Species Richness (solid line) and Site Productivity at Riparian Forest Sites across the Study Region (Dashed line shows abiotic controls, dot-dashed line shows biotic controls.)

From these general climatic patterns and empirical relationships among climate, productivity and forest structure (Appendix A), and the importance of biotic and abiotic processes (Figure 4.2), we constructed Figure 4.3, which illustrates the hypothesized relative importance of habitat heterogeneity, legacy retention, resource availability, and physiological stress across the climatic gradient.

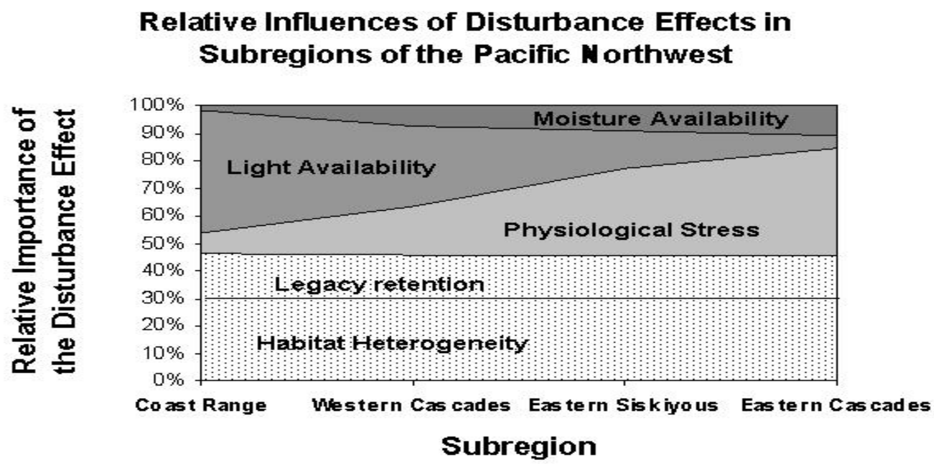

Figure 4.3 Relative Importance of Disturbance Effects on Factors Affecting Species Diversity in Sub-Regions of the Pacific Northwest

4.2 Evaluating the Effect of Riparian Forest Management on Biodiversity across the Pacific Northwest

In preceding sections, we developed a model for evaluating disturbance/biodiversity relationships using basic ecological principles. Here we describe how to make predictions about forestry disturbances in riparian zones using this model. Geographic variation in disturbance-associated stress and resource availability must also be accounted for. To illustrate this, we evaluate the model here for two relative geographic extremes in the region. These extremes are the wet temperate riparian forests west of the Cascades, and the xeric, interior riparian forests on the east slope of the Cascades and extreme easternmost Siskiyou Mountains. As a starting point, we will assume that forests in both areas are 30-60 years old, conifer-dominated, and regenerated post-harvest. There are many other scenarios, but these are likely to be relatively common in riparian forests of private timberland.

It is important to emphasize that life history traits (Sections 6 and 7) will cause variation in how different taxonomic groups respond to disturbance, particularly in terms of physiological stress. We discuss this further in Section 9.0. Nonetheless, this comparison should illustrate how different forestry disturbances affect biodiversity maintenance in varied geographic settings.

In Table 4.1, we evaluate several forest management options and rank their potential to maintain biodiversity based on our conceptual model. The management approaches comprise a gradient in intensity from no touch buffers to even-aged, short rotation clearcutting. It is assumed in no touch buffers that managers will prevent natural disturbances like fire and insect epidemics to the maximum extent possible. In the numerical rankings in the table, heterogeneity both at local and larger scales contributes to biodiversity maintenance. Therefore, heterogeneity is more heavily weighted. Relative potential for biodiversity maintenance increases with numerical value calculated as the row sum in the table with low=1, medium=2, and high=3 for heterogeneity and legacy columns, and low and high =1 and medium =3 for resource availability and physiological stress. Values ranging from 1-2 or 2-3 were given the average (i.e., low-medium = 1.5 for legacy retention).

Table 4.1 Effects of Different Forest Management Options on the Fundamental Drivers of Biodiversity and Their Relative Potential for Maintaining Biodiversity in 30-60-Year-Old, Second Growth Riparian Forest Landscapes on the Wet Temperate West Side (W) and Xeric East side (E) of the Cascades

	RANKING CRITERIA					Potential for Biodiversity Maintenance
MANAGEMENT APPROACH	Local Heterogeneity	Landscape Heterogeneity	Legacy Retention	Resource Availability	Physiological Stress	Sum
Protection from disturbance (no touch buffer)	W&E-Low	W-Low E-Low-Medium	W&E-High	W- Low E -Low-Medium	W&E-Low	W=7 E=8.5
Uneven-aged Single tree selection	W-Low-Medium E- Medium	W-Low E-Low-Medium	W&E-Medium-High	W-Low- E-Low-Medium	W-Low E-Low-Medium	W=7 E=10
Uneven-aged Thinning from below	W&E-Medium-High	W&E-Low	W&E-Medium-High	W-Low E-Medium	W-Low-Medium E-Medium	W=9 E=12
Uneven-aged Small patch selection	W&E-Medium	W&E-High	W&E-Medium	W&E-Medium	W-Low-Medium E-Medium-High	W=12 E=12
Even-aged shelterwood	W&E-Medium	W&E-Medium	W&E-Low-Medium	W&E-High	W-Medium-High E-High	W=8.5 E=7.5
Even-aged clearcut (80-120 years)	W&E-Low	W&E-Medium	W&E-Low	W&E-High	W-Medium-High E-High	W=7 E=6
Even-aged clearcut (40-60 years)	W&E-Low	W&E-Low to Medium	W&E-Low	W&E-High	W-Medium to High E-High	W=6.5 E=5.5

Based on the rankings in Table 4.1, a productive westside riparian forest with little or no disturbance (single tree selection or no touch buffer) and a forest subjected to short-rotation clearcutting would harbor the lowest potential for biodiversity maintenance. In the former case, this would be expected because of low heterogeneity and resource availability resulting from strong dominance by the young conifers. With short-rotation clearcutting, low heterogeneity, low legacy retention, and possibly high physiological stress would be the limiting factors. Small patch harvesting, followed by thinning from below, and even-aged shelterwood are the forest management actions that ranked best in terms of maintaining biodiversity in the westside example. These management options would likely allow maintenance of biodiversity by adding heterogeneity and increasing resource availability, while maintaining biological legacies. In the exceptionally mild environments of the coastal mountains, where sites quickly reforest following disturbance, relatively severe disturbances (shelterwood) may be consistent with the maintenance of biodiversity, so long as they are not too frequent and retain sufficient legacy features and heterogeneity remain to provide refugia for sensitive taxa.

In the eastside environment, small patch selection and thinning from below ranked best for biodiversity maintenance. Several important differences are likely to be associated with forest management in eastside vs. westside riparian forests. First, greater ambient fluctuations in temperature and insolation increase the importance of the forest for moderating temperature extremes on the eastside. Second, the forests are less likely to develop as densely and understory light may be relatively less limiting than on the westside (Grier and Running 1977). In addition, other forms of heterogeneity (e.g., topographic moisture gradients with upslope areas) are increasingly important for differentiating habitats in eastside forests (Appendix A). Consequently, undisturbed stands may be less uniformly dominated by the overstory conifers and less dependent upon disturbance for maintenance of biodiversity. At the same time, the negative effects of moderate and severe disturbances on microclimate are more likely, so even-aged systems (shelterwood, clearcutting) would likely create stressful environments and lead to other stresses on biodiversity, such as stream heating. As in westside forests, the intermediate intensity management techniques blending legacy retention, moderate microclimatic stress and resource availability, and moderate to high local and landscape heterogeneity appear likely to be most appropriate for biodiversity maintenance. Thus, despite geographic differences, a number of intermediate intensity forest management approaches may be consistent with the maintenance of biodiversity, as predicted in Section 3.2.

The principles articulated here should provide a consistent conceptual basis to evaluate both natural and human disturbances. Up to this point we have kept the discussion intentionally general, discussing biodiversity in it broadest sense. Some groups may thrive on the resources liberated by disturbance, whereas others are more closely linked to the protected environment of established riparian forests. It is likely that managers will have interest in the evaluation of forest management effects on specific taxonomic groups, such as those that are imperiled (spotted owls, murrelets), or have great economic or societal interest (salmon). For such analyses, a much more detailed understanding of life history characteristics is desirable. In Sections 6 and 7, we summarize effects of riparian forest management on several different taxa groups to illustrate the important interactions between management and species' life history.

5.0 POLICIES FOR PROTECTING RIPARIAN AREAS

Prior to 1970, there were no streamside protection policies related to timber harvest. Since 1970, numerous state and federal regulations have been developed and modified, and future modification appears likely. The complex, context-dependent regulations are not detailed here. Recent reviews are available: Gregory (1997) and Young (2000).

Riparian management policies that have been developed in the last decade or so focus on a) widths of riparian management zones, b) retention of living and dead trees within the riparian zone, c) extent of shade, d) floodplain protection, e) yarding corridors, f) culvert dimensions, g) road crossings, h) felling techniques, and i) erosion protection (Gregory 1997). The riparian management zone may be much narrower than the riparian area based on a functional definition of that area, and the criteria used in the definition (Gregory 1997).

Regulations for private landowners differ considerably among the states of Alaska, Washington, Oregon, and California. Standards are further modified within the states based on regional conditions. All jurisdictions protect all but ephemeral streams using riparian management zones, but the zones differ in size and what is allowed within them. The state of Oregon has a policy of protecting streambanks with a no-harvest zone extending 3.5 m from the edge of the stream. Alaska allows no harvest within 30 m of unconstrained anadromous fish bearing streams. Criteria for what must be retained after harvest differ. It should be noted that while riparian management zone guidelines do not necessarily severely restrict activities in riparian areas, these guidelines might often be subordinated where there are water quality issues or species protected under state or federal law.

Riparian guidelines lack an explicit vision of desired future conditions or dynamics. Such a vision would be highly context-dependent. Young (2000) notes that because existing guidelines allow the harvest of large conifers, typical harvest rotations of 60-80 years under existing guidelines can eliminate natural riparian sources of *large* woody debris resources from streams. This illustrates how resource availability in many cases may be under the discretion of the landowner. In many state and private lands, past harvest has already eliminated the larger sources of woody debris. Near-term approaches to increasing large woody debris, regulatory or not, could involve bringing it in from outside the riparian area. Riparian management guidelines in Oregon include an active management option that uses a basal area credit system, which allows for increased volume of harvest where logs are placed in streams and for other resource enhancements.

Gregory (1997) notes that the basis for selection of specific numerical criteria may have been the result of a negotiated consensus, and the specific reasons for numerical criteria may be poorly documented. Whether these or more explicit rule-based approaches are more effective in terms of protection goals is unclear. However, the diversity of riparian practices that have occurred, including those on federal lands, should prove to be an asset in advancing our understanding of what are the most effective approaches where resource protection is a goal.

PART II: TAXA SPECIFIC INFORMATION

Researchers from Oregon State University and the U.S. Geologic Survey who are knowledgeable with the taxonomic groups provide separate chapters here. These treatments are summaries of existing information, not extensive treatments, but sources of further information are provided. There is no treatment of mollusks. Mollusks have very narrow ranges and specific habitat requirements, and warrant attention, but there are relatively few experts on these organisms in the Pacific Northwest.

Each of the authors below describes life history attributes, habitat requirements, and ecological processes in relation to forestry practices. The authors also provide guidelines for management that can help sustain the particular taxonomic group, as well as a section on research needs. More general research needs are discussed in Section 14. There are, however, differences between these chapters based on specific attributes of the group of organisms.

6.0 FORESTRY PRACTICES AND AQUATIC BIODIVERSITY

6.1 Fish

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6.1.1 *Introduction*

In the Pacific Northwest, fish communities are found in a diverse array of aquatic habitats ranging from the large coastal rivers of the temperate rainforests, to the fragmented and sometimes ephemeral streams of the xeric interior basins, and high-elevation streams and lakes in the mountainous areas (Rieman et al. 2003). Only high-elevation lakes and streams isolated above barriers to fish passage remained historically devoid of fish because they were never invaded following Pleistocene glaciation (Smith 1981). Despite this widespread distribution and once great population abundances, taxonomic diversity of fishes in these forested systems is naturally lower than in aquatic habitats in the eastern U.S. (Reeves, Bisson, and Dambacher 1998).

Interactions among factors that influence species richness in aquatic systems (e.g., basin size, long-term stability of habitat, and barriers to colonization; Smith 1981) continue to influence the occurrence and persistence of fishes in these systems today. Consequently, the larger low-elevation rivers and estuaries support the greatest variety of fish species. In the high-elevation tributary streams, fish communities are less complex because these aquatic systems were less climatically and geologically stable, and fish populations were smaller and more prone to local extirpation. Furthermore, barriers to fish passage inhibited dispersal and colonization (Smith 1981). Streams in forested landscapes generally support salmon and trout, *Oncorhynchus* spp., whitefish *Prosopium* spp., sculpins *Cottus* spp., suckers *Catostomus* spp., and minnows (Cyprinidae), but in some of the colder streams, chars (e.g., *Salvelinus confluentus* and *Salvelinus malma*) and lampreys (Petromyzontidae) may also occur (Rieman et al. 2003).

Although biodiversity defined in terms of fish species richness is low in the Pacific Northwest, intraspecific variability is high, and polytypic fish species are common in the diverse aquatic habitats of the region. For example, the salmonids in the coastal rivers and streams, and the larger interconnected streams, rivers, and lakes of the interior exhibit a variety of ecotypes and migratory life histories (Healey 1986; Trotter 1989; Larson and McIntire 1993; Northcote 1997). This life-history variation appears to be associated with adaptation to spatial and temporal variation in environment (e.g., Schaffer and Elson 1975; Carl and Healey 1984; Beacham and Murray 1987), and there is some evidence of the genetic heritability of life-history traits (Carl and Healey 1984; Gharrett and Smoker 1993; Hankin, Nicholas, and Downey 1993). Persistence of any level of biological organization (e.g., life-history type, population, metapopulation, subspecies, species, community) is related to the interaction of environmental and biological components, and intraspecific diversity is a means of spreading risk (*sensu* den Boer 1968) of extirpation in dynamic environments (Gresswell 1999).

Unfortunately, despite the broad distribution and extensive intraspecific diversity, persistence of native fishes is uncertain in the Pacific Northwest. Many populations of anadromous salmonids, once synonymous with vigorous biological communities throughout the region, are threatened with extinction (Nehlsen, Williams, and Lichatowich 1991; Frissell 1993; Thurow, Lee, and Rieman 1997). Furthermore, over half of the native taxa in the Columbia River Basin are either listed under the Endangered Species Act, are being considered for listing, or are deemed sensitive by the management agencies (Lee et al. 1997; Thurow, Lee, and Rieman 1997). Potamodromous species like bull trout *Salvelinus confluentus* are estimated to occur as strong populations in less than 5% of their potential range (Rieman, Lee, and Thurow 1997). Although not currently listed under the endangered species list, the coastal cutthroat trout *Oncorhynchus clarki* is managed as a sensitive species in Oregon and California (Hall, Bisson, and Gresswell 1997). Native non-game fishes have rarely been monitored, but populations of species such as large-scale suckers (*Catostomus macrocheilus*), squawfish (*Ptychocheilus umpquae*), and Pacific lamprey (*Lampetra tridentata*) also are declining in some drainages (Oregon Department of Fish and Wildlife, unpublished data).

6.1.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

Compared to other land uses (e.g., agriculture or urban development), forestry is typically associated with greater aquatic diversity (Potter et al. 2004) and fish abundance (Pess et al. 2002). Nevertheless, historic timber harvesting and associated road construction practices that did not consider stream protection needs are linked to declines in diversity and abundance of salmonid species in the Pacific Northwest (Bisson et al. 1987, 1992; Nehlsen, Williams, and Lichatowich 1991; Sedell and Beschta 1991; Reeves, Everest, and Sedell 1993). Effects of timber harvest on aquatic systems may be both direct and immediate (i.e., pulsed disturbance; Yount and Niemi 1990) or indirect and sustained over an extended period (i.e., press disturbance; Yount and Niemi 1990). Timber harvest can affect stream ecosystems by altering hydrological patterns, stream temperature and solar insolation, habitat complexity, organic debris delivery and accumulation, sedimentation, and channel morphology (Hall and Lantz 1969; Brown and Krygier 1970; Everest et al. 1987; Gregory et al. 1987; Bilby and Ward 1991; Chamberlin, Harr, and Everest 1991; Johnson and Jones 2000). The magnitude and scale of effects are related to the size and intensity of the harvest, yarding techniques, geology, topography, watershed size, and amount, magnitude, and timing of post-harvest precipitation events (Murphy and Hall 1981; Swanson et al. 1989; Hicks et al. 1991). Furthermore, Best Management Practices, such as changes in logging systems, reforestation techniques, and riparian management areas, can moderate both generation and delivery of materials and energy to streams (Bisson et al. 1992; Ice 2004).

Reeves, Everest, and Sedell (1993) surveyed 14 watersheds (200-5,200 hectares) between 1985 and 1988 that had from 0-100% of the basin harvested. They used a two-sample t-test to compare density of juvenile anadromous salmonids between the two groups of watersheds (<25% and >25% harvested). There was no reference to when harvest occurred prior to 1985. They determined that diversity of juvenile salmonid assemblages was directly related to the proportion of the watershed that had been harvested, and diversity decreased when >25% of the basin had been harvested. Furthermore, basins that experienced a high level of harvest were more frequently dominated by a single salmonid species. Instream habitat heterogeneity was also directly related to level of harvest, and streams in low-harvest basins had significantly more pieces of wood per 100 m and more pools per 100 m than streams in high-harvest basins (Reeves, Everest, and Sedell 1993). Similar results have been documented for streams in Washington (Bisson and Sedell 1984) and coastal Oregon (Hicks et al. 1991).

Examining the extent of harvest provides important context for these findings. In the 23 years between 1972 and 1995, almost 20% of 4.6 million forested hectares in three provinces of western Oregon were subjected to clear-cut harvest, and the greatest concentration of cutting occurred on private industrial lands in the moist Coast Range Province (Cohen et al. 2002). When compared to public and non-industrial private landowners, private industrial landowners also had larger individual cutting units that were more spatially aggregated through time (Cohen et al. 2002). These data suggest that lands managed by private industrial landowners would have lower salmonid diversity and a higher probability of supporting only a single salmonid species.

The amount of time between natural disturbance events varies within and among basins (Poff and Ward 1990), and the temporal aspect of disturbance is an important part of the habitat template that likely influenced evolution of fishes inhabiting individual watersheds. Because natural recovery of stream systems may take decades to centuries (Gresswell 1999), substantially altering disturbance intervals by reducing times of harvest rotation may alter the system response and reduce persistence of fish in the system. Those systems that may exhibit the slowest recovery time following disturbance (e.g., eastside and higher elevation systems) would likely exhibit the largest negative response to repeated disturbances (Yount and Niemi 1990). Furthermore, it is important to recognize that response to a particular disturbance event is contingent on conditions remaining from previous events. Where natural recovery rates are unacceptably low, forest managers sometimes use active management practices in an attempt to restore key functions (Bisson et al. 1992; Ice 2000), but the results of these actions are often mixed (Kauffman et al. 1997).

Timber harvest potentially can affect water quality, water temperature, sedimentation, and channel structure (Chamberlin, Harr, and Everest 1991). For example, wood recruitment processes in small headwater streams are dominated by streamside landslides, toe slope-creep, and wind throw (May and Gresswell 2003a). A cycle of filling and spilling is common to many headwater streams in steep mountainous terrain. Over time, these channels fill with wood and sediment, which is episodically scoured by debris flows. Without a source of wood to increase the storage capacity of tributaries that have been scoured to bedrock following debris flow, these systems lose the capacity to store sediment and may persist in a bedrock state for an extended period (May and Gresswell 2003a).

The effects of vegetation removal on water temperature vary greatly among sites. Greater solar insolation following removal of riparian vegetation sometimes increases primary and secondary productivity in otherwise shaded streams (Murphy and Hall 1981; Murphy, Hawkins, and Anderson 1981; Hawkins, Murphy, and Anderson 1982; Hawkins et al. 1983), but lack of shade can also increase stream temperature and reduce salmonid habitat quality (Lantz 1971; Beschta et al. 1987; Johnson and Jones 2000). Where stream temperatures are not elevated excessively, however, instream productivity may increase (Gresswell 1999; Wilzbach 2005). There is also some evidence that even when water temperature increases are great at sites in headwater streams, changes may be negligible downstream (Gresswell 1999). Although water temperature of streams may increase after streamside vegetation is removed (Gray and Edington 1969), predicting the biological consequences is difficult (Beschta et al. 1987). Effects depend on the harvest intensity, size of harvest area, stream size, stream network complexity, watershed topography, normal temperature ranges of affected stream reaches, and life-history stage of the organisms present.

Effects of riparian vegetation diminish with increasing distance from the stream channel (Beschta et al. 1987; VanSickle and Gregory 1990; May and Gresswell 2003b). Estimates of the buffer width necessary to protect various riparian functions generally remain uncertain, but definitely vary according to individual function (e.g., root strength, large woody debris delivery to streams, large wood debris delivery to riparian areas, input of organic nutrients, shade, microclimate, water quality) (Castelle and Johnson 2000). In addition, site factors including geology, vegetation type, climate, topography, and watershed size influence riparian vegetation and its ecological role in an individual watershed.

Rieman et al. (2000) found a strong inverse relationship between stream biodiversity and road density throughout the Columbia River basin. Some negative effects of roads include persistent erosion of fine sediments, increased potential for slope failure, and passage barriers associated with stream crossing structures, especially culverts. In large-scale assessments, road density may indicate potential for declines in aquatic biodiversity, but these relationships are inextricably linked to historic practices and road locations. There is evidence that where problems can be identified with specific portions of a road, renovation can be very effective (NCASI 2003). For example, direct-delivery culverts, where road sediment is discharged directly to the stream network, have been found to be an important and persistent source of fine sediment (Bilby, Sullivan, and Duncan 1989), but these locations can be corrected. In some watersheds where abundance of native fish populations has declined and passage barriers have isolated small headwater populations, vulnerability of fish populations to reductions in genetic diversity (Wofford, Gresswell, and Banks 2005), disturbance (natural and anthropogenic), and potential extirpation has increased (Medina and Martin 1988; Propst, Stefferud, and Turner 1992; Rinne 1996). Recognizing and correcting these passage barriers is critical for reconnecting the stream network and reducing this risk (Kauffman et al. 1997).

Watershed or larger scale considerations. Although many ecosystem components, and relationships among these components, are poorly understood in the Pacific Northwest, there is growing recognition that the answers to many of these ecological questions are scale-dependent (May 1974; White and Pickett 1985; Frissell et al. 1986). For instance, the spatial and temporal patterns of large woody debris inputs to streams are important because they influence channel morphology, routing and storage of water and sediment, and provide structure and complexity associated with habitat for numerous aquatic organisms. Streams exist in a dynamic environment where relatively frequent site-scale and episodic broad-scale disturbances play a major role in creating and maintaining aquatic habitat. These natural cycles of disturbance create a diverse array of habitat types and availability through time and space (Reeves et al. 1995).

Investigations concerning the effects of spatial scale on scientific understanding of the organization of aquatic species are becoming more abundant (e.g., Imhof, Fitzgibbon, and Annable 1996; Richards et al. 1997), and it is increasingly apparent that a multiscale approach may facilitate interpretation of spatially extensive data (Poff and Allen 1995; Caselle and Warner 1996; Wiley, Kohler, and Seelbach 1997). For instance, numerous studies have investigated, or are currently evaluating, the relationships between physical habitat and anadromous salmonids (Nickelson et al. 1992; Reeves et al. 1995). It is difficult, however, to develop strong inferences because anadromous fish spend at least part of their lives in the marine environment where they are affected by a much different array of environmental variables, including commercial harvest. In contrast, nonmigratory freshwater fishes (e.g., sculpins) and fishes that migrate only in freshwater (potamodromous fishes, such as some populations of coastal cutthroat trout) are dependent on adequate freshwater habitat throughout their lives. Freshwater fishes, therefore, may be more tightly linked to changes in aquatic habitats than anadromous species, but much less effort has been expended to describe these linkages. The relationship between land management and aquatic habitat may be especially relevant for the coastal cutthroat trout because land management activities are among the factors that may have contributed to their decline (Williams and Nehlsen 1997).

Regional Variation. At the regional spatial scale, Thurow, Lee, and Rieman (1997) investigated the current and historical distribution of seven native salmonids in the Upper Columbia River Basin (east of the Cascade Mountains) and attempted to expand the results of site-level information to broad landscape scales. Other studies have identified some factors influencing distribution and abundance for some potamodromous species, such as bull trout (*Salvelinus confluentus*) (Rieman and McIntyre 1995; Rieman, Lee, and Thurow 1997; Watson and Hillman 1997), but the relationships between habitat and freshwater aquatic organisms across broad spatial scales in western Oregon are poorly understood.

6.1.3 *General Guidelines for Sustainable Forestry Practices*

Available information suggests that managers need to recognize that maintaining habitat and population diversity is critical to the persistence of most aquatic species, and management activities that lead to environmental simplification and homogeneity may ultimately have substantial negative effects at several levels of biological organization. Although habitats have been degraded and numerous fish species have declined, the region, and forested landscapes in particular, still supports important elements of the historical diversity in native fishes. Large interconnected networks of stream habitats continue to exist in some aquatic systems of the Pacific Northwest, and listed species have migration corridors through large rivers and lakes, to and from the ocean. Furthermore, high quality habitats can be found in large roadless areas, and some depressed fish species (e.g., bull trout) still occupy the majority of their historical range (Rieman, Lee, and Thurow 1997; Thurow, Lee, and Rieman 1997; Williams and Nehlsen 1997). Given a continuation of historical management trends, increasing human density, increased pressure from invasive species, and extractive demands, the trend for some of these systems is not positive (e.g., Rieman, Lee, and Thurow 1997). Conservation of remnant strongholds, often associated with the forested watersheds, and restoration of natural processes critical for habitat formation, will both be necessary for organism persistence (Lee et al. 1997; Young 1995).

6.1.4 *Information Needs*

Determining the relative importance of interrelated factors underlying past declines and current trends in populations of wild anadromous fish is particularly difficult. Changes in ocean productivity and amounts of fish harvested have been superimposed on varying influences on freshwater habitat over several decades and increasing hatchery populations. The genetic integrity and overall fitness of many naturally functioning taxa are difficult to predict without a better understanding of how freshwater habitat is functioning at landscape and regional scales. For example, much of the information available concerning factors that regulate woody debris input is limited to smaller spatial areas; little is known about woody debris recruitment, retention, and distribution at broader scales. Increasing our understanding of the physical and biological factors that influence these processes, and how these processes interact to influence fish populations, is crucial for the development of land management strategies that are compatible with persistence of native organisms.

Very little is known about fishes in the headwater portions of stream networks, and these small channels are often directly impacted by land use activities. Previous policies and management activities often failed to recognize the value of small channels and their associated riparian habitats (Beschta and Platts 1986). Indeed, small headwater stream channels can represent greater than 70% of the cumulative channel length in mountain watersheds (Benda et al. 1992). These headwater channels are important conduits for water, sediment, and wood routed from hillslopes to larger streams (Naiman et al. 1992; May and Gresswell 2003a).

Interactive effects of current logging practices on public lands administered by federal and state agencies and private timber lands that are managed primarily for commodity production are poorly understood. Determining the effects of land management activities on aquatic habitat, and ultimately aquatic organisms, is hampered by the complexity of interrelationships among physical, chemical, and biological characteristics of terrestrial, riparian, and aquatic systems, especially at broader spatial scales. The difficulty in developing a reliable long-term record of land use, land cover information, and additional stressors of species being studied also hampers progress. Frissell et al. (1986) developed a method for classifying stream systems in the context of the watersheds of which they are a part, and this approach is becoming more broadly accepted as a means of expanding understanding of the influence of disturbance and land management at the watershed scale (Imhof, Fitzgibbon, and Annable 1986). Despite a call for an increased emphasis on broader scale, integrated research, most studies are still conducted at the habitat-unit scale, and protocols for examining these data in a broader context have not been adequately developed (Imhof, Fitzgibbon, and Annable 1986; Poole, Frissell, and Ralph 1997; Smith, Gresswell, and Hayes 1997).

6.2 Stream Amphibians

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6.2.1 *Introduction*

Stream amphibians of the Pacific Northwest often are the dominant vertebrates in headwaters and small streams in terms of numbers of individuals and biomass (Bury et al. 1991). Larvae mostly eat aquatic invertebrates with a shift toward larger prey items with growth. Adults of giant salamanders eat large-sized invertebrates, other amphibians, and even small mammals (Bury 1972).

Three endemic families of stream amphibians occur in the Pacific Northwest, and all breed and deposit their eggs in flowing water (Nussbaum, Brodie, and Storm 1983; Bury 1994): torrent salamanders (4-5 species), Pacific giant salamanders (4 species), and tailed frogs (2 species). They apparently require rocky, flowing streams with closed forest canopies where temperatures remain cool year round, and they are most abundant in late seral stages of forests (Welsh 1990; Adams and Bury 2002; Welsh and Lind 2002). There is concern about the conservation status of the Southern torrent salamander because it was earlier petitioned for Federal listing, but was found not warranted at that time. The Southern torrent salamander is the most sensitive of the stream amphibians to timber harvest impacts (Corn and Bury 1989; Welsh and Lind 1991), closely followed by the tailed frog.

The torrent salamanders are small-sized, stocky salamanders that frequent seeps, headwaters and cascading, small streams. Juveniles and adults rarely venture farther than 1 m from water. Welsh and Lind (1996) reported that the Southern torrent salamander occurs in a narrow range of conditions: cold, clear headwaters to low-order streams with loose, coarse substrates (little sedimentation), in humid forest habitats with large conifers, abundant moss, and >80% canopy closure.

Tailed frogs occur from headwaters to third order streams (generally <2 m wide) but on occasion are found in larger waters, perhaps being swept downstream in seasonal flooding. Tailed frogs are adapted to fast, rocky streams: larvae have a streamlined body, muscular tail, and a large suctorial mouth to attach to rocks. The eggs of this primitive amphibian are slow to develop, averaging 6 weeks to hatching (Brown 1989). Most larvae metamorphose after 2 years in the Coast Ranges and 3-4 years in inland or northern sites (Daugherty and Sheldon 1982; Bull and Carter 1996; Brown 1990; Wahbe 1996). They transform in only 1 year in coastal parts of southern Oregon and northern California (Wallace and Diller 1998; Bury and Adams 1999).

Both torrent salamander and tailed frogs appear to be negatively impacted by elevated stream temperatures and siltation resulting from clearcut timber harvest (Bury and Corn 1988a; Corn and Bury 1989; Dupuis and Steventon 1999; Biek, Mills, and Bury 2002; Welsh and Lind 2002). They are absent in open or dry slopes such as oak woodland. Pacific giant salamanders range from headwaters to large streams, sometimes in waters with partial canopy. Most egg deposition sites are in subterranean habitats (e.g., underground seeps). The larval stage is about 17-18 months long. Larvae and adults may grow to 1 ft long (300 mm). Adults sometimes occur in upland forests during rainy periods. They co-occur with native salmonid fishes in some larger waters. Although their numbers may be depleted by logging in some areas, the giant salamander appears to persist in watersheds that receive timber harvest.

In the Pacific Northwest, there are also several plethodontid (lungless) salamanders in or near streams. The Van Dyke's Salamander (*Plethodon vandykei*), frequents seeps and rocky talus in forested stands. They seem to prefer streams within older forests, but there are no trend data available on populations. Similarly, Dunn's Salamander (*Plethodon dunni*) occurs in wet rock rubble or talus, most often along creeks and streams.

Larger streams have slow pools and side waters that are used by several other species of amphibians, mostly for breeding. Stream order 3 and larger waters may have yellow-legged frogs (*Rana boyli*), a stream specialist. These frogs live along the edges of the waters, often jumping from shore or land into the stream when disturbed.

6.2.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

The number of species, densities and biomass of stream amphibians are significantly greater (2–10X) in streams flowing in natural than in logged forests (Corn and Bury 1989). Tailed frogs are reduced by timber harvest where logging opens up large tracts in watersheds or leaves no buffer areas (Bury and Corn 1988b; Corn and Bury 1989; Welsh 1990; Bury et al. 1991; Bull and Carter 1996; Dupuis and Steventon 1999).

Immediate changes occur in stream habitats due to loss of streambank vegetation or from upslope activities (e.g., road development, timber harvest). Logging practices often result in sedimentation, which degrades amphibian habitat by reducing access to cover sites (Corn and Bury 1989; Dupuis and Steventon 1999). Steeper gradients tend to flush out sediments, which may mitigate silt inputs from logging (Hall, Murphy, and Aho 1978). Large downed wood in streams also provides energy input to the stream ecosystem and adds stability to the stream flow. Natural recruitment of such material is lost with removal of large trees. Replacing and maintaining woody debris in streams where natural recruitment is lost or reduced is a simple step that will help maintain aquatic biodiversity regardless of stand objectives.

The nature of the bedrock or parent geology is important in determining the severity of sedimentation. In managed stands in western Washington, Wilkins and Peterson (2000) found greater abundance of all stream species as gradient increased. Further, they reported fewer giant salamanders and absence of tailed frogs in drainages underlain by sediment of marine origin compared to those underlain by basalt (more fragmented rock present). In contrast, stream amphibians were found in many drainages with marine sediments inside Olympic National Park, a large forested reserve (Adams and Bury 2002). This suggested that timber harvest has a more severe effect on amphibians in areas underlain by marine sediments.

Forest harvest, especially clearcutting, increases insolation and raises stream temperatures, thereby increasing microbial respiration, primary production, invertebrate consumers, and populations of predators on amphibians. Tailed frogs are the least tolerant of any frog to elevated temperature, and die quickly when exposed to water temperatures near 29.6°C (deVlaming and Bury 1970). Exposure to temperatures at 24°C for several hours may be lethal (Metter 1966; Claussen 1973). Further, their eggs perish at >18.5°C (Brown 1975). This may be their "weak link" because tailed frogs nest in summer when water temperatures are at their peak. Similarly, torrent salamanders require cool temperatures (Bury and Nebeker, pers. obs.; Pilliod et al. 2003): larvae die at 26.7°C (range 25.6-27.4°C) and adults at 27.9°C (range 26.3-29.3°C). These are the lowest values known for any stream amphibian. Thus, timber harvest or other factors that elevate stream temperatures to approach or exceed these values are of concern to the maintenance of "healthy" conditions for headwater amphibians in the Pacific Northwest.

There are no studies that directly relate size of forest harvest areas to stream conditions for stream amphibians. Corn and Bury (1989) found that tailed frogs and torrent salamanders sometimes occurred in logged areas if there was uncut mature forest upstream. In contrast, these species were lacking in some uncut forest if some logging had occurred above the area. These spatial relationships need further study.

Stream amphibians occur in managed stands (Diller and Wallace 1996; Olson et al. 2000a, 2000b, 2000c; Wilkins and Peterson 2000), but often at reduced numbers except in maritime influenced areas, where summer fogs ameliorate summer temperatures. We know little about how frequency of harvest regimes (rotation age) affects stream amphibians. However, we recently found that stream amphibians in the Oregon Coast Range had not recovered 35-50 years after clearcut harvesting (Bury and Pearl 1999; B. Bury and D. Major, in prep.). Harvest of stands every 60-70 years may be too frequent for populations of sensitive species (torrent salamander and tailed frog) to recover.

In northern California, some recent forest practice rules have improved riparian protection over the past. Streams now have equipment exclusion zones and tree retention from 15-30 m on each side of streams (Diller and Wallace 1996). However, Welsh and Lind (1996) reported that there is still a serious problem with the misclassification of streams where the faulty assumption is made that aquatic life does not exist in particular channels; this results in inadequate protection. Further, the protection width is very narrow (15 m) and may not maintain required microclimatic conditions for the torrent salamander. And, the canopy can be reduced to 50% of pre-harvest levels after each entry.

Effective width of buffer areas is unclear. Maximum shading capacity may be within a width of 25 m, and 90% of that capacity occurs at 17 m (see Budd et al. 1987). However, widths of 30 m or more are needed to stabilize microclimates within streamside riparian zones (Brosofske et al. 1997). To reduce sediment flow and maintain other riparian functions, the minimum buffer width may need to be 60-80 m wide (Ledwith 1996; Welsh et al. 1998) or up to 100 m (McComb, McGarigal, and Anthony 1993). Recently, Vesely and McComb (2002) reported that minimum buffer strips on most private forests (6.1 m along medium-sized streams and no buffers along headwaters in Oregon) may not be sufficient to ensure that amphibian communities in managed stand remain as diverse as in unlogged forests. They recommended extension of riparian buffer strips to permanent headwater streams and buffer strips 20 m wide or more on all streams.

Watershed or larger scale considerations. Although adults of the tailed frog are considered to be closely associated with streams (Blaustein et al. 1995), they occur in forests 100-500+ m from water during the wet season (Corn and Bury 1991; Wahbe, Bunnell, and Bury 1999). Also, Bury and Corn (1987, 1988b) captured many recently metamorphosed *Ascaphus* in pitfall traps set in forested stands near streams. These data suggested recently metamorphosed animals disperse into surrounding forests by autumn. Thus, retention of shade from riparian zones and adjacent forests may be critical to the survival and dispersal of even those stream amphibians with high fidelity to the stream channel.

Tailed frog populations show strong genetic differences among watersheds (Ritland et al. 2000), suggesting low movement potential. Similarly, genetics of torrent salamander in most watersheds of are highly distinctive (Good and Wake 1992). In fact, the four species currently recognized are as genetically distinct as most families of birds and mammals, reflecting the ancient lineage and isolation of torrent salamanders.

Regional Variation. In northern California, Diller and Wallace (1996) found that torrent salamanders appeared to occur in managed stands more frequently in maritime areas than at interior locales. These finding are consistent with a hypothesis that the local effects of harvest depend upon the magnitude of temperature changes with canopy removal, which increase from the coast inland. Similar findings may also conceivably apply along latitudinal gradients.

6.2.3 *General Guidelines for Sustainable Forestry Practices*

Diller and Wallace (1996) report that recent forestry practices are improvements over the past, including tree retention standards for 15-30 m on each side of streams, better road construction, and logging practices (e.g., cable logging). A two-tiered approach (deMaynadier and Hunter 1995; Vesely and McComb 2002) suggests protection of a narrow (e.g., 10-25 m, each side) no-cutting zone adjacent to the stream corridor, surrounded by a wide zone where limited harvesting may take place (e.g., removal of 25% of the basal area).

We may profit from suggestions of Budd et al. (1987) who stated "Width of the riparian zone varies from stream to stream and along the course of an individual stream. In the Pacific Northwest, stream buffer widths for each side range from 11 m to 38 m, depending on the riparian ecosystem element studied." Further, deMaynadier and Hunter (1995) recommended adjusting buffer width proportionally to a) stream width, b) the intensity of adjacent harvest, and c) slope. We may need to invoke "adaptive management" (learn as we go) or "adaptive protection" where we employ a variable buffer width that reflects a suite of local (e.g., type of harvest, stream gradient) and regional factors (climate, soils).

6.2.4 *Information Needs*

Our collective goals may be better served by reducing reliance on a set of physical criteria such as buffer widths at 30 m and paying more attention to the desired outcomes to ensure for streams a) cool waters (e.g., temperature <15 C year round); b) little or no sedimentation; and c) input of large woody debris over the long haul. These are essential elements not only for amphibians but also for a host of other biota (stream invertebrates to salmonid stocks) that depend on the same suite of stream conditions. Our barometers or indicators can be accurate tracking of stream temperatures (via automated recorders that are now relatively inexpensive) and rapid habitat assessment (as siltation is obvious to the trained eye).

We need to better determine the occurrence and abundance of aquatic biota across major climatic gradients. There are generally cooler conditions from south to north, inland to coast, and from low to high elevation. Impacts from timber harvest may be ameliorated or less severe in northern, maritime, or high elevation as opposed to southern, interior or low elevation locales. We may find that buffer widths can be less wide in streams subjected to maritime conditions than sites located farther inland. We really do not know right now.

Still, stream temperatures in harvested stands may increase to harmful levels for the critical period of egg survival for stream amphibians. We know the maximum tolerance level for only one of the three stream amphibians groups: eggs of the tailed frog perish at 18.5 C. There may be sublethal effects at lower temperatures. Again, we lack data.

Intermittent streams or headwaters are often overlooked in forest management plans, but these are critical to the survival of stream amphibians and for maintenance of stream conditions downstream that include the economically important salmonid fishes. Preliminary studies in the Umpqua River basin in southern Oregon (Bury, pers. obs.) indicate that tailed frogs and torrent salamanders are restricted to headwaters that are stream order 1-2 (basically <2m). Tailed frogs and giant salamanders may occur down to third order waters and, rarely, larger waters. We lack information on the distribution of herpetofauna along streams in other areas along river continua. We have no studies on how logging affects large-stream biota that include the yellow-legged frogs and Western pond turtles.

Substrate and parent geology appear to be important to presence of stream amphibians. Although we have tended to debate the role of marine deposits, the bigger question is the amount of sedimentation that is occurring in any watershed. Although in need of better documentation, the key here is maintenance of streams with interstitial spaces that may be essential to survival of many stream amphibians.

Nesting areas of stream amphibians need protection, but this is difficult because few egg masses have ever been found due to a) their placement under large rocks, boulders or underground seeps; b) communal nesting that concentrates eggs in a few sites; and c) lack of extensive searches. This chapter in their life history is mostly blank.

We need data on the movement patterns, dispersal abilities, and general landscape-level needs of tailed frogs for effective conservation. For example, newly metamorphosed tailed frogs travel upslope but we lack information on how far they travel into forests and how long they spend on land.

In the past, clearcut logging occurred over large areas that tended to fragment blocks of suitable habitat. Continuing timber harvest at these remaining sites may further deplete the remaining populations. Although there are many improved methods of timber harvest (e.g., thinning of stands and retention of riparian reserves), studies are just starting on how these practices may affect amphibians.

We need to determine the effectiveness and cost efficiency of different widths of buffer zones along streams to protect aquatic organisms. Thinning of stands is increasingly used as a management tool to harvest timber and to reduce fuel loads, but there are few or only small-sized treatments in the region. Wide-scale implementation of thinning of forests needs further consideration and review.

6.3 Macroinvertebrates

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6.3.1 *Introduction*

Aquatic invertebrates play many key roles in the stream ecosystems of the Pacific Northwest. They are the major consumers of detrital input from the riparian area and are often abundant when food sources are available (Hawkins and Sedell 1981; Gregory et al. 1991; Naiman and Anderson 1997). In streams in relatively undisturbed forests, aquatic invertebrate diversity is extensive and abundance remains generally stable (McElravy, Lambert, and Resh 1989; Richards and Minshall 1992). In a modest sampling effort over a hundred different species may be detected (Hershey and Lamberti 1998). In addition, they are prey species for many aquatic vertebrate species including salmonids and amphibians (Moring and Lantz 1975; Wipfli 1997), as well as terrestrial vertebrate species. According to the Oregon-Washington Species-Habitat Project database (O'Neil et al. 2001), 196 species of amphibians, reptiles, birds, and mammals use aquatic invertebrates as a food resource.

Major taxa of aquatic invertebrates of the Pacific Northwest include: *Ephemeroptera* (mayflies), *Odonata* (damselflies, dragonflies), *Plecoptera* (stoneflies), *Megaloptera* (dobsonflies, alderflies), *Tricoptera* (caddisflies), *Coleoptera* (beetles), *Collembola* (springtails), *Diptera* (true flies), *Gastropoda* (snails), *Decapoda* (crayfish), and *Amphipoda* (amphipods) (Moring and Lantz 1975, Porter and Meehan 1987, Hershey and Lamberti 1998). Few species appear on any “sensitive species” short list because most are both relatively widespread and frequent.

Life histories are highly varied according to seasonal fluctuations, life cycle length, and developmental strategies (McIntire and Colby 1978; Anderson and Sedell 1979; Hawkins and Sedell 1981; Porter and Meehan 1987; Progar and Moldenke 2002). Temperature and photoperiod associated with seasonal change play a role in determining when a particular species may emerge, transition from a larval to adult stage, or export downstream (Porter and Meehan 1987; Piccolo and Wipfli 2002). In the Pacific Northwest, many species overwinter in a larval stage and synchronize their emergence in order to increase the probability of finding mates (Hershey and Lamberti 1998). During much of the summer, more than 100 insects may emerge from a single square meter of stream per week (Progar and Moldenke 2002). In four geographical areas of study in Oregon, *Diptera* was the most abundant order collected in summer and fall and *Ephemeroptera* were most abundant in the winter (Porter and Meehan 1987).

Composition and abundance of aquatic invertebrates are largely dependent on sources of available organic matter (Murphy and Hall 1981; Bilby and Bisson 1992; Naiman and Anderson 1997), canopy structure (Gregory et al. 1991; Richards and Minshall 1992; Progar and Moldenke 2002), water quality (i.e., dissolved oxygen) (Gibbons and Salo 1973; Hershey and Lamberti 1998), local geomorphology (Naiman et al. 1992), and basin position (Vannote et al. 1980; Naiman et al. 1992). The River Continuum Concept suggested that the types of invertebrate feeding functional groups located in the stream are related to the location in the stream network, because of changes in the riparian canopy and the relative influences of riparian-derived (allochthonous) vs. internally produced (autochtonous) carbon. Many studies in the Pacific Northwest are consistent with this view (Vannote et al. 1980; Hawkins and Sedell 1981; Hawkins, Murphy, and Anderson 1982; Anderson 1992). Hawkins and Sedell (1981) found that low-order streams with at least partial canopy closure often have aquatic invertebrate composition consisting of 30-50% shredders. In western Oregon, third through fifth order streams create better habitat for collectors and grazers as a shift from heterotrophy to autotrophy occurs due to increases in light and algal production, increases in broken down coarse organic matter, and as stands become dominated more by *Alnus rubra* and *Populus trichocarpa* (Swanson et al. 1982). In general, terrestrial and aquatic interactions decrease with increasing stream size (Swanson et al. 1982).

Substrate and velocity help determine which species will be present in a particular stream reach. Coarse substrate provides better interstitial habitat than fine sediments. For example, water penny beetles, hellgrammite larvae, perlid stoneflies, and case-building caddisflies use the undersides of rocks (Hershey and Lamberti 1998). Large wood and leaf packs are “hot spots” of invertebrate activity because they can be both substrate and a nutritional resource (Anderson and Sedell 1979; Hershey and Lamberti 1998). Only specialized aquatic invertebrates ingest large wood directly, such as cranefly larvae (*Lipsothrix*) and the elmid beetle (*Lara avara*) (Wallace and Anderson 1996). A far larger proportion of aquatic invertebrates feed on the microorganisms that coat the wood (Anderson and Sedell 1979; Bilby and Bisson 1998).

Disturbance effects are dependent on the nature and magnitude of the event, geomorphic and hydrologic stream characteristics (Resh et al. 1988), and community structure (Minshall et al. 1985). The intermediate disturbance hypothesis predicts that the greatest aquatic invertebrate diversity will occur at intermediate levels of disturbance. At low disturbance levels, large, longer-lived invertebrates dominate communities; while at high disturbance, those species that have poor colonization mechanisms or are long-lived are excluded, leaving small, short-lived invertebrates (McAuliffe 1984; Hershey and Lamberti 1998). Flow variation (from floods to desiccation) can be responsible for large, often temporary, decreases in abundance and diversity (Lamberti et al. 1991). Large wood reduces shear stress during high discharge events, helping to stabilize the substrate

(Naiman and Anderson 1997). The hyporheic zone can provide refuge and source for colonizers, but is usually only available to smaller invertebrates (Stanford and Ward 1992; Naiman and Anderson 1997; Edwards 1998).

In the Pacific Northwest, mechanisms for aquatic invertebrates to recolonize stream reaches after a natural disturbance include downstream drift, drift from tributaries, upstream flight, flight from other watersheds, upstream swimming, upstream crawling, and movement from hyporheic refugia (Hershey and Lamberti 1998, adapted from Smock 1996).

6.3.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

Effects from clearcut logging on the stream ecosystem include the introduction of sediment, removal of leaf litter and other organic inputs, and the change in flow, temperature, and nutrient composition (Gibbons and Salo 1973; Murphy and Hall 1981; Swanson et al. 1987; Gregory et al. 1991), altering energy flow from terrestrial to aquatic habitats (Murphy and Hall 1981; Wipfli 1997). Open canopied streams showed higher abundances of aquatic invertebrates and increases in the abundance of all the feeding functional groups besides shredders, which remained relatively stable, compared to streams with closed forests over them (Hawkins, Murphy, and Anderson 1982). In the Oregon Cascades, the clearcut section had four times the insect emergence, although biomass was relatively the same (Grafius 1976).

Canopy vegetation composition is more influential than substrate in determining total abundance and trophic levels present in the stream (Hawkins, Murphy, and Anderson 1982). The count and biomass density of aquatic invertebrates were found to be 3 times greater in 35- to 40-year-old red alder sites than in 35 to 40 year old conifer sites (Piccolo and Wipfli 2002). Aquatic invertebrate abundance has been found to be higher in *Alnus rubra* stands than stands dominated by *Thuja plicata* or *Tsuga heterophylla* (Kolodziejczk and Richardson 2001). Deciduous tree leaves that are high in nutrients (e.g., alder and maple) are broken down within four to six months, while leaves from other deciduous tree species (e.g., oak), conifer needles, and shrubs and herbs with waxy cuticles (e.g., willow and sword fern) may take 1 to 2 years (Gregory et al. 1991).

There are no studies that directly relate size of forest harvest areas to stream conditions for aquatic invertebrates. Primarily, studies have been conducted on stream reaches that are relatively short (100-300 m). Robinson and Minshall (1986) have stressed the need to compare a variety of patch sizes to get a more accurate understanding of aquatic invertebrate response.

In early succession, species richness is highest, which drops to a lower number of species once the canopy is closed, then eventually returns to intermediate levels of aquatic invertebrates as the canopy ages (Franklin 1992; Triska, Sedell, and Gregory 1982). Murphy and Hall (1981) found that predatory insects in the Oregon Cascades showed that species richness was 28% greater and biomass was 88% greater in clearcut than in old-growth sites, yet communities were predominantly stoneflies. Their densely shaded second-growth sites contained fewer predatory aquatic insects than old-growth or clearcut sites. However, Anderson (1992) found that when comparing insect taxa at three sites in the Oregon Cascades, there was no significant difference: old-growth had 196, the 5-year-old clearcut had 191 and the 40-year old deciduous (primarily alder) had 165. It is important to note that the 40-year-old deciduous site experienced a debris flow within 10 years of the study, but many of the aquatic insects had recolonized.

Aquatic invertebrate abundance recovers relatively quickly from openings in the canopy (Murphy and Hall 1981; Anderson 1992), but diversity does not recover as quickly (Erman, Newbold, and Roby 1977; Newbold, Erman, and Roby 1980). Newbold, Erman, and Roby (1980) found that although overall density was significantly higher in streams logged without buffers than in unlogged streams, diversity was significantly higher in unlogged streams. Results from streams with narrow buffer strips were too varied to show a significant difference than either unbuffered or unlogged, but with a buffer of 30 m, they determined there was no significant difference in aquatic invertebrate populations compared with unlogged reaches.

Other important aspects in considering logging frequency are the rate of recovery of large wood that can enter the stream (Murphy and Hall 1981), and the frequency of shifting in substrate materials that can alter food abundance and aquatic invertebrate communities (Robinson and Minshall 1986).

Tree species composition and canopy structure alter the composition and abundance of aquatic invertebrates (Gregory et al. 1991; Koloziejczyk and Richardson 2001). Low, dense canopies allow little sunlight for primary production, whereas high, more open canopies allow some light through (Murphy and Hall 1981; Gregory et al. 1991). In a study in the Oregon Cascades where clearcutting and burning occurred next to a stream without a buffer, Murphy and Hall (1981) found that density of aquatic invertebrate taxa increased more in riffles than in pools because of increased primary production and less sedimentation.

Logging practices have altered the size and amount of large wood in the stream ecosystems of the Pacific Northwest (Sullivan et al. 1987; Bilby and Ward 1991; Benda, Bigelow, and Worsley 2002). Riparian buffers have protected some of the large wood inputs (Gregory and Ashkenas 1990), and it has been suggested that riparian buffers may increase the amount of wood entering the stream in the short term due to windthrow (Hairston-Strang and Adams 1998). Removal of large wood serves to simplify habitat or increase sediment loading which can reduce aquatic invertebrate density (Niemi et al. 1990).

Watershed or larger scale considerations. One of the reasons it is difficult to quantify how disturbance affects aquatic invertebrates is because disturbance operates at varied spatial scales that range from an individual rock to an entire watershed and broad temporal scales from a single event to long successional time periods (Gregory et al. 1991; Naiman and Anderson 1997).

Canopy type significantly alters quantity and biomass of aquatic invertebrates exported downstream (Piccolo and Wipfli 2002); therefore, timber harvest in the headwaters is expected to alter riparian inputs and the energy flow through the food web (Triska, Sedell, and Gregory 1982; Wipfli and Gregovich 2002). In southeast Alaska, it has been found that every 1 km of salmon-bearing stream has the potential to receive enough food resources, by means of aquatic invertebrates and detritus, from non-fishbearing headwater streams to support 100-2000 young-of-the-year salmonids (Wipfli and Gregovich 2002).

Road building associated with logging has the potential to decrease aquatic invertebrate abundance and diversity by altering peak flow in streams (Jones and Grant 1996; Jones 2002) and increasing the frequency and severity of landslides and debris flows in the Pacific Northwest (Swanson et al. 1987). Lamberti et al. (1991) found that 99% of macroinvertebrates were removed by a debris flow in the Oregon Cascades, while the flood alone removed 90% of the species. In the same region, Anderson (1992) found that most aquatic insects are able to recolonize debris flow opened habitat within a few months, and herbivore shredders and piercers were the most affected by the debris flow.

Regional variation. There is a large diversity of habitats in the Pacific Northwest; therefore, seasonal emergence varies depending on region. A study conducted by Porter and Meehan (1987) in four regions of Oregon (Coastal, Cascades, Central, and Eastern) determined that in most areas and seasons, *Diptera* and *Ephemeroptera* combined comprised over half of all stream invertebrates collected. High percentages of mayflies were found in Coastal and Cascades Oregon in the spring, while high amounts of both mayflies and stoneflies were found in Eastern Oregon and springtails were more abundant in Central Oregon.

Timing and magnitude of flood flow can vary from coastal to inland regions, scouring bed materials and depleting aquatic invertebrate populations (Naiman and Anderson 1997; Hershey and Lamberti 1998). In coastal streams and the Cascades, rain-on-snow events can produce high flow events that reset the stream biota (Lamberti et al. 1991). Debris flow frequencies vary among regions (Swanson et al. 1987), and are often triggered by large precipitation events, exacerbating impacts of a flood on aquatic invertebrates.

6.3.3 *General Guidelines for Sustainable Forestry Practices*

We don't yet know much about the impacts of specific buffer widths and current forest management practices on aquatic invertebrates, but studies are underway. The most influential elements of the riparian area on aquatic invertebrates include canopy cover, litter and organic input, and the input of large wood. Riparian management practices should take into account these needs and the influence of the headwater streams. In addition, because of the large amount of information on aquatic invertebrate responses to environmental conditions, many responses can be used as indicators of stream conditions (Naiman 1998; Hershey and Lamberti 1998).

6.3.4 *Information Needs*

The relationship between the terrestrial ecosystem and the stream ecosystem is well understood qualitatively, but not quantitatively (Hershey and Lamberti 1998). The role of disturbance to the riparian canopy structure needs to be further examined. Aquatic invertebrate biodiversity appears to increase as patches are opened by disturbances (Section 2.3) that increase habitat heterogeneity (Anderson 1992). More studies need to be done on how the frequency and intensity of logging and associated road building influence dispersal, rate, and speed of recovery of aquatic invertebrate populations (Jones et al. 2000). The functionality of riparian buffer widths at maintaining diversity and population densities of both aquatic and terrestrial-riparian arthropods needs to be assessed in both the mesic forests of the north and the more xeric forests of the south (see research of Rykken and Moldenke, Oregon State University, unpublished).

Many studies have been conducted in the coastal region and the Cascades, but more studies need to be initiated in the interior like those conducted in central Oregon by Progar and Moldenke (2002). These have documented that a) temporary headwater streams produce higher densities of invertebrates (i.e., migratory bird food) than continuous flow primary streams; b) continuous flow primary streams have higher levels of diversity/species richness than temporary ones; and c) the unique species in the temporary streams are not true aquatic species, but are related to terrestrial-riparian species.

The linkages between aquatic invertebrates and vertebrate species should be of interest to managers and have not been well studied. In the short term, there is a need to carefully consider the often opposing needs of both salmon and terrestrial birds/bats for the food resource provided by aquatic insects. For example, very few hard data have been collected about the diel flight periods of the major species of aquatic insects (bat food versus bird food) (Alex Farrand, Department of Fish and Wildlife, Oregon State University, unpublished data).

7.0 FORESTRY PRACTICES AND TERRESTRIAL BIODIVERSITY

7.1 Vascular Plants

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7.1.1 *Introduction*

Vascular plant distributions are governed by the availability of critical resources for growth, suitable temperature conditions for photosynthesis and growth, and availability of growing space. Most vascular plants require comparable growth resources, including mineral nutrients, water, and with the exception of a few saprophytic species, light. High levels of specialization for unique or even transient combinations of these resources are typical of most flora and this is especially true for riparian forests. Spatial gradients in resources (light, soil moisture), stress (flooding) and temporal gradients caused by disturbance are well exploited by riparian plant species, yielding high diversity. Species and life form groups vary sharply in their tolerances to the various physiological stresses imposed by the riparian environment. Ephemeral species are well represented in well-illuminated, spatially complex, and frequently flooded streamside environments (Pabst and Spies 1998). These conditions are more stressful for longer lived tree and shrub species, which may show lowest richness next to streams (Sarr 2005). Still other species show strong substrate preferences, and primarily occur on either mineral or organic substrata (Minore and Weatherly 1994; Pabst and Spies 1998). In short, the diversity of plant species is driven by the spatial and temporal complexity of the riparian environment.

An array of local and regional processes creates this complexity and provides opportunities for plant establishment and development. Local processes include erosion and deposition associated with fluvial disturbances, colluvial processes such as raveling and slope failure that bring organic and inorganic materials from uplands to the riparian area, the dispersal of propagules into and out of the riparian zone by various means, periodic fire, and biotic interactions among plants as well as among plants and animals. At larger scales, history, climate, geology, and watershed processes govern the spatial distributions of plant species in the Pacific Northwest (Whittaker 1960; Waring and Major 1964; Waring 1969; Zobel, McKee, and Hawk 1976; Ohmann and Spies 1998; Wimberly and Spies 2001).

Riparian forests harbor a high diversity of understory plant species, some of which are unique to the streamside environment and many others that also occur in the uplands (Mouw and Alaback 2003). The assembly of riparian communities is determined in part by environmental gradients that vary and interact at multiple spatial scales (Baker 1989; Bendix 1994). Climatic gradients and geologic substrata may control what plant species are found at the landscape scale (Sarr 2005). For example, salmonberry (*Rubus spectabilis*) is widespread at lower elevations in western Washington and in Oregon's Coast Range, where it occurs along streams as well as in the uplands, but in drier westside locations (i.e., eastern and southern Coast Range, Cascades, and Siskiyou Mountains) it is absent or its extent is limited to streamsides (Sarr 2005). Woody plant diversity has been shown to vary strongly along landscape climate gradients coincident with changes in upland floras (Collins, Risser, and Rice 1981; Sarr 2005). West of the Cascades, woody plant richness typically is lowest in the wettest most productive environments, where canopy densities are greatest and topographic moisture gradients are most muted (Sarr 2005). Studies of upland forests suggest that herbaceous plant diversity shows a similar pattern (Whittaker 1960), but a regional study has yet to be implemented that addresses total plant diversity in riparian environments.

Within watersheds, the composition of riparian plant communities changes as stream gradient, valley floor width, and the complexity of valley bottom landforms change from the upper to lower reaches of a stream network (Hupp 1986). This has been demonstrated in both the Oregon Cascades (Campbell and Franklin 1979) and the Oregon Coast Range (Pabst and Spies 1999). In addition, total species richness, as well as the proportion of exotic and pioneer species, has been shown to increase from the upper, constrained reaches of the McKenzie River in Oregon to the piedmont zone where the river gradient lessens and the valley floor widens (Planty-Tabacchi et al. 1996; Tabacchi et al. 1996).

At the scale of stream reaches, variability in plant communities is associated with elevation above the stream and other factors related to it, including susceptibility to flooding, soil moisture, soil texture, site productivity, and microclimate. Microclimate has not been directly associated with plant community composition in the Northwest, but has been shown to change relative to buffer widths in both western Washington (Brosofske et al. 1997) and northwestern California (Ledwith 1996a, 1996b). Along a stream-to-hillslope gradient in the central Coast Range of Oregon, species richness and diversity of understory vegetation were higher on valley floors than on lower hillslopes; within the valley floor, more frequently disturbed areas such as active floodplains and vegetated gravel bars were more species-rich than less disturbed areas such as terraces and seeps (Pabst and Spies 1998). In contrast, Sarr (2005) noted lower diversity of woody plants on floodplains than adjacent hillslope forests, despite greater geomorphic complexity on the floodplains. These contrasts may be due partially to differences in physiological tolerances of herbs and woody plants, as well as to the fact that understory diversity is typically highest where environmental conditions limit dominance by trees and tall shrubs.

Microhabitats such as back channels, seeps, depressions, and boulders or logs in or near the stream provide unique environments for plants and introduce finer-scale heterogeneity in the riparian understory (Campbell and Franklin 1979; Pabst and Spies 1998). Understory composition in the Coast Range changed along the stream-to-hillslope gradient in relation to landform type, topographic position, and coniferous tree cover (Minore and Weatherly 1994; Pabst and Spies 1998). Similarly, distinct plant communities were identified along an elevation/disturbance gradient along streams in the Oregon Cascades (Hawk and Zobel 1974; Campbell and Franklin 1979). The distribution of these communities was further differentiated by substrate texture and corresponding moisture stress (Hawk and Zobel 1974). In the Hoh River Valley on the Olympic Peninsula, vegetation succession was associated with landforms of different ages and elevations that were defined by historical flooding and deposition (Fonda 1974). Riparian plant associations in the national forests of central Oregon are also tightly linked to landform (Kovalchik and Chitwood 1990).

Diversity/equitability patterns and community patchiness in riparian areas are strongly influenced by major disturbances such as floods and debris flows. Scour and deposition from these events create a variety of new substrata and landforms for the establishment of opportunistic herbaceous and woody pioneer species, including non-natives (Gecy and Wilson 1990, Pabst and Spies 2001). Propagules (seeds, spores, vegetative parts) for these plants may have been dispersed by wind, animals, carried by the stream itself (Johansson, Nilsson, and Nilsson 1996), or germinated from the seed bank in the soil. Other species survive catastrophic disturbance by resprouting from buried rootstocks or rhizomes (Gecy and Wilson 1990; Pabst and Spies 2001) or by being flexible or resistant to the force of peak flows. Composition of the riparian seed bank was evaluated on three types of alluvial surfaces along streams in the central Cascades of Oregon (Harmon and Franklin 1995). Seed banks were dominated by herbaceous species, with nearly 77% of the species not represented in the existing aboveground vegetation. This indicates the potential for an expansion or shift in species composition in riparian areas following disturbance. Species abundance and richness of the seed bank were higher on vegetated gravel bars that experienced occasional fluvial disturbance than on surfaces disturbed more or less frequently (Harmon and Franklin 1995). This trend associating richness or diversity with disturbance intensity or frequency has been documented elsewhere as well. For example, in southeast Alaska, species richness was highest in areas with intermediate frequency of flooding and a high degree of spatial variability in flood frequency (Pollock, Naiman, and Hanley 1998). In contrast, richness was found to increase with increasing flood severity in southern California (Bendix 1997). Community patchiness along streams is further enhanced by local disturbance events such as windthrow of overstory trees and small slope failures, which increase the availability of light and possibly soil moisture for plants.

Biotic interactions also influence the composition of riparian plant communities. Competition among plants for light, moisture and nutrients is an important force in the organization of plants and communities, while herbivores such as beaver (Wright, Jones, and Flecker 2002), elk, deer (Liang and Seagle 2002), and mountain beaver (Neal and Borrecco 1981) can dramatically affect the makeup and stature of the riparian understory. Beaver-impounded sites in the Coast Range host a unique assemblage of graminoid species at the water's edge, making these sites compositionally distinct from unimpounded sites and debris jams, leading to greater biodiversity in the landscape (Perkins 2000). Competitive effects appear to be the reason for lower species diversity in red alder/salmonberry community types in the Coast Range, compared with other community types associated with alder (Carlton 1989). Furthermore, species richness within the alder/salmonberry type did not change across different successional stages (i.e., over time) (Henderson 1978). It has been hypothesized that dense cover of shrubs, particularly salmonberry, limits the regeneration of conifer tree species along streams in the Coast Range of Oregon (Carlton 1989; Hibbs and Giordano 1996; Hibbs and Bower 2001). However, in a controlled experiment in western Washington, Beach and Halpern (2001) found that seed source availability (as a function of dispersal distance) was the primary factor governing conifer regeneration, and that competitive interactions and substrate type were of secondary importance. This has implications for riparian understories since the presence or absence of conifers and the year-round shade they cast affect plant community composition (Pabst and Spies 1998).

7.1.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

Only a few studies directly examined the effect of forest practices on vascular plant diversity in the understory of riparian forests in the Pacific Northwest. Therefore, in addition to summarizing these studies, we also reviewed findings from studies outside the region or in upland areas. In many cases, the studies did not explicitly address questions of disturbance intensity, size, frequency, or proximity to stream, so these findings represent only a partial description of the effects of harvest on vascular plant diversity.

The Washington Department of Natural Resources undertook a study to determine how different methods of riparian forest management affected wildlife and vegetation in low elevation, second growth, conifer-dominated riparian forests in western and northeastern Washington (O'Connell et al. 2000). Sampling was done two years before and two years after clearcut harvest in the adjacent uplands to compare current (mid 1990s) riparian forest protection rules ('state' treatment) with 'modified' (more protective) rules, and an unharvested control. The state and modified rules included a no-entry zone as well as some selective cutting in the riparian buffer. Multiple sites were selected for each treatment along Type 3 (fish bearing) or Type 4 (perennial, non-fish bearing) streams. An apparent drawback in the study design was that treatments were not replicated at the site level; that is, each site received a different treatment, possibly confounding site effects with treatment effects.

Results from the western Washington sites show that prior to harvest, cover of mosses, ferns, deciduous shrubs, and berry-producing shrubs was significantly greater in riparian areas than in uplands. Following harvest, the cover of ferns and mosses in the riparian zone was significantly greater in the control sites than in the treated sites. However, cover of berry-producing shrubs was significantly greater in riparian zones bordered by clearcuts, possibly in response to increased light from the uphill edge. At the northeastern Washington sites, the pretreatment comparison between riparian and upslope vegetation showed that the riparian areas had significantly lower richness of shrub species, significantly higher richness of herbaceous species, and that most species occurred in both upland and riparian areas. Following harvest, there was no difference in the richness of shrub species among treatments. Richness of herbaceous species did not differ after treatment in the control or under the modified rules, but was significantly less following harvest under state rules.

In the Oregon Coast Range, Hibbs and Giordano (1996) compared vegetation in red alder-dominated buffer strips to that in alder-dominated riparian forests undisturbed by logging. The buffer strips, ranging from 5 to 50 m wide, represented a chronosequence of time since harvest that ranged from 0 to 32 years. Nearly twice as many herbaceous species were detected in the buffers as in the undisturbed forest, with 22 herbaceous species being unique to the buffers. However, measures of herbaceous species richness, evenness, and diversity did not show a statistically significant difference between buffered and undisturbed sites. Cover of the shrub layer was higher on average in the buffers, with salmonberry showing the most pronounced difference.

Also working in the Oregon Coast Range, Hibbs and Bower (2001) examined understory vegetation in unharvested buffer strips under four overstory canopy types across a chronosequence of time since harvest (from 1 to 33 years). The number of shrub species (richness) was highest in the buffer strips with a conifer cover type, whereas the number of herbaceous species was highest in the mixed, conifer-dominated type. The pure hardwood type had the fewest herb and shrub species. Percent cover of the herbaceous layer did not change with buffer age or with distance from the clearcut edge. Cover of shrubs also showed no association with distance from edge but was significantly (weakly) correlated with buffer age. As part of the same study, Hibbs and Bower (2001) compared their buffer data to that from unmanaged, conifer-dominated riparian forests (Pabst and Spies 1999). They found that total cover of the herb and shrub layers was not significantly different between buffered and unmanaged riparian forests, leading to the conclusion that buffers differed little in composition from undisturbed forest. More detailed comparisons, such as the cover of individual species, total species richness, and the proportion of non-native species, were not made between the two data sets.

Research from outside the region or from upland areas may also shed light on understory response to harvesting. Deal (1997) studied riparian forests in coastal Alaska that developed after 45 years of logging. He found that richness of vascular plants in the understory of alder-dominated or mixed alder-conifer stands was about twice that found in a conifer-dominated stand. In upland forests of the Pacific Northwest, species richness increased with thinning intensity in Washington (Thomas et al. 1999) and Oregon, although in the Oregon study, a portion of the increase was due to the presence of non-native species (Bailey et al. 1998). Conversely, He and Barclay (2000) found no significant effect of thinning and fertilization on vascular or nonvascular species richness in the understory of a young Douglas fir stand 27 years after treatment. In the coastal redwood forests of northern California, species richness increased in association with a sunlight gradient from within 30-50 year-old regenerated clearcuts (low light) into old-growth forests (higher light) (Russell and Jones 2001).

Tree harvesting adjacent to or within riparian buffers could influence the composition and structure of vegetation in the buffers, given the potential for disturbance to soils, increased rates of windthrow of overstory trees (Steinblums, Froehlich, and Lyons 1984), and changes in the light regime and microclimate (Brosofske et al. 1997; Ledwith 1996a, 1996b). Existing evidence from the Pacific Northwest does not give clear support for this hypothesis (O’Connell et al. 2000; Hibbs and Giordano

1996; Hibbs and Bower 2001). Hibbs and Bower (2001) speculated that edges created by upslope harvesting in the Coast Range are mitigated by rapid growth of the vegetation, although in less productive climates of the Pacific Northwest (e.g., east side of both Cascades and Klamath Mountains; Appendix A) it is likely that functional regrowth of vegetation would take longer. It is plausible that plant species that are adapted to the dynamic riparian environment are not particularly sensitive to changes in resource availability brought about by upslope logging. For instance, Bendix (1998) postulated that valley bottom plant communities may be in a relatively stable 'quasi-equilibrium' with disturbance from the flood regime. However, herbaceous plants in riparian areas respond to finer-scale environmental variability than the shrub and tree layers (Decocq 2002).

Roads and logging also have been associated with higher rates of erosion and sedimentation (Hagans, Weaver, and Madej 1986) and possibly increased incidence of debris flow (Robison et al. 1999) and the volume of sediment they entrain (Johnson, Swanston, and McGee 2000), all of which have the potential to alter streamside plant communities. Yet we know of no study that has attempted to link these dynamics to vascular plant diversity.

Regional Variation. We know of no studies that have examined a consistent set of harvest techniques in riparian forests across the region, but other field studies may provide important insights into potential effects of forest management. Sarr (2005) studied woody plant diversity in forest interiors and disturbance gaps in riparian forests of four watersheds ranging from the wet western Coast Range through the western Cascades to the dry eastern Siskiyou Mountains in southwestern Oregon. He found that gaps had significantly higher woody plant richness than forest interiors in the Cascades, but differences were not significant in the wettest and driest climates. In the wettest climate, salmonberry dominance limited diversity in gaps, whereas in the driest climate the relatively open forests were of comparable richness to gaps. Parallel comparisons of species composition between gaps and forest interiors at the sites suggested that differences in plant composition were associated with increased light availability with disturbance in wet climate riparian forests, whereas in the driest climate, disturbance caused a shift toward more drought-tolerant taxa. The implications of this study for forest management may be that disturbances of similar sizes may lead to different vegetation responses in sharply contrasting climates, as the relative roles of light limitation, microclimatic stress, and shrub competition change.

7.1.3 *General Guidelines for Sustainable Forestry Practices*

Because vascular plant species show strong differences in microhabitat specialization, land managers can protect biodiversity by maintaining heterogeneity in vertical and horizontal stand structure. They should also take care to avoid damage to unique intra-riparian habitats or substrata that contribute to habitat heterogeneity and presence of legacies. Examples include snags, downed logs, boulders, seeps, depressions, back channels, beaver ponds, and mountain beaver dens. Furthermore, the "quality" of riparian vegetation—in terms of the presence and proportion of native versus non-native species—must be considered when evaluating potential impacts of logging and roading. Roads can be conduits for the spread of non-native plant species (Trombulak and Frissell 2000) as well as pathogens (e.g., *Phytophthora*) (Jules et al. 2002). Some highly invasive, non-native plant species are particularly troublesome in riparian areas, with the ability to form monocultures at the expense of native plants. Examples include Himalayan blackberry (*Rubus discolor*), reed canarygrass (*Phalaris arundinacea*), false brome (*Brachypodium sylvaticum*), butterfly bush (*Buddleia davidii*), creeping buttercup (*Ranunculus repens*), and Japanese knotweed (*Polygonum cuspidatum*) (Native Plant Society of Oregon, http://www.emeraldnpso.org/PDFs/Invas_Orn.pdf). Although it is certain that natural forests historically experienced disturbance of a wide spectrum of intensities, moderate or low disturbance may currently be warranted in watersheds known or suspected to have high potential for non-native plant invasion.

In summary, the effects of timber harvest on vascular plants appear to be variable and dependent upon life form group and site characteristics such as climate and existing vegetation type. Because many vascular plant species show broad associations on disturbance gradients (Spies and Turner 1999), differences in species richness between different silvicultural treatments and stand ages are often nonsignificant. In addition, since many studies are retrospective stand comparisons, it is unknown if differences noted are due to site or treatment. Some tentative conclusions seem appropriate: a) vascular plant species diversity as a whole is not strongly affected by harvest, but silviculture will affect vegetation composition; b) non-native species increase in proportion with disturbance and vegetation quality may decline; c) more open disturbed, old-growth, or deciduous forests may have somewhat higher diversity than young even-aged conifer stands.

7.1.4 *Information Needs*

Even these broad summary statements require considerably stronger substantiation. A more consistent application of specific treatments replicated across landscape gradients of the Pacific Northwest may be needed to clarify the effects of riparian management on plant diversity in different locales. Other potential research topics include the following:

1) Conduct a before/after control treatment study of harvesting effects on understory vegetation in Oregon and northern California. Ideally, this type of study would be stratified across a range of stream sizes and stand types, with treatments replicated at each site, and sampling conducted over a long time frame. Attributes to measure/monitor would include cover (preferably biomass) and constancy of all plant species, disturbance, light regime and soil moisture, etc. This would considerably strengthen the findings of existing retrospective studies. Treatments might include variable thinning approaches, small patch cutting, and shelterwood harvest with variable green tree retention levels.

2) Develop successional models to simulate vegetation development in riparian (and upland) areas following disturbance. A greater understanding of the life history of many riparian plants is needed to develop a predictive basis for determining forest management effects on riparian plant diversity.

3) Understand the factors driving the distribution of key native and non-native plant species in riparian areas. We lack basic information on the distributions of most native riparian species on geographic, hydrologic, or disturbance gradients. A gradient perspective of riparian vegetation may allow researchers to extrapolate from geographic areas where research has been most intensive (e.g., Oregon Coast Range) to areas where studies are sparser. In addition, we need an empirical and predictive basis for understanding non-native species distributions and to estimate their responses to different management techniques.

4) Conduct additional chronosequence studies of riparian buffers, similar to that of Hibbs and Bower (2001), in areas outside the Oregon Coast Range. Successional trajectories vary sharply depending upon such geographic factors as climate, species present, so more geographic coverage is needed to evaluate the generalities of such local findings.

5) Examine how riparian restoration efforts affect plant communities. In many areas of the Pacific Northwest, geomorphic alterations due to loss of large woody debris or beaver, non-native species invasions, or other changes have led to degraded vegetation condition. It is unknown if such changes can be reversed through vegetation planting, large wood emplacement, or non-native plant eradication. More work is needed to determine the feasibility or impact of such restoration efforts.

7.2 Non-Vascular Plants

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7.2.1 *Introduction*

The humid temperate rainforests of the coastal mountains are typified by the luxuriant growth of lichens and mosses, both on the forest floor and as epiphytes on live trees and snags. Bryophytes (mosses, liverworts, and hornworts) and lichens (dual organisms, individuals of which are composed of an alga and a fungus) represent two widely disparate groups with similar habitat preferences in forests of the Pacific Northwest. For simplicity, when referring to these two groups collectively, I will refer to them as nonvascular plants for the remainder of this section. As photosynthetic organisms, nonvascular plants not only comprise an important element of riparian biodiversity, but they contribute functionally to carbon fixation, and in the case of cyanolichens, to nitrogen fixation in riparian ecosystems. I discuss here basic lichen and bryophyte biology as it relates to the vascular plant community (trees and shrubs), and the riparian ecosystem as a whole.

Non-vascular plants play a major role in forest ecosystems, capturing and converting light (primary productivity). Lichen biomass may exceed 1 metric ton per hectare and bryophytes several times this (McCune 1993, and references in Longton 1992) reaching their greatest abundance in oceanic regions (Schofield 1984). They function to filter air and water, and they are major contributors of nitrogen and carbon to the terrestrial and aquatic food webs. They serve as food for a variety of vertebrates (including elk and flying squirrels) and invertebrates, as well as nesting material for birds and small mammals. They act as sources and sinks for nitrogen and in contact with water they act as filters and sponges to absorb nutrients in terrestrial and aquatic settings.

The narrow habitat requirements of some non-vascular plants, where they are known, make them valuable indicators of the forest type or environmental conditions (e.g., soil pH, Klinka et al. 1995). Some are associated with particular substrate types, be it rock, bark, wood, soil, etc., which helps to identify them and in turn has led to their use as environmental indicators. In Europe, bryophytes are frequently used to classify forest types, although this is rarely done in North America (Klein and Vanderpoorten 1997, Klinka et al.1995).

Basic Biology, Distribution and Diversity. The importance of riparian areas to non-vascular plants arises because of specific habitat features important to non-vascular plants, particularly humidity, light, and substrate. Non-vascular plants typically do best with abundant moisture and light but moderate temperatures, hence their abundance in the moist regions of the coastal temperate rainforests of the Pacific Northwest. They may be restricted to locations where faster growing vascular plants are limited and therefore frequently associated with particular habitats including bare rock, the darkened forest understory, or as epiphytes growing high in the canopy on live or dead tree branches and boles.

In this way they are well adapted to take advantage of the favorable growing conditions in the Pacific Northwest. Although, as noted, they are fundamentally different taxonomically, the lifestyles of bryophytes and lichens are often strikingly similar. Perhaps the most remarkable feature they share is their ability to withstand frequent desiccation, with the ability to resume metabolic activity after rewetting. Generally, nutrients and water may be absorbed over the entire surface of the organism. While advantageous in some ways, this feature makes them sensitive to water borne and air borne pollutants (McCune 2000). The cyanolichens are unique in their ability to fix atmospheric nitrogen, a feature that has likely helped them gain attention among forest managers. Many bryophytes may be tolerant of temperature extremes when dormant, but more heat sensitive when hydrated and undergoing active growth. Respiration rates typically exceed photosynthetic rates at over 25ºC, making bryophytes among the most sensitive organisms to increased heat loads in riparian forests (Longton 1980).

Reproduction differs markedly between bryophytes and lichens. Each, however, is capable of sexual and asexual reproduction. The resulting propagules are typically small and dispersed by wind, water, or attached to animals. The reproductive output is often high but survival is more often dependent on the probability of arriving at a site with conditions amenable to growth and establishment (substrate and microclimate).

The accumulation of biomass for non-vascular plants is generally low and highly variable, in comparison to vascular plants. Given adequate time free of disturbance, non-vascular plants may build up considerable biomass, as observed in temperate rainforests throughout the Pacific Northwest. Cool, moist conditions conducive to their growth may occur out of phase with that of vascular plants (e.g., in fall and winter). Even the driest parts of the region, including the interior valleys and mixed forests of northern California, provide abundant habitat for epiphytes and good growing conditions in winter when precipitation tends to be highest. However, riparian zones may be the locations most likely to support bryophyte species that remain active and require appropriate microclimatic conditions through summer (J. Shevock, pers. comm).

Schofield (1988) reports that 85% of the bryophyte species occurring in British Columbia have a Holarctic distribution, whereas the remaining 15% are limited (endemic) to western North America. The Pacific Northwest is recognized as a center of endemism for both lichens (Brodo, Sharnoff, and Sharnoff 2001) and bryophytes (Tan and Pocs 2000) so it is of particular importance to the conservation of non-vascular plant species diversity. Approximately 1000 species of bryophytes occur in the Pacific Northwest: 700 mosses, 300 hepatics (liverworts), and 5 hornworts (Schofield 1984).

Old-growth forests provide conditions for lichens and bryophytes that are unmatched by other forest types due to the continuity of substrates and variety of microsites and microclimates that exist there, from the forest floor upward into the highest part of the canopy (McCune et al. 2000).

7.2.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

Schofield (1988) suggests that the influences of humans on bryophyte distribution have been less than for vascular plants. The number of introduced species, for example, is limited in quantity and geographic extent. However, indirect impacts on non-vascular plants by humans likely have been extensive. This is primarily through extensive conversion of plant communities that altered habitat availability.

The effects of forest management on non-vascular plant diversity tend to be indirect, by removing habitat structure and substrate heterogeneity and by changing microclimates. The reduction in stand ages and structural diversity has likely had effects on non-vascular plant abundance and diversity, but these effects are currently undescribed. Forest practices may change the microclimates and substrata available to non-vascular plants, including tree and snag density, abundance of large woody debris on the ground, soil surfaces, vascular plant size and composition. Trees and shrubs provide structure or influence microclimate, or both, for non-vascular plants and, therefore, are primary determinants of habitat suitability. Consequently, it is not surprising that forest practices should impact population viability of non-vascular plants, especially those with narrow habitat requirements. Sillett et al. (2000) point out that many species of lichen, while considered old-growth dependents, do not have narrow habitat requirements, but do seem to be limited by dispersal. Some rare and sensitive lichen species (e.g., *Lobaria oregana*) may require a long time to establish and grow in forests (Sillett et al 2000). Such species appear to be favored by retention of old-growth patches in the landscapes as well as large green trees within managed stands (Sillett and Goslin 1999; Sillett et al. 2000).

Dead standing trees (snags) and large woody debris (LWD) on the ground are important habitat substrates for many non-vascular plants (Jonsson 1996b; Rambo and Muir 1998; Rambo 2001). LWD increases the surface area and heterogeneity of the forest substrata. As the composition of LWD changes over long periods of time, a single piece may, in various stages of decay, serve as habitat for a variety of species and in particular may be more important for non-vascular plants (and fungi) than to seed plants. Live trees, particularly large trees, provide for a wide variety of microclimates as temperature, wind speed, light levels, and humidity all change in various ways from the deep understory to the top of tall trees 60-80 meters into the upper canopy. Bryophytes and lichens are susceptible to changes in humidity brought on by removal of the forest canopy. Some may take advantage of the increase in light availability, although this increase in resources may be short lived as a dense tree canopy develops.

Few studies have attempted to describe non-vascular communities specific to riparian areas (but see Rocky Mountains - Glime and Vitt 1987; Oregon Cascades - Jonsson 1996a, 1996b; McCune, Hutchinson, and Berryman 2002). Bryophytes in particular may take advantage of unique substrata in the near stream environment (the splash zone). Here they may colonize stable boulders or embedded rocks where seed plants may be limited, due to flooding or lack of soil (Englund 1991; Glime and Vitt 1987). Changes in flow regime, sedimentation or debris flows may directly impact the bryophyte community with indirect effects on the aquatic invertebrate community (Englund 1991).

Riparian areas provide a variety of unique characteristics to the larger landscape due to their topographic position. The composition of vascular plants is often highly variable along riparian corridors, where they are subject to frequent disturbance. This may lead to a greater diversity of stand ages and forest types in close proximity, compared to an equivalent area of upslope forest (Naiman et al. 1998). A number of lichen species appear to benefit from the occurrence of hardwoods and associated understory shrubs occupying hardwood stands (Neitlich and McCune 1997; Ruchty, Rosso, and McCune 2001). In a single watershed in Oregon, larger “fish bearing” streams contained a greater number of rare epiphytic lichens (those listed on the survey and manage list—Bureau of Land Management and USDA Forest Service) compared to upland sites or small and intermittent streams (McCune, Hutchinson, and Berryman 2002).

The abundance of hardwood tree species in riparian areas may be an important habitat factor for non-vascular plants. Conifer canopies tend to be dense and carry their needles year-round, making for a darkened understory and mid-canopy. In contrast, deciduous canopies are leafless during winter, when conditions are optimal for bryophyte growth, and understory light levels are higher year-round. The texture and chemical properties of tree bark tend to vary among species and age of individuals, each suited to different kinds of non-vascular epiphytes (Pike et al. 1975).

7.2.3 *General Guidelines for Sustainable Forestry Practices*

The influences of climate, topography, distance from stream, vascular plant composition and age, and abundance of large woody debris on non-vascular plant diversity operate at different scales (Jonsson 1996b) and must be taken into account when managing for particular species or for non-vascular plants as a whole in riparian areas. What is most apparent is the relationship between non-vascular plant abundance and diversity and stand structure (Lesica et al. 1991; Sillett et al. 2000). For the most part, habitat loss has been due to the conversion of older, structurally and compositionally heterogeneous stands to younger, more dense and homogenous forests. Consideration should be given to the amount and distribution of legacy trees and stands to ensure the existence of suitable habitat over the landscape in the long term, and to retention of large green trees within managed stands to serve as sources of propagules to inoculate younger stands (Sillett et al. 2000). In addition to large trees, retention of other legacy features or sources of within-stand heterogeneity including snags, logs, hardwood and shrub patches, and variable density stands, are all factors that may be favorable for maintenance of non-vascular plant diversity in the riparian zone.

7.2.4 *Information Needs*

Despite their abundant growth and diversity, non-vascular plants have rarely received much consideration in forest management in the Pacific Northwest. This has changed to some degree since the implementation of the Northwest Forest Plan. A legacy of neglect is also apparent in the scarcity of personnel capable of identifying non-vascular plants efficiently and accurately. For epiphytic species (i.e., those growing on trees and snags), difficulties also arise in simply accessing the locations where they thrive (e.g., canopies of mature and old-growth forests). The same may also hold true for species growing on cliffs and outcrops over streams and along ridges.

Research on the diversity and conservation of non-vascular plants has developed along various lines of questioning but much remains unanswered. Research has only begun to consider the specific habitat requirements of most species and the trade-offs among life history traits of individual species. Development of an understanding of life history characteristics and habitat requirements, as has been done for some vascular plants, should be part of an adaptive management strategy for maintaining non-vascular species diversity.

The direct harvesting of moss for commercial use has received some attention from research as well (Peck and Muir 2001). Given that riparian areas may be locations of both high diversity (and hence conservation value) and high productivity, there may be some conflict between economic and biological values that warrants further monitoring.

It should be recognized that recent advances in silviculture, especially stand management techniques to increase structural characteristics, are being discussed and implemented to varying degrees (Franklin et al. 2002; Tappeiner, Emmingham, and Hibbs 2002). These methods may favor the persistence of non-vascular plants in managed landscapes, but the short- and long-term implications of such silvicultural techniques have received little attention to date.

7.3 Fungi

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7.3.1 *Introduction*

Life History and Ecology. Although the degree of association of most native fungi with riparian areas is poorly understood, maritime forests of the Pacific Northwest have a rich diversity of macrofungi, and they are abundant in riparian forests (Molina pers. com.). Native macrofungi include mycorrhizal, saprophytic, and pathogenic species. Most native conifer and hardwood tree genera form ectomycorrhizal associations with fungi in the Basidiomycotas, Ascomycota, or Zygomycota (Molina et al. 2001). Douglas fir alone may associate with as many as 2000 fungal species across the region. Fungi have tremendous functional importance in forest ecosystems, occurring in the full spectrum of climates, stand types, and successional stages. Many species are essential to decomposition of carbon materials, such as wood and leaf detritus, both in terrestrial and aquatic habitats. Hypogeous sporocarps (truffles) form a major food source for small forest mammals, including the northern flying squirrel (*Glaucomys sabrinus*) (Zabel and Waters 1997). In addition, mutualisms between certain fungi and plant roots (mycorrhizae) have important roles in water and nutrient acquisition for most native riparian trees (Amaranthus and Perry 1987). Pathogenic fungi may also play an important role in maintaining heterogeneity in natural riparian forests by creating small islands of tree mortality and shrub or hardwood establishment.

Aquatic fungi form an essential element of the detrital food web in streams. After leaching of soluble carbohydrates, microbial colonization is the primary form of breakdown of more refractory leaf or stem parts. The layers of aquatic molds that frequently occur on decaying leaves increase the protein content of these substrates, encouraging ingestion by detritivorous insects. Colonization by fungi is rapid and spore densities may exceed 1000 spores /liter (Dix and Webster 1995). Fungi may comprise 63-95% of the microbial biomass on submerged hardwood leaves (Findlay and Arsuffi 1989).

Fungi have evolved to exploit a tremendous wealth of habitats in the Pacific Northwest. Most species of fungi require sufficient seasonal moisture to reproduce. Therefore, fungal reproduction shows high seasonality and interannual variation, depending upon temporal patterns in temperature, precipitation, and relative humidity. In the Pacific Northwest, fungal sporocarp abundance and diversity are believed to be positively correlated with climatic moisture (O'Dell, Ammirati, and Schreiner 1999), reaching peaks in the humid forests of the coastal mountains. Fungi are also positively associated with the presence of large wood and the diversity of decay classes (Molina et al. 2001).

7.3.2 *Ecological Processes and Habitat Features related to Forestry Practices*

Over 500 species of fungi were identified by the FEMAT process as being closely associated with old growth forest in the Pacific Northwest (Marcot 1997). It is unknown how many are susceptible to forest management of various intensities. The intensity of harvest may be important for fungal diversity primarily through its effect on residual woody debris and soil organic matter. Across gradients of harvest intensity, compositional shifts from forest interior taxa to more disturbance-associated taxa have been noted (Jones, Durall, and Cairney 2003). In a study that transplanted conifer seedlings from forest interiors into forest openings, fewer ectomycorrhizal morphotypes, lower average richness per seedling, and steeper, less even species distribution curves were found on seedlings transplanted into openings (Kranabetter and Friesen 2002). Standing crop of hypogeous (below ground) sporocarps has been noted to be lower in managed stands than in natural-mature and old-growth stands (North, Trappe, and Franklin 1997), and to decrease with the size of the forest opening (Durall et al. 1999). However, richness and abundance did not appear to differ strongly between mature and old-growth stands on Vancouver Island, B.C. (Goodman and Trofymow 1998).

The effects of harvest on fungal persistence may show important regional variation in the Pacific Northwest. In dry, high elevation clearcuts of southwest Oregon, where temperature and moisture extremes pose severe stress, declines in mycorrhizal fungi have been noted (Amaranthus and Perry 1987). Similarly, studies in the northern Rocky Mountains have shown differences in fungal diversity between clearcut and undisturbed sites (Byrd et al. 2000). Hagerman et al. (1999), studying high elevation clearcuts in interior British Columbia, noted no initial differences between clearcuts and undisturbed sites, but recorded declines in fungal diversity two and three years after harvest. In contrast, cool coastal sites with greater organic matter and more equitable climate may be better able to maintain fungal populations through harvest cycles (Molina pers. com.). In addition, ectomycorrhizal fungal species show strong associations with specific tree or shrub species (Molina et al. 2001). It is currently unknown how fungal species diversity, biomass, and function are affected by conversions from mixed species assemblages to conifer plantations, but it is likely that species depending upon noncrop trees may be detrimentally affected. Hagerman, Sakakibara, and Durall (2000) demonstrated that retention of native understory shrub species may be important in maintaining diversity of ectomycorrhizal fungi morphotypes through harvest cycles. It should be mentioned that the few published studies of harvest effects on fungal communities have occurred in upland forests. Although many species and mechanisms overlap between upland and riparian forests, the unique conditions in riparian zones probably warrant further study.

Fungal communities are rich in species, and there are important successional relationships among taxa (Dix and Webster 1995; Molina et al. 2001). It is likely that most activities along the spectrum of timber harvest approaches will be beneficial to some taxa, but detrimental to others. However, as heterotrophic organisms, fungi require a carbon source and are typically most abundant and diverse where a variety of decaying wood substrates are readily available. Intact organic soils also provide habitat for many mycorrhizal species.

7.3.3 *General Guidelines for Sustainable Forestry Practices*

Forest management should aim to maintain diversity in stand species, structures, and ages and abundant, heterogeneous sources of large wood debris through space and time. Ground disturbing activities may be detrimental to fungal communities that require well-developed soils for establishment and growth. Flushes of successional vascular plant species (e.g., *Alnus, Ceanothus, Arctostaphylos* spp.) associated with disturbance may be important mycorrhizal host plants and essential for maintenance of diverse fungal populations at the stand and landscape scales (Hagerman, Sakakibara, and Durall 2000). Intensive shrub control practices may need to be relaxed following disturbances, to ensure that these seral species remain. Fungi require adequate moisture for growth and reproduction, so microclimatic changes associated with disturbance may pose considerable physiological stress in large disturbed areas, especially in more seasonal and droughty interior climates.

7.3.4 *Information Needs*

As organisms that exist primarily under the ground or in decaying wood, fungi are out of sight during most of the year. Basic information on the life history of fungi is needed for both upland and riparian ecosystems in the Pacific Northwest. Field inventories are needed, especially in remote areas, far from research universities. However, seasonal low-intensity inventories may be largely inaccurate and of limited utility for determining presence or absence of fungal species (Molina et al. 2001). The considerable time and taxonomic effort required to properly ascertain the status of fungal and lichen biodiversity argues for an ecosystem study approach, perhaps in concert with study of other small or cryptic organisms, such as bryophytes and forest invertebrates.

Great effort and expense have been expended to survey rare fungi on federal lands of the Pacific Northwest (Molina et al. 2001). A parallel effort of similar magnitude on private lands is not likely and may prove redundant. From the base of information being gathered under the Northwest Forest Plan, and targeted inventories on private lands, it may be possible to develop habitat models to determine distributions on private lands.

However, private lands may provide the best opportunity to evaluate the effects of multiple, relatively short rotation harvest, herbicide use, and perhaps other factors that are uncommon or absent on public land. In general, manipulative studies designed to determine relationships between harvest or disturbance intensity and fungal diversity and biomass are needed to determine harvest impacts and conservation strategies for managed landscapes. Such a research program might be replicated in a consistent way across landscape gradients to determine the interactions between landscape setting and harvest effects. Preliminary evidence indicates that heavy thinning (many trees removed) can substantially reduce mushroom productivity immediately after the harvest, but that lighter thinning (fewer trees removed) has a lesser impact on productivity (Pilz and Molina, unpublished data). The rate at which mushroom productivity rebounds as the remaining trees reoccupy a site has yet to be determined. Although it is likely that mixed stands will harbor greater diversity than stands of single species, it is unknown how quickly fungal species diversity might respond to disturbance or interplanting with native hardwood in otherwise conifer dominated stands (or vice versa. The recent finding that different tree species can translocate carbon below ground via fungal mycelium (Simard et al. 1997) may represent a functional relationship whereby species richness and ectomycorrhizae interact to maintain ecosystem integrity.

7.4 Mammals

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7.4.1 *Introduction*

Riparian forests in the Pacific Northwest are particularly important to mammals. Some 65% of the mammalian species occurring in Oregon and Washington use riparian areas and 27 species are threatened, endangered, or of special interest (Kauffman et al. 2001). Nine species are considered to be riparian obligates (Table 7.1). Many other species occur in both upland and riparian habitats, but are frequently more abundant (Doyle 1990; McComb, McGarigal, and Anthony 1993; West 2000a, 2000b) or have greater fitness (Doyle 1990) in riparian habitats (Table 7.2). In addition, because riparian forests are subject to relatively frequent disturbance, early succession-associated species of mammals are often more abundant in riparian forests than in upland habitats (Anthony et al. 2003). The degree of association with riparian areas often varies depending on geographic context, surrounding forest condition, and season (McComb, Chambers, and Newton 1993; Gomez and Anthony 1998; Kelsey and West 1998; Anthony et al. 2003). For example, whereas woodrats (*Neotoma* sp.) are not typically associated with riparian habitats throughout most of their range, they are associated with riparian areas in the more xeric forests of northwestern California and southern Oregon. In addition, generalist carnivores such as black bears (*Ursus americanus*), ringtails (*Bassariscus astutus*), and ermine (*Mustela erminea*) become more abundant in riparian areas in response to prey availability during spawning runs of salmon (Anthony et al. 2003).

Table 7.1 Riparian Obligate Species of Mammals in Coniferous Forests of Western Oregon and Washington (from Anthony et al. 2003)

Common Name	Scientific Name
northern water shrew	*Sorex palustris*
marsh shrew	*Sorex bendirii*
water vole	*Microtus richardsoni*
beaver	*Castor Canadensis*
muskrat	*Ondatra zibethicus*
nutria	*Myocastor coypus*
river otter	*Lutra Canadensis*
mink	*Mustela vison*
Columbia white-tailed deer	*Odocoileus virginianus leucurus*

The degree of association between riparian habitats and mammalian species depends, in part, on the size of the stream. Large-bodied highly aquatic mammals, such as river otters (*Lutra canadensis*), mink (*Mustela vison*), beaver (*Castor canadensis*), and riparian-associated mammals such as elk (*Cervus elaphus)* and deer (*Odocoileus* sp.), are most frequently associated with mid- to large-sized streams and rivers whereas most riparian-associated small mammals such as water voles (*Microtus richardsoni)* are associated with small sized streams (Kelsey and West 1998). In addition, riparian areas at low- to mid-elevations appear to be used more as travel corridors by some species than are riparian areas at high elevations (Kelsey and West 1998).

Table 7.2 Riparian-Associated Species of Mammals in Coniferous Forests of the Pacific Northwest (adapted from Anthony et al. 2003)

Common Name	Scientific Name	Notes
masked shrew	*Sorex cinereus*	early seral [a]
montane shrew	*Sorex monticolus*	
Pacific shrew	*Sorex pacificus*	
fog shrew	*Sorex sonomae*	
shrew mole	*Neurotrichus gibbsii*	
California myotis	*Myotis californicus*	
long-eared myotis	*Myotis evotis*	
little brown myotis	*Myotis lucifugus*	
fringed myotis	*Myotis thysanodes*	
long-legged myotis	*Myotis volans*	
Yuma myotis	*Myotis yumanensis*	
hoary bat	*Lasiurus cinereus*	
silver-haired bat	*Lasionycteris noctivagans*	
big brown bat	*Eptesicus fuscus*	
Townsend's big-eared bat	*Corynorhinus townsendii*	
Allen's chipmunk	*Tamias senex*	
bushy-tailed woodrat	*Neotoma cinera*	N. CA and S. Oregon [b]
dusky-footed woodrat	*Neotoma fuscipes*	N. CA and S. Oregon [b]

(Continued on next page. See notes at end of table.)

Table 7.2 Continued

Common Name	Scientific Name	Notes
white-footed vole	*Phenacomys albipes*	
long-tailed vole	*Microtus longicaudus*	
Oregon vole	*Microtus oregoni*	early seral [a]
western jumping mouse	*Zapus princeps*	early seral [a]
Pacific jumping mouse	*Zapus trinotatus*	early seral [a]
black bear	*Ursus americanus*	
raccoon	*Procyon lotor*	
ringtail	*Bassariscus astutus*	
American marten	*Martes americana*	
fisher	*Martes pennanti*	
ermine	*Martes erminea*	seasonal [c]
gray fox	*Urocyon cineroargenteus*	
Roosevelt elk	*Cervus elaphus roosevelti*	seasonal [c]

[a] Species is associated with early seral habitats, but is often more abundant in riparian areas due to frequent disturbances and thus early seral habitat characteristics in riparian areas.
[b] Species appears to be associated with riparian habitats only in the drier regions of northern California and southern Oregon.
[c] Species appears to be associated with riparian habitats seasonally.

The high degree of association between riparian habitats and mammalian species can be explained in part by riparian areas having predictable sources of water, abundant streamside insects, favorable microclimates, and high plant compositional and structural diversity (Kauffman et al. 2001; Anthony et al. 2003). Structural features that appear to enhance habitat value of riparian areas include large amounts of cover of deciduous shrubs and ferns, logs and woody debris piles, and snags (McComb, Chambers, and Newton 1993; Steel, Naiman, and West 1999; Hayes 2003).

Although bats are not typically considered riparian obligates, riparian areas are of particular importance to bats because they provide abundant insects for feeding and reliable sources of open water for drinking (Christy and West 1993; Waldien and Hayes 2001; Hayes 2003). The amount of use of any particular riparian area is dependent partly on the size and structure of the stream channel and the age or structural condition of both the riparian area and the surrounding forest. Bats forage more frequently over still water than over moving water and bats that forage close to water may avoid areas with surface clutter (e.g., rocks, woody debris) as well as sections of streams that generate a large amount of surface noise (Hayes 2003). In a study of bat use of intermittent streams in Douglas fir forests of northern California, Seidman and Zabel (2001) found that bat activity was greater over streams with channel widths > 1.8 m than over streams with channel widths < 1.2m or in upland areas. Relative to stand age and stand structure, bat activity tends to be greater in old growth than in young forests, and levels of bat activity appear to be negatively influenced by tree density (Grindal 1998; Humes, Hayes, and Collopy 1999; Erickson and West 2003; Hayes 2003). However, low levels of use by myotis bats and high levels of use by non-myotis bats of small clearcuts in western Oregon (Hayes and Adam 1996), suggests that the relationship between bat activity and tree density may be species-dependent and that for myotis bats, some intermediate density of trees may be optimal.

Mammals play an important role in actively shaping riparian habitats through activities such as selective herbivory, predation, burrowing, and trampling (Kelsey and West 1998; Kauffman et al. 2001). Mammals, especially carnivores that feed on salmon carcasses, play an important role in transferring nutrients from riparian to upland habitats (Anthony et al. 2003). Beavers are of particular importance. The beaver is considered a keystone species because it can dramatically alter riparian zones through selective herbivory and damming of creeks (Kauffman et al. 2001; Hayes and Hagar 2002). Their activities shape plant communities and the influence of many beavers within a single basin can significantly change habitat and hydrological conditions (Kelsey and West 1998). Impoundment of water results in creation of ponds, retention of organic matter and sediment, death of trees, and creation of meadow habitat (Hayes and Hagar 2002). These features are in turn utilized by a variety of other wildlife species. Amphibians, fish, and aquatic birds utilize pond habitat, snags are used for nesting by cavity-nesting birds and for roosting by bats, and meadows are utilized by several small mammal and amphibian species (Suzuki 1992; Anthony et al. 2003). Beavers are not found in all riparian habitats, but instead prefer low gradient streams (< 3% gradient) in valleys about 25 to 30 m wide and with streams 3 to 4 m wide (Suzuki and McComb 1998).

Large ungulates such as elk and deer also have potential to influence riparian plant communities through selective herbivory. Ungulates tend to forage selectively on the most palatable hardwoods and conifers and in some cases heavy foraging can impede efforts in riparian restoration (Brookshire et al. 2002; Anthony et al. 2003). In addition, heavy use by ungulates can impact soil and affect stream channel structure (Kauffman et al. 2001; Anthony et al. 2003).

7.4.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

Very few studies have examined the effects of timber harvest in riparian areas on mammals. Studies have been conducted on the effects of logging in riparian areas on bats in Oregon (Hayes and Adam 1996) and western Washington (West 2002b). Both studies found that activity levels of myotis bats were higher in wooded riparian areas than in adjacent clearcuts, whereas the reverse was true for non-myotis bats (mostly silver-haired bats [*Lasionycteris noctivagans*]). West (2002b) also found that level of bat activity did not differ between streams with wide (> 30m) and narrow (< 30m) buffers. This suggests that where myotis bats are negatively affected by clearcutting, retention of even narrow buffers should be beneficial.

West (2002a) studied the effects of logging in riparian areas on small mammals in western Washington. He determined that clearcut logging adjacent to riparian areas resulted in high species turnover and declines in abundance of forest-associated species of small mammals, including the marsh shrew (*Sorex bendirii*), Trowbridge's shrew (*Sorex trowbridgii*), shrew-mole (*Neurotrichus gibbsii*), and the forest deer mouse (*Peromyscus keeni*). However, retention of even narrow buffers appeared to be effective in maintaining many forest-associated species. Stream buffers appeared to provide habitat that was intermediate in quality between uncut control stands and clearcut uplands for forest-associated small mammals.

To my knowledge, no other studies in the Pacific Northwest have directly examined the effects of forestry in riparian areas on mammals. However, Anthony et al. (2003) have suggested that riparian buffers also may serve as refugia, but that to be effective for such a function, buffers would need to be large enough and to retain sufficient trees and shrubs to allow riparian-associated species to persist until sufficient tree canopy cover is reestablished in adjacent uplands. Riparian buffers likely will not be effective refugia for all mammal species, however, because some species are associated with upland forests (e.g., western red-backed vole [*Clethrionomys californicus*], McGarigal and McComb 1993; Hayes and Hagar 2002). Riparian buffers may also function as travel corridors. The role of riparian areas as travel corridors is well established for river otter, raccoons, black bears, bobcats, and deer (Anthony et al. 2003). The role of stream buffers to maintain function of riparian zones as travel corridors is not well studied, but has been documented for black-tailed deer (*Odocoileus hemionus columbianus*) in northern California (Loft, Menke, and Burton 1984) and for American marten (*Martes americana*) in Newfoundland (Forsey and Baggs 2001).

Watershed Scale Considerations. Only a few studies have assessed the role of landscape composition and configuration on mammals in the Pacific Northwest. In a study of associations of bats with local and landscape features in western Oregon and Washington, local stand structural conditions appeared to be more important than landscape features in influencing bat activity (Erickson and West 2003). In a study that examined patch and landscape level (sub-basin) habitat associations of small mammals in the Coast Range of Oregon, Martin and McComb (2002) found that red tree voles (*Arborimus longicaudus*), California red-backed voles (*Clethrionomys californicus*), and shrew moles were most frequently associated with unfragmented landscapes, deer mice (*Peromyscus maniculatus*) and white-footed voles (*Arborimus albipes*) were associated with fragmented landscapes, and marsh shrews, Pacific shrews (*Sorex pacificus*), fog shrews (*Sorex sonomae*), Trowbridges's shrews, vagrant shrews (*Sorex vagrans*), and Pacific jumping mice (*Zapus trinotatus*) were unaffected by landscape pattern. It appears that riparian-associated small mammals were associated more with local patch and microsite characteristics than landscape level patterns, with a possible exception of the shrew mole.

7.4.3 *General Guidelines for Sustainable Forestry Practices*

To ensure the maintenance of riparian-associated mammal communities within forested landscapes, retention of unharvested riparian buffers apparently is important when clearcutting adjacent to streams. Although current research suggests that even narrow buffers may be effective in retaining small mammal and bat communities in riparian areas, long-term studies of the effectiveness of riparian buffers are needed. Trees in riparian buffers, especially narrow ones, appear especially susceptible to blowdown (Bunnell, Kremsater, and Wind 1999; West 2000a, 2000b), and extensive windthrow of buffer trees reduces functionality of buffers (Kelsey and West 1998). Thus, whereas narrow buffers may be effective initially, wide buffers may be needed for long-term effectiveness. Bunnell, Kremsater, and Wind (1999) suggested that buffers dominated by hardwood trees may be more wind-firm than are buffers dominated by conifers. McComb, McGarigal, and Anthony (1993) suggested that buffers at least 50m wide would provide at least marginal habitat for the small mammal species that they studied. Kelsey and West (1998) suggest that buffers of 100–150 m wide will be needed to maintain preharvest microsite conditions.

Little information is available to inform guidelines to retaining mammalian biodiversity at large scales (e.g., watersheds). However, Martin and McComb (2002) suggest that landscapes that provide a full range of vegetation patterns (levels of fragmentation) and composition (variety of stand types and forest ages) will maximize species richness over large scales. Because some species of mammals are associated with upland habitats (e.g., red tree vole), use of only riparian buffers to provide travel corridors or late-seral forest refugia may not be effective to maintain biodiversity over landscapes managed for timber production (McComb, McGarigal, and Anthony 1993; Hayes and Hagar 2002).

Plantations bordering riparian areas may influence use of these areas by mammals. Plantations in the closed-canopy stem exclusion stage are structurally simple, are thought to host relatively few wildlife species, and no species is known to depend on this developmental stage (Hansen et al. 1991; McComb, Spies, and Emmingham 1993; Hayes et al. 1997). However, the value of even-aged plantations can be increased through retention of large-diameter trees, hardwood trees and shrubs, snags, and logs at time of harvest, and structural diversity can be increased through time by thinning (McComb, Spies, and Emmingham 1993; Carey and Curtis 1996; Hayes et al. 1997).

7.4.4 *Information Needs*

Although many studies have documented associations of mammals with riparian areas, information is still lacking on the influence of timber harvest on riparian obligate and associated mammals. To my knowledge, only one study has examined the role of riparian buffers on mammals in the Pacific Northwest (a multi-taxa study with effects on bats and small mammals reported in West 2000a, 2000b). Similar studies are needed elsewhere and may be especially pertinent in the more xeric areas of the Pacific Northwest where the degree of association with riparian areas may be greater (Anthony et al. 2003).

Long-term studies are needed to examine effectiveness of buffers through time. To date, West (2000a, 2000b) has examined the influence of stream buffers only through two years post-harvest. It is possible that delayed influences of harvest may occur or that influences may decline over time as upland habitats become reforested. Long-term studies should also consider longevity of buffers, factors that influence windthrow, and implications of buffer longevity on the ability of buffers to continue to retain riparian- and closed canopy forest-associated wildlife communities in riparian areas.

Information is needed on the effects of partial harvest (e.g., commercial thinning, shelterwood harvest, individual tree selection) in riparian zones. To date, research on the influence of forest practices on wildlife in riparian zones in the Pacific Northwest has focused on effects of clearcutting and the influence of buffer width on bat and small mammal communities. Studies are needed on both a) influences of buffer width within the context of partial harvest in uplands, and b) the influence of partial harvest within both uplands and riparian zones (i.e., with no buffer). Both types of studies should explore varied levels of partial harvest (e.g., individual tree selection to commercial thinning or shelterwood harvest) and incorporate multiple taxa of mammals (e.g., carnivores, ungulates, bats, small mammals).

Current research has focused on effects of riparian zone management with regard to state forest practices rules (e.g., West 2000a, 2000b). There are currently no studies that compared prescriptions of state forest practices rules to those implemented under the Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994). It has been suggested (e.g., Hayes and Hagar 2002) that continuous fixed-width buffers and use of only riparian buffers as refugia may not be effective or efficient approaches to maintaining biodiversity over large scales. Both observational and manipulative experiments are needed to address the relationship of various riparian management alternatives and response by wildlife. Kelsey and West (1998) suggest that maintaining islands of leave trees rather than continuous fixed-width buffers may be an effective approach to preserve microsite conditions and reduce risk of windthrow while also retaining both riparian and upland habitats. This approach is one alternative that could be studied and compared to various buffer retention strategies.

7.5 Birds

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7.5.1 *Introduction*

Riparian areas provide important habitat for bird communities in the Pacific Northwest. Although riparian zones make up only 1-2% of western landscapes, they provide breeding habitat for more species of birds than any other vegetation type (Kauffman et al. 2001). In Oregon and Washington, it is estimated that 266 species of birds occur in riparian habitats and that 103 of those species are closely associated with riparian habitats for breeding and foraging (Kauffman et al. 2001). In arid regions, a large proportion of bird species is more abundant or completely restricted to riparian areas (Knoph 1985). However, in wetter portions of the Pacific Northwest (Appendix A) where moisture is abundant due solely to high rainfall, fog, and high density of small streams, comparatively fewer bird species are dependent on riparian areas (McGarigal and McComb 1992). Nevertheless, riparian areas in the Pacific Northwest are often more structurally diverse and support more deciduous and berry-producing shrubs, important food and nesting resources for birds, than do upland areas (Kelsey and West 1998; Lock and Naiman 1998). Riparian areas also frequently contain abundant large logs and woody debris piles which function as important structures for nest sites of winter wrens (*Troglodytes troglodytes*) (Waterhouse 1998) and as foraging and resting perches for many bird species (Steel, Naiman, and West 1999). Riparian habitats in the Pacific Northwest are particularly important for neotropical migrating birds; approximately 60-85% of neotropical migrating birds in the western United States breed in woody, deciduous riparian vegetation, and riparian areas are used more than are upland habitats by birds during migration (Kauffman et al. 2001). Riparian habitats also are important for raptors; roughly 50% of raptor species in western Oregon and Washington breed and 60% forage primarily in some type of riparian habitat (Knight 1988).

In the Pacific Northwest, composition of bird communities in riparian habitat is related in part to the size of the stream. Rivers and larger streams seem to be more important in providing habitat for riparian obligate bird species such as herons, ducks, and kingfishers than are smaller streams (Kelsey and West 1998; Hayes and Hagar 2002). In a comparison of bird communities between large and small rivers in western Washington, Lock and Naiman (1998) found that bird species richness and total abundance was higher in riparian areas of large rivers (active channel width 67-140m) than smaller rivers (active channel width 12-21 m). They also found that the ratio of deciduous to coniferous cover was a good predictor of bird species richness, which increased with deciduous cover, and that large rivers had a higher number of unique species not found in riparian habitats of the smaller rivers. Riparian habitat adjacent to large rivers appeared to be particularly important for raptors, neotropical migrants, and deciduous-associated species such as the black-throated gray warbler (*Dendroica nigrescens*) and warbling vireo (*Vireo gilvus*) (Lock and Naiman 1998).

Plant communities in riparian areas of small rivers and streams in western Oregon are typically similar to those in the adjacent uplands in moist coniferous forests (McGarigal and McComb 1992). Consequently, bird communities in riparian habitats of small streams are frequently similar to those found in adjacent upland habitats (McGarigal and McComb 1992; Lock and Naiman 1998). In a comparison of bird communities in riparian habitats around second and third order streams with adjacent uplands, McGarigal and McComb (1992) found that upland habitats actually supported higher bird species richness and total bird abundance than did riparian areas. They found that no species was unique to headwater riparian areas and only the Swainson's thrush (*Catharus ustulatus*) and winter wren were more abundant in riparian than in upland areas. In a similar comparison in western Washington, Pearson and Manuwal (2001) found that in addition to the winter wren, American robins (*Turdus migratorius*), black-throated gray warblers, and Pacific-slope flycatchers (*Empidonax difficilis*) also were more abundant in riparian than in upland areas.

Despite the fact that no bird species appears to be unique to riparian habitats of small rivers and streams in the Pacific Northwest, these habitats still provide vegetative structure and food resources important to birds. Small river and stream riparian areas typically support greater densities and cover of deciduous trees, deciduous shrubs, berry-producing shrubs, and herbaceous vegetation than do adjacent uplands.

7.5.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

There is relatively little research on the relationships between timber harvest and birds in riparian areas of the Pacific Northwest, and thus far, research has focused on the function of riparian buffers to maintain pre-logging bird communities. In addition, research in the Pacific Northwest has been limited to examining buffers only along smaller rivers and streams.

Although no research has been conducted specifically on the influence of forestry on birds within riparian habitats of large rivers, there is some evidence suggesting that certain forest practices within this zone may negatively affect some species of birds. Nest sites of bald eagles (*Haliaeetus* *leucocephalus*) in Oregon and Washington are often associated with mature coniferous forest within 2 km of large rivers (Garrett, Watson, and Anthony 1993; Buehler 2000). Productivity of bald eagles in Oregon was negatively correlated with proximity to clearcuts and major logging roads (Anthony and Isaacs 1989); however, territory occupancy and productivity were not negatively influenced by selective harvest in the Klamath Basin of Oregon (Arnett et al. 2001). In a study of habitat associations of riparian obligate bird species in the Coast Range of Oregon, Loegering and Anthony (1999) found a positive association between habitat use and the presence of forested cover (trees > 5 m tall) for American dippers (*Cinclus mexicanus*), belted kingfishers (*Ceryle torquata*), mallards (*Anas platyrhyncos*), and great-blue herons (*Ardea herodias*). Saab (1999) studied the influence of landscape pattern on birds in cottonwood riparian habitat with varying degrees of fragmentation in Idaho. She determined that bird species richness was positively associated with natural and heterogeneous landscapes, large patches of cottonwoods, close proximity to other patches of cottonwoods, and microhabitats with relatively open canopies.

Logging within riparian areas of small streams has a similar effect on bird communities as does logging in upland areas. Typically, logging within riparian areas results in a change in bird communities from dominance by species associated with closed-canopy coniferous forest to a community with greater representation of open-canopy or shrub associated species of birds (Hagar 1999; Pearson and Manuwal 2001). The extent of species turnover is related, in part, to retention of buffers. In a comparison of logged and unlogged riparian areas of the Coast Range of Oregon, Hagar (1999) found that unlogged riparian areas (>30 m buffer) retained bird communities dominated by closed-canopy forest-associated species (e.g., Hammond's [*Empidonax hammondii*] and Pacific-slope flycatchers, brown creepers [*Certhia americana*], chestnut-backed chickadees [*Poecile rufscens*], winter wrens, golden-crowned kinglets [*Regulus satrapa*]), whereas in logged sites, species associated with disturbed or open habitats (e.g., Rufous Hummingbird [*Selasphorus rufus*], northern flicker [*Colaptes auratus*], house wren [*Troglodytes aedon*], orange-crowned warbler [*Vermivora celata*], MacGillvray's warbler [*Oporornis tolmiei*], dark-eyed junco [*Junco hyemalis*], and American goldfinch [*Carduelis tristis*]) became more abundant and species associated with closed-canopy forests declined in abundance.

To my knowledge, no studies have been conducted on effects of partial cutting within riparian buffers or partial cutting that extends from the uplands and into the riparian zone (i.e., with no buffer). Hagar (1999), however, suggested that thinning or partial harvest of large-diameter trees within stream buffers might negatively affect species that are positively associated with tree density, such as Pacific-slope flycatchers and winter wrens.

Retention of unharvested buffer strips between streams and logged upland habitats is one approach that is often used to maintain biodiversity in riparian areas. Although buffer strips are often left for the goal of protecting habitat for fish and other aquatic vertebrates, they also provide habitat for many species of birds. Buffer width influences bird community composition. In western Washington and Oregon, buffers > 30 m wide retained similar bird communities compared to those present prior to harvest and in unlogged controls, whereas more narrow buffers (< 30 m wide) experienced higher species turnover (Hagar 1999; Pearson and Manuwal 2001). Even wide buffers, however, may not be adequate to support all species of birds (Kinley and Newhouse 1997, Hagar 1999, Pearson and Manuwal 2001). Interior-forest species, conifer forest species, and riparian-associated species appear to decrease in abundance even within buffers > 30 m (e.g., brown creeper, golden-crowned kinglet, black-throated gray warbler; Pearson and Manuwal 2001).

Watershed and landscape scale considerations. In addition, no studies have yet been conducted on the effects of riparian forestry at large scales such as watersheds. One study, however, suggests that landscape scale effects may occur. Saab (1999) studied the relative effects of microsite, stand-level, and landscape pattern on abundance of songbirds in cottonwood-dominated riparian areas of Idaho.

Within her study sites, riparian areas were fragmented primarily by agriculture. She found that landscape pattern (e.g., habitat patch size, shape, distance to other patches, fragmentation) was the primary influence on the distribution and that occurrence of most species and was more important than site specific or microsite characteristics. Although forest cutting is not common within hardwood-dominated riparian zones, the results of Saab (1999) suggest that where cutting does occur, it may be important to consider potential effects of landscape pattern as well as the cumulative effects from multiple land uses. Metrics are available for quantifying landscape patterns (e.g., FRAGSTATS, available at http://www.umass.edu/landeco/research/fragstats/documents/fragstats_documents.html). See online documentation and McGarigal, Cushman, and Stafford (2000) for use of multivariate landscape metrics. Landscape metrics are commonly used to look at how patch size, fragmentation, etc. may affect wildlife.

7.5.3 *General Guidelines for Sustainable Forestry Practices*

More information is needed before comprehensive guidelines for sustainable forest management in riparian areas can be formulated. Because current research has been limited to use of riparian buffers, guidelines suggested here are restricted to use of riparian buffers along small streams (class 1-3). Within small stream riparian areas, current research suggests that buffers at least 30 m wide are needed to sustain pre-harvest bird communities (Hagar 1999; Pearson and Manuwal 2001). Thus, if sustaining pre-harvest bird communities is an objective, riparian buffers > 30 m wide should be left in some places to provide adequate habitat, especially for species associated with riparian habitats, such as Swainson's thrushes, black-throated gray warblers, and winter wrens. However, because bird communities within small stream riparian areas are similar to those found upland, smaller riparian buffers may sometimes be adequate, provided that sufficient forested habitat is provided in upland areas. In fact, given that more species of birds appear to be associated with upland than with riparian habitats (McGarigal and McComb 1992), it has been suggested that a combination of riparian management zones and upland stands with retained structure may be a useful strategy (Hayes and Hagar 2002).

Plantations in the closed-canopy stem exclusion stage are structurally simple and are thought to host relatively few bird species compared to more heterogeneous forests. No species is known to depend on this developmental stage (Hansen et al. 1991; McComb, Spies, and Emmingham 1993; Hayes et al. 1997). However, the value of even-aged plantations can be increased through retention of large-diameter trees, hardwood trees and shrubs, snags, and logs at time of harvest; structural diversity can be increased through time through thinning (McComb, Spies, and Emmingham 1993; Carey and Curtis 1996; Hayes et al. 1997).

7.5.4 *Information Needs*

Relatively little is understood about the effects of forestry within riparian zones on birds. Most research has focused on the role of riparian buffers to conserve pre-logging bird communities in headwater streams. There appear to be four major information gaps with regard to the influence of forest practices in riparian zones on birds. First, research is needed to determine potential impacts of forestry on birds in riparian areas of large streams and rivers (stream class 3 and above). As noted above, most riparian obligate and many riparian associated species of birds are associated with the open water and riparian habitat of large streams and rivers; however, little is known regarding the potential influence of forest practices on these species.

Second, information is needed on the effects of partial harvest (e.g., commercial thinning, shelterwood harvest, individual tree selection) in riparian zones. Research on the influence of forest practices on wildlife in riparian zones has focused on effects of clearcutting and the influence of buffer width. With the exception of Arnett et al. (2001), no information has been published on the influence of partial harvest on riparian-associated wildlife. It is likely that partial harvest in both riparian and upland habitats will have less influence on microclimate and habitat within riparian zones than does clearcutting and that these differences would be reflected by responses of bird species. Studies in upland habitats suggest that commercial thinning does not have a dramatic negative effect on bird abundance (e.g., Hagar, McComb, and Emmingham 1996; Hayes, Weikel, and Huso 2003). However, of the species that were negatively affected by thinning, some are generally considered to be riparian-associated species (e.g., Pacific-slope flycatcher and black-throated gray warbler). Thus, partial harvest both in uplands and in riparian zones may have larger implications for some species' sustainability. Studies are needed on both a) influences of buffer width within the context of partial harvest in uplands, and b) the influence of partial harvest within both uplands and riparian zones (i.e., with no buffer). Both types of studies should explore varied levels of partial harvest (e.g., individual tree selection to commercial thinning or shelterwood harvest).

Thirdly, information about the ability of riparian buffers to conserve bird populations over the long term is needed. Although research has shown that wide buffers can be used retain pre-logging bird communities, no research has yet been conducted on whether buffers are effective in conserving these species as adjacent forests regrow over the long term or whether reproductive productivity of birds within buffers is affected. The positive association of Steller's jays (*Cyanocitta stelleri*), a known predator of bird nests, with riparian buffers in Oregon (Hagar 1999), suggests that nest depredation within buffers may be a concern. Pearson and Manuwal (2001) and Hagar (1999) have indicated that research is needed on nest depredation and productivity of birds within riparian buffers. Research is also needed on population viability of species that decrease in abundance with decreasing buffer width.

Lastly, current research has focused on effects of riparian zone management with regard to state forest practices rules (e.g., Hagar 1999; Pearson and Manuwal 2001). There are currently no studies that compare prescriptions of state forest practices rules to those implemented under the Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994). Both observational and manipulative experiments are needed to address the relationship of various riparian management alternatives and response by birds.

7.6 Invertebrates

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7.6.1 *Introduction*

Animals without backbones comprise most of what has been termed "the hidden 99.5%" of life's diversity, and have been described as the "little things that run the world" (Wilson 1987). Not surprisingly, invertebrate life forms are important parts of the biodiversity spectrum that greatly influence the productivity and function of forests (Marcot 1997; Showalter et al. 1997). This section will focus on Arthropods (i.e., the Phylum that contains insects, arachnids, and crustaceans), which are by far the most diverse group in any forest landscape (Showalter et al. 1997). In fact, there are many times more species of native arthropods than all the vertebrates, higher plants and macro-fungi combined. Although the present treatment does not discuss all invertebrate life forms, it does not imply that often overlooked invertebrates are not important elements of biodiversity. For example, Frest (2002) recently described the importance of native snails as indicators of ecosystem health. Many mollusk species are local endemics (Marcot 1997); presumably a large number of flightless arthropods are likewise endemic (e.g., amaurobiid spiders, Melanopus grasshoppers, caseyid millipedes).

Because of the vast number of taxa and life history variations, the invertebrate discussion here will be general, but sources of more specific information are provided.

Invertebrates inhabit all environments from the mineral regolith underlying riparian forests to the upper reaches of the canopy and beyond. Although relatively large herbivores and predators are the most conspicuous invertebrates in any forest, the more inconspicuous detritivore group is the richest in species. As a general rule, the "soil" is the most diverse component of the riparian ecosystem, hence the focus here. There are probably in excess of 10,000 species of arthropods within the region. Most of these arthropods are intimately associated with the soil/litter for at least part of their life cycles. Most are found in both riparian and upslope habitats, but there are differences as described below.

No comprehensive listing exists of these species, nor is there any systematic compilation of the relevant literature (Moldenke and Ver Linden 2002). For an ongoing literature review and database development project on Pacific Northwest Arthropods, Moldenke and Ver Linden have searched the databases (tens of thousands of references) for major taxonomic groups: spiders (Araneae), turtle-mites (Oribatida), carabid ground-beetles (Carabidae), longhorned-beetles (Cerambycidae), bark-beetles (Scolytidae), flies (Diptera), springtails (Collembola), bees (Apoidea). Additionally, they searched the database for inventory techniques and soil ecological terms in order to compile an annotated bibliography of the most pertinent references (most recent continually updated version subsequent to this one may be found at http://www.ent.orst.edu/moldenka). Although this database does not focus strictly on riparian species, it does serve as a valuable source of information about arthropods in riparian areas.

Despite gaps in knowledge, it is clear that mesic forests west of the Cascade Crest, from which fire has been excluded, have very deep litter layers and quite probably support the highest densities and diversities of soil arthropods anywhere in the world (Moldenke 1999). Though the relative arthropod biomass cornucopia of the stream habitat versus the upland has been documented in desert environments, it seldom has been quantified in temperate forest habitats. However, it is likely that the same phenomenon occurs in the Pacific Northwest conifer biome. Studies by Brenner (2000) and Moldenke (in prep) in the past decade have shown that both the diversity and the abundance of large terrestrial arthropods increases dramatically as one approaches a stream.

These higher densities are comprised of two distinct types of species: a) those restricted in all of their activities to the terrestrial-riparian zone; and b) those typical of the upland but which visit the riparian zone presumably in search of both more abundant food and available drinking water. There is probably a far higher level of available arthropod biomass in the ground-surface stratum than in either the foliage-gleaning or the fly-catching strata. It is known that the arthropod species inhabiting this zone are active at distinct times of the year, and that species richness of arthropods is extraordinarily high. In fact, the Moldenke and Joseph Furnish studies (unpublished) have shown that terrestrial-riparian zones are twice as species rich as upland forest floor areas, and that nearly 100% of the true forest floor species occur within the terrestrial-riparian zone (>75% of them are actually more abundant within the terrestrial-riparian zone). Microclimatic heterogeneity plus habitat diversity created by the unpredictability of streamflow events (as well as an abundance of food resources) causes the significantly enriched biodiversity in the streamside environment (Antvogel and Bonn 2001; Collinge et al. 2001).

Arthropods are of particular significance to all biodiversity as well for a multitude of reasons. Perhaps most critical are:

1) They play an important role in the development of soil organic matter as regulators of microbial growth, nutrient decomposition and plant growth. The long-term health of any ecosystem is intimately dependent upon minimizing the rate of nutrient loss: the greater the biomass of soil

microbes, the slower the rate of nutrient loss. The progressive decrease in soil organic matter (both living and dead) in agricultural soils in North America is a major factor leading to chemical pollution in waterways and aquifers. Soil degradation is associated with a decrease in soil fauna diversity (Coleman and Crossley 1996; Benckiser 1997).

2) They are a major food source for vertebrates and are therefore vital to trophic transfer. Insects on the wing are the prime food resource of fly-catching birds and bats. In large regions of the forested Northwest, both the abundance of shrubs and caterpillar biomass increase as one approaches a stream channel (Jiquan Chen, Department of Environmental Sciences, Michigan Technical University, unpublished data). There is a greater diversity of vertebrates utilizing the terrestrial ground-surface arthropod food resource than either the foliage or aerial environments.

3) They provide trophic sustainability through pollination. Pollination services are often taken for granted. It is often assumed that plants are being efficiently pollinated. However, many environments are pollinator-limited—even within environments richly characterized by pollinators, a large percentage of species lose out in competition to species with big showy flowers (Moldenke 1975, 1976a, 1979b, 1979c). Pollinator services are thus of concern in all environments. The vegetation zones of the Pacific Northwest have rarely been characterized with respect to their pollinator abundances and behaviors, unlike those in California (see Moldenke 1975, 1976a, 1979a). Marcot et al. (1998) estimated that in the interior Columbia River Basin, invertebrates pollinate most of the rare or potentially rare vascular plants (66%), and about half of these plants (33%) are pollinated by solitary bees (Rathcke and Jules 1993; Bond 1994; Haynes, Graham, and Quigley 1996; Spira 2001). In this region, insects as a whole play vital roles in reproduction of rare flowering plants, whose viability depends on the presence of their invertebrate pollinators and dispersal mutualists.

4) They function as disturbance agents. Bark beetles and associated fungi are often critical in the formation of forest gaps through tree mortality, increasing habitat heterogeneity, and allowing light to reach the forest floor. This creates opportunities for a different suite of herbs and shrubs in the understory. The downed trees become decomposing logs, which provide for numerous species, not only of arthropods and fungi, but nitrogen-fixing bacteria, etc. Trees killed by insect/pathogen interactions may also facilitate fire occurrence and promote heterogeneity in fire effects (Agee 1993).

7.6.2 *Ecological Processes and Habitat Features Related to Forestry Practices*

Two general management practices will have the most significant and farthest-reaching effects in Pacific Northwest forestry: timber removal and prescribed fire (including both broadcast and pile burning of slash). Regardless of the logical and pertinent arguments that compare these processes to natural disturbance events, it is recognized that many aspects of both are not similar to natural disturbances. *A priori* we know that forest management practices all have strong effects, either directly on species (e.g., effects of heavy machinery on soil organisms) or indirectly through initiating successional changes (i.e., there are open-canopy taxa and closed-canopy taxa) and associated changes in microclimate, litter depth, or habitat heterogeneity. Ongoing studies by Niwa and Rappaport (unpublished) are an excellent start to understanding the role of controlled underburning on soil macro- and micro-fauna. Niwa's studies have revealed either no detectable effect on most individual species and functional groups area-wide of either spring or fall underburning or small quantitative effects which rapidly return to pre-burn conditions, in contrast to Miller and Moldenke's previous studies which showed legacies lasting 35+ years from "hot" site-prep burns (Estrada-Venegas 1995). It may also be instructive to compare results from these studies with results from similar studies in other regions (e.g., Kalisz and Powell 2000; Wikars and Schimmel 2001), particularly regarding influence of management practices on arthropod functional groups and ecological roles.

The structural and microclimatic changes resulting from logging disturbance have differing implications for arthropod diversity that are strongly dependent upon the habitat needs of a specific species group. For example, for some flying arthropods which depend upon sunlight for warming of flight muscles (e.g., butterflies), the increased insolation associated with disturbances may be favorable (Meyer and Sisk 2001). Yet other species may be negatively affected by the same disturbance if they are dependent upon undisturbed soil, tree boles, or downed wood for food or habitat.

Forestry practices will affect both the community of soil arthropods and the soil processes in which they participate in direct proportion to the intensity of the disturbances. In forest environments the litter layer is apparently positively correlated with arthropod density and diversity in most cases (Madson 1997). Forestry practices (such as clearcutting, underburning, and fuel removal) may initially decrease total density of soil-dwelling arthropods by 75-90% and decrease species richness by more than 50% (Moldenke et al. unpublished data). Deep soil compaction (through mechanized vehicle use and log skidding) is probably the most radical change in any environment to soil arthropods. The ability of species to recover from such impacts is not known.

Does protection of riparian buffers actually protect the entire forest floor fauna? The actual functionality of riparian buffer widths for maintaining diversity and population densities of both aquatic and terrestrial-riparian arthropods needs to be assessed in both the mesic forests of the Northwest and the more xeric forests of the eastern Klamath region and the east side of the Cascades (see research of Rykken and Moldenke, unpublished). Studies by Chan and Olson and others (Olson et al. 2000a, 2000b, 2000c; Tappeiner, Olson, and Thompson 2000; Rundio and Olson 2001) have shown that the diversity of terrestrial-riparian microhabitats is positively related to increased frequency and amplitude of natural disturbance. Future management of riparian zones needs to focus on appropriate levels of habitat disturbance and overall effects (i.e., habitat heterogeneity, legacy retention, and physiological stress) within the riparian zone on arthropod diversity. Literature reveals that in Europe and eastern North America, the terrestrial-riparian fauna varies by stream order, canopy cover, and disturbance regime (Erwin, Ball, and Whitehead 1979; Desender 1994).

Intensive forestry has been shown to affect both the diversity and functional role of soil arthropods. Springett (1976) compared the diversity and abundance of soil arthropods and litter decomposition in natural woodlands and pine plantations. Although there was no clear relationship found between arthropod abundance and decomposition, there was a significant correlation between species diversity and decomposition rates. Notably, there was a large effect at low diversity values, and a decreasing effect at more usual levels of diversity. This has been taken as evidence that a certain minimum number of species may be required for full ecosystem function (Davis et al. 1996), which supports theoretical models relating diversity and functioning of ecological processes (Vitousek and Hooper 1993).

7.6.3 *General Guidelines for Sustainable Forestry Practices*

Any management practices that tend to decrease stream heterogeneity are likely to alter arthropod diversity. The microhabitat associations of riparian arthropod species still need to be documented especially as they relate to stream width and lateral distribution of species into the upland forest floor (current unpublished research by Moldenke, Chan, and Olson). It must be emphasized that the richness of riparian-associated terrestrial species far exceeds the richness of true aquatic species.

There are sensitive arthropod species, assemblages, or communities that appear excluded from present management set-asides that should be considered. Most species that are “sensitive” to soil disturbances in unprotected natural sites are probably restricted to areas without a forest canopy. However, there still exists the question of what sensitive taxa might be restricted to forested habitats with commercial potential (the cryptic-sensitives); hence the need to inventory and sample in a statistically rigorous fashion. Riparian areas are the most likely areas to support habitat-restricted arthropod taxa, regardless of canopy cover. Ancillary wetland habitats (i.e., bogs, marshes, etc.) perhaps sustain even higher species richness and endemicity of terrestrial arthropods (e.g., the majority of species of ground-beetles [Carabidae and Staphylinidae] are associated with microenvironments that are basically "riparian") and need appropriate attention in practical measures to protect riparian biodiversity as well (Lindroth 1961-69; Thiele 1977; Stork 1990; J. Richardson, Dept. Entomology, University of British Columbia, Vancouver, BC., in prep.).

Having a regional forest conservation program in place does not necessarily mean that very localized endemic species and subspecies will be protected. Historically, the elevated biodiversity of arthropods has functioned to hinder community-wide faunal analyses. Current development of computer imaging algorithms is at the point when the entire aquatic insect community of the region will be recognizable at the level of species, once a data file of appropriate pictures can be assembled.

7.6.4 *Information Needs*

A substantial percentage of the forest land base is/will be tied up in riparian buffers. How wide must buffers be to protect aquatic and terrestrial-riparian invertebrates? How do requirements of these individual species differ between headwater/first order, third order, and fifth order/rivers and streams? More studies like those conducted in central Oregon by Furnish/Progar/Moldenke (Progar and Moldenke 2002) need to be initiated. These have documented that: a) temporary headwater streams produce higher densities of invertebrates (i.e., migratory bird food) than continuous-flow primary streams; b) continuous-flow primary streams have higher levels of diversity/species richness than temporary ones; and c) the unique species in the temporary streams are not true aquatic species, but are related to terrestrial-riparian species—whether they are unique or widespread in terrestrial habitats is unknown.

If biodiversity-linked assays are critical anywhere within the forested Pacific Northwest, then they must be critical in the Klamath region. Studies on soil arthropods conducted elsewhere (e.g., Postle, Majer, and Bell 1991; Deharveng 1996; Pankhurst, Doube, and Gupta 1997; Neher et al. 1998; Bird, Coulson, and Crossley 2000; Haskell 2000) may not necessarily directly pertain to conditions in the Klamath region. Quaternary history has seen alternating northward and southward migration of thousands of arthropod taxa associated with the glacial cycles. Throughout this long and heterogeneous geological period, the Klamath region has served as refuge for both Arcto-Tertiary (Temperate) and Madro-Tertiary (Tropical) fauna. Additionally, the region's edaphic heterogeneity has served as a focus for neoendemic radiation of species (e.g., *Malanopus* grasshoppers).

8.0 KEYSTONE AND ENDANGERED SPECIES

8.1 Keystone Species and Related Concepts

In this section, we discuss the concept of keystone species and related topics of keystone processes and habitat features. Specific effects of keystone species such as beavers and ungulates on vegetation are described in Section 7.1, and on wildlife habitat in Section 7.4. Keystone concepts are also described for insect assemblages in Section 7.6.

Specific management for all elements of biodiversity is impossible. This logistic reality has led to great interest in keystone species or endangered species as surrogates for biodiversity as a whole. In concept, keystone species are those whose removal would cause a disproportionate alteration of a critical biological process, and presumably, loss of biodiversity. In many cases, these species are required to maintain a trophic balance, such as top predators that exert controls on herbivore populations which in turn, govern vegetation structure and composition of entire ecosystems (Perry 1994; Thompson and Angelstam 1999). Examples of such top predators include wolves, which control moose, which exert secondary effects on vegetation and soils on Isle Royale in Lake Superior (Brandner, Peterson, and Risenhoover 1990), and sea otters, which prey on sea urchins, and allow persistence of kelp forests along the Pacific Coast (Estes and Palmisano 1974). Other species, such as beavers or earthworms, that function as ecosystem engineers are also considered keystone species (see Sections 7.4 and 7.6). Beyond these apparently clear examples (e.g., there is much more to Isle Royale plant/animal dynamics than interactions between moose and wolves), there may be many other species that play disproportionate roles in maintaining biodiversity, through their effects on nutrient capture (e.g., nitrogen fixation in *Alnus, Ceanothus* spp.), large wood development (Douglas fir, Port Orford cedar, redwood and other large conifers), or water and nutrient capture by plants (various mycorrhizal fungi species). More recently, the keystone concept has been expanded to include keystone ecosystems (e.g., riparian forests, aspen groves, wetlands), or processes (e.g., fire, flooding dynamics) as landscape factors associated with high biodiversity (Perry 1994; Stohlgren et al. 1997). In other cases, the impacts from introductions of non-native species can be so great that they act as artificial keystones (e.g., Port Orford cedar root rot (*Phytophthora lateralis*); giant reed (*Arundo donax*) in riparian forests of California).

The premise that single species or small groups of species function as strong controls on ecosystems is controversial, and related to scientific discussions of the role of species diversity on ecosystem stability (e.g., Elton 1958; Tilman 1996). Clearly, most species perform roles that are both distinctive and to some degree redundant, whereas some others are of more singular importance. Quite aside from the problematic ethical assertion that any species has more value than another, management for only the most distinctive species may be an incomplete approach, because native ecosystems should have a degree of species redundancy in most functional roles (Tilman 1996). However, identification of potentially important species or processes that are absent or impaired may be one of the most effective starting points for restoring the biodiversity potential of a degraded riparian forest.

Although it is clear that the keystone species concept encompasses a very important and practical topic, most descriptions of keystone species are anecdotal, and probably clouded by our unbalanced knowledge of and affection for individual species. Although few would argue that charismatic species such as wolves or beavers play essential roles in ecosystems where they occur, there is little way of evaluating whether these roles are more essential than, say, those of seed caching-rodents, nitrogen-fixing bacteria, or certain species of soil arthropods. In some cases, manipulative studies have clearly demonstrated the importance of single species for biodiversity. A classic example is the study that demonstrated the role of the food-web keystone, the starfish *Pisaster*. It was only discovered to have a key role in regulating the abundance of other species in Pacific Northwest intertidal communities when it was experimentally removed. Removal caused a drastic shift in dominance, and species richness fell from 15 to 8 (Paine 1980). Such experimental approaches are illustrative, but it would both be logistically impossible and ethically untenable to advocate such species removal experiments to be applied more broadly in riparian forests of the Pacific Northwest. However, the unregulated trapping, livestock grazing, logging, and hydrologic alterations of the 1800s and early 1900s may have already removed many species, structures, or processes from riparian forests of the region, establishing, in effect, unintentional experiments.

In riparian forests in the Pacific Northwest, several factors may be sufficiently impaired to be limiting biodiversity. Past trapping of beavers and loss of associated wetland complexes may have resulted in significantly reduced biodiversity in managed forests as compared to historic levels, essentially by reducing habitat heterogeneity. Similarly, removal of large conifers and large woody debris from streamside forest, or the active channel in the case of Port Orford cedar, may have caused substantial changes in the habitat quality of streams and floodplains, with important implications for many elements of biodiversity. In drier locales, fire may represent a keystone process that has been affected, with unknown effects on biodiversity. Introduced plant or animal species such as tree of heaven (*Ailanthus altissima)* and the brown-headed cowbird (*Molothris ater*) may presently be exerting strong influences on the biodiversity of riparian forests in parts of the region. In each of these examples, restoration of the impaired species or processes, or removal of the introduced species, may be valuable for increasing or maintaining riparian biodiversity.

Restoration ecology may provide a platform for manipulative research evaluating “keystoneness” that is both scientifically and ethically desirable. Carefully designed experiments that restore certain keystone elements (e.g., beaver, large wood, fire) and measure responses in riparian biodiversity may provide a means to test hypotheses about keystone function as well as mechanisms of ecosystem recovery.

8.2 Rare, Sensitive, and Endangered Species

Although an integrated view of biodiversity would encompass a vast array of species and processes in any landscape, rare, sensitive, and endangered species draw disproportionate attention because they are especially imperiled, are charismatic or otherwise well known, and they are protected by law. It is beyond the scope of this problem analysis to speculate on whether society should allocate special status to endangered or threatened species. That has been dealt with authoritatively in appropriate forums (USDA and USDI 1994). None of the concepts in this problem analysis challenge existing policy with regard to rare or endangered species. We support the general premise that especially vulnerable species warrant greater attention from researchers and managers, yet we aim this document toward articulation of principles for maintaining habitat for diverse species assemblages in riparian forests.

Rarity is an important characteristic of certain elements of biodiversity that may or may not intersect with forest management. As with most species occurring in riparian forests, rare species can be affected by the changes in forest structure and composition associated with forestry practices. However, they merit special consideration because for species with very restricted distributions, entire populations may be easily extirpated by forest disturbances (Thompson and Angelstam 1999). Strict preservation may also be detrimental if the species in question are associated with specific seral states or otherwise dependent upon disturbance. Managers, therefore, should seek information not only on the locations of rare species populations, but the actual structural or microhabitat requirements that the species require through time.

Sensitive species may perform an important role as indicators of subtle habitat changes associated with management. Amphibians, for example, may be among the most sensitive taxa in riparian forests of the Pacific Northwest, and may illustrate the biological significance of management changes more effectively than other prominent organisms, such as vascular plants. As Sections 6 and 7 illustrate, species differ strongly in sensitivity to similar silvicultural treatments. Sensitive species may require special consideration when planning activities in riparian forests. If management takes a completely protective approach, however, other elements of biodiversity may be detrimentally affected. Viewed at a landscape perspective, relatively stable, low stress environments and disturbed environments are probably both required for maintenance of overall biodiversity.

As mentioned above, the methods for establishing categories of extinction risk developed nationally and internationally are detailed and exhaustive (Thompson and Angelstom 1999), and exceed the scope of this document. Impacts to individual rare species and/or related ones for which applicable information exists are described in Sections 6 and 7. The general approach in recovery plans for such species is to identify and protect habitat areas considered particularly important and to manage these with an overriding goal of fostering the species recovery as illustrated by the FEMAT process (USDA and USDI 1994). It is clear from examination of the lists of threatened and endangered species, that the species most likely to find critical habitat in riparian forests or the Pacific Northwest are fish, particularly anadromous salmonids.

We refer the reader to the following websites for further reading about threatened and endangered species in the Pacific Northwest.

Federally protected threatened and endangered species in the Pacific Northwest are listed by the U.S. Forest Service's Pacific Northwest Research Station at http://www.fs.fed.us/r6/nr/wildlife/tes/list/index.htm.

Lists of threatened and endangered species recognized by the state of California are at http://www.dfg.ca.gov/hcpb/species/t_e_spp/tespp.shtml; those recognized by Oregon at http://oregonstate.edu/ornhic/2004_t&e_book.pdf; and those recognized by Washington at http://wdfw.wa.gov/wlm/diversty/soc/concern.htm.

PART III APPROACHES FOR PROTECTING RIPARIAN BIODIVERSITY

9.0 SYNTHESIS AND SUMMARY OF TAXA-SPECIFIC RESPONSES

Silviculture, timber harvest, yarding, and transportation infrastructure, through their effects on legacy retention, physiological stress, and related resource availability, can affect riparian biodiversity. How these factors affect biodiversity can vary considerably with the taxonomic group as described in detail in Sections 6 and 7. We synthesize this detailed information here.

9.1 Effects on Aquatic Biodiversity

Forestry practices, including roading, have been linked to declines in diversity of fish populations in the Pacific Northwest (Section 6.1). Primary explanations are poor egg and juvenile survival because of increased temperatures, frequent fine sediment input, and reduced legacies, primarily large woody debris in streams. Large woody debris provides energy and nutrients for fish production via decomposition processes; it also provides sediment-trapping value. Effects are complicated however, with important regional variation. In relatively cool, maritime climates west of the Cascades, dense tall conifer canopies may limit within-stream photosynthesis; as a result, invertebrate and fish productivity often increase in streams with removal of canopy vegetation. Whether this translates to competitive dominance is unclear. In drier southern or interior climates, sharp temperature rises are often noted with canopy loss. The net effects of logging practices on solar radiation, sedimentation, and hydrology appear to have important regional variation. In contrast, removal of sources of large woody debris leads to degraded conditions for spawning and rearing of salmonids in most environments. These same processes affect habitat quality for stream-dwelling amphibians to varying degrees depending on species' life history attributes and behaviors and the geomorphic stability of the stream channel (Section 6.2). Stream amphibians appear to be among organisms most sensitive to the effects of timber harvest in most climatic settings. For aquatic invertebrates (Section 6.3), community composition changes with alteration of physical factors such as fine sediment concentration and organic inputs, including woody debris. Biomass of some grazing aquatic invertebrates may increase with increasing solar radiation and consequent rises in within-stream photosynthesis, whereas removal of streamside vegetation can ultimately be detrimental to animals such as detritivores dependent on the supply of organic matter provided by streamside plants.

9.2 Effects on Terrestrial Biodiversity

Effects of timber harvest on terrestrial species that occupy riparian areas for most or all of their life cycles also show considerable variation by life form, type of harvest, and location. For vascular plants, the direct effects on merchantable tree species depend on harvest and regeneration dynamics in a particular location. Effects on other species will depend on soil disturbance and physical damage incurred during harvest and yarding. Intensity of disturbance effects and legacy retention will have important influences, as will related, widely varied rates of recovery. In addition, the changes in physiological stresses and growth resources associated with disturbance will vary geographically. Following harvests, composition shifts to fewer shade-tolerant species and an increase in ruderal and non-native species (Section 7.1). Depending on legacy retention, physiological stress, and non-native species, recovery of biodiversity may be hindered.

Non-vascular plants (Section 7.2) generally are more shade-tolerant and less tolerant of heat than most vascular plants. Consequently, they are often more sensitive to removal of shade-casting vegetation. Moreover, tree boles and downed wood serve as substrata to these species, so they are affected by both physiological stress and resource availability following timber harvest. Fungi are species rich, with many forms occurring in all successional stages of native forests (Section 7.3). However, as heterotrophs, fungal species are strongly dependent on specific carbon sources in the form of woody debris, soil organic layers, or mycorrhizal host plants of a variety of species. Where forest management removes or changes these carbon sources, fungal biodiversity can be affected.

The responses of terrestrial wildlife to silvicultural activities in riparian areas are often uncertain and complicated (Sections 7.4 and 7.5). Many species use riparian areas to varying degrees. Both riparian conditions and those of adjacent uplands will be important. Riparian areas along large streams are especially important to birds. A number of wildlife species may be keystones, with beavers being the most obvious. They enhance local and landscape scale diversity. Effects of forestry practices on terrestrial arthropod communities are more predictable and generally more detrimental than to other animals as the diversity and abundance of these species tend to be directly proportional to amounts of litter, soil organic matter, and understory deciduous species (Section 7.6). The extent to which arthropods are affected will depend again on yarding and other factors affecting soil disturbance. Other species of arthropods are directly affected where timber harvest or understory management removes host species.

Effects of forestry disturbances on habitat heterogeneity and quality, physiological stress, and resource availability show considerable variation across taxa groups and landscape settings. Variations in stress tolerance or requirements for resources among life history groups illustrate clearly that no single species or group approach will be most beneficial for all taxa. Moreover, because species groups show differential sensitivity to forestry practices, it will be difficult to assess overall impacts of treatments. Although an evaluation of the effects of any management activity must ultimately include a consideration of life history requirements, these data are not available for most taxa. Further, the relative influences of resource availability and physiological stress will also vary geographically (Figures 4.2 and 4.3). For example, in the drier, more variable climate of the eastern Cascades and Siskiyous, open, disturbed environments are expected to pose relatively greater physiological stress for a given disturbance intensity, frequency or size.

Despite the uncertainty and variation, it may be possible to manage so as to emphasize protection of target taxa that are most sensitive to logging disturbance in a given region. These will be species for which conditions with respect to both resource needs and physiological stress are detrimental. Amphibians, non-vascular plants and soil arthropods are very sensitive taxa because essential resources (i.e., habitat, food) are removed and supportive environmental conditions are changed by timber harvest. Moderately sensitive taxa, such as ectomycorrhizal fungi or stream fauna, may be only lightly affected by harvest in the coastal mountains but may be detrimentally affected by harvest in somewhat more severe climates, such as the Klamath region (Molina pers. com.). Still other groups, such as vascular plants and birds, may show greater sensitivity to riparian harvest east of the Cascades.

Considering these factors, along with habitat heterogeneity and quality, broader conclusions exist as well. Riparian forests with varied tree species and age classes, occasional shrub-dominated patches, and woody legacies such as snags and downed logs, have high potential to maintain biodiversity, whereas even-aged, single-species stands often lack the legacies and heterogeneity that underlie biodiversity. Of particular interest to forest managers wishing to restore biodiversity is the proportion of heterogeneity that can be manipulated by forest managers at the stand scale. Other, less changeable characteristics such as geomorphology or geological diversity may be most useful in helping to characterize areas with high potential for diversity. *A better understanding of effective habitat heterogeneity for different life history groups may be one of the greatest information needs for biodiversity conservation in riparian forests.*

10.0 STAND-SCALE APPROACHES FOR PROTECTING BIODIVERSITY

Species habitat is strongly governed by availability of limiting resources such as energy and nutrients, which are influenced by such things as organic inputs in streams, the vertical and horizontal arrangement of vegetation, logs, and snags, and associated variation in plant species composition. All of these factors can be strongly influenced by stand-level silvicultural practices.

Stand-scale approaches for improving biodiversity maintenance should be effective where physiological stress can be reduced and where habitat heterogeneity, and legacy and limited resource retention can be improved. Even at the stand scale, an ecosystem perspective is needed to evaluate potential management adjustments, and determine their overall effects (Gregory 1997). Stand starting conditions and potential, as well as site-specific factors will likely weigh heavily in analyses of what treatments are effective, feasible and worthwhile. Many tradeoffs exist. For example, in some cases it may be more effective to make small management adjustments across many stands; in others, it may benefit biodiversity protection more to focus more intensively on specific locations to get the “biggest bang for the buck.” Maintaining high production in portions of the landscape may make it possible to manage the most important locations for biodiversity more strictly to restore and maintain it. Here we outline general procedures that can be employed after such tradeoffs are considered by landowners. More specific guidelines for sustaining particular taxonomic groups are discussed for each group in Sections 6 and 7. These are important to consider as well, especially for the most sensitive species (stream amphibians, non-vascular plants and terrestrial arthropods).

The primary sources of physiological stress in aquatic habitats that can be reduced are excess erosion, sedimentation, or stream temperature increases (Section 6). Maintaining partial canopy cover ameliorates the effects of stream heating caused by canopy removal, and riparian buffer strips up to 30 m wide provide comparable shading to old-growth forest (Beschta et al. 1987). Forest roads may have a larger effect on both chronic and episodic erosion and sedimentation than harvest alone (Section 6). Minimizing road construction in riparian areas and along steep terrain potentially susceptible to mass failure can significantly reduce potential physiological stress from such effects.

In terrestrial portions of riparian forests, silvicultural treatments of various intensities can create distinct seral environments. Where herbicide use has been applied judiciously, even-aged systems often create floristically diverse early seral communities that have higher herb, shrub, and deciduous tree cover than older stands. These may be biologically rich environments if residual features of the former stand, such as logs, snags, or mature green trees remain. However, the greater variability in temperature in these environments may be detrimental for fungi, bryophytes, and amphibians in all except the mildest climates. Where biological legacies are lacking, they can be comparatively poor habitats for sensitive species. The stem exclusion phase of stand development that occurs after canopy closure following fires or in even-aged silvicultural systems may help shade streams, but may actually be the poorest terrestrial habitat with regard to biodiversity (Gregory et al 1987; Franklin et al. 1997). Dense young stands of conifers often lack horizontal or vertical heterogeneity in structure and composition, resulting in depauperate diversity of shrubs, herbs, and deciduous trees. Management approaches for improving terrestrial conditions include a) maintaining or adding woody debris and creating snags to more closely approximate amounts found where natural disturbance processes have operated at intermediate levels; b) enhancing structural and habitat heterogeneity by planting multiple crop species and/or leaving some native trees unharvested to remain through a second rotation; c) controlling exotic species that act as artificial keystones/pest plants to eliminate artificial keystone threats to biodiversity; d) site-preparation following harvesting that creates conditions that occur with natural disturbances and that conserve coarse woody debris to help maintain many non-crop species; and e) lengthening rotations and developing earlier thinning schedules to increase structural biodiversity. Hartley (2002) describes evidence that these methods do benefit biodiversity and may also entail various economic benefits.

Many of the valuable structural elements and biodiversity associated with early seral communities can be obtained through variable retention systems, which can be designed to cause much less physiological stress on sensitive taxa following harvest. Moreover, retaining legacies from older stands can maintain heterogeneity in structurally simple young conifer plantations (Franklin et al. 1997). Variable retention harvest regimes allow flexibility to plan harvest disturbances to recreate the array of structures and resources created by natural disturbances in indigenous riparian forests. Variable retention strategies spanning the continuum from largely even-aged systems (including shelterwood or seed tree methods) to small patch cuts and light thinning or single tree selection may all be valuable in helping to maintain a spectrum of disturbance sizes and intensities and associated variability in vegetation structure and composition.

Where even-aged management is preferred, extended rotations will enhance protection while integrating riparian management zones into landscapes managed primarily for wood or fiber production. Heterogeneity begets diversity. Therefore, regardless of the primary harvest systems used, biodiversity-oriented riparian management should aim to preserve and create legacies, complex physical and vegetative structures, and as full an array of disturbance regimes as possible.

Riparian buffers reduce erosion and stream temperatures compared to harvesting in close proximity to streams. Effective buffer size will depend on topography and stream size. Effects will be maximized where erosion potential is high, and where removing shade has the biggest effect on stream temperatures (E. Cascades and Klamath regions). In forestlands where variable retention methods are economically viable, a gradient of increasing management intensities from streamsides to uplands may be more consistent with intermediate disturbance regime goals due to greater natural disturbance near streams (although streams below impoundments may have low levels of disturbance, and this may not apply). For example, a three-level approach grading from full retention within 30 m of the streamside to variable retention thinning or patch cuts from 30 to 100 m, and even-aged management beyond 150 m might provide equivalent protection and greater heterogeneity than a fixed 60 m full retention buffer that abuts the intensively managed matrix. Because successful regeneration of tree species like red alder and Douglas fir would be difficult in the first 30 m in this scheme, a landscape view (see Section 11) that allows for periodic regeneration of these zones is also needed.

Initial conditions strongly affect the array of strategies that might be employed to maintain local (alpha) riparian biodiversity. Stands with residual uneven-aged, multi-species forests may be best protected in riparian buffers. At the other extreme, a young, single species conifer plantation may require active thinning or patch cutting to increase growth and crown depth of potentially dominant trees, encourage horizontal and vertical heterogeneity, and favor deciduous trees or shrubs, and herbs. Most managed riparian forests probably fall between these two extremes, and may require a mixture of active or passive management approaches to develop and maintain habitat supporting the full complement of riparian biodiversity.

Given the extremely variable life history traits of the many species inhabiting riparian forests, it is probably inappropriate to assume that we can maintain habitat for all with any single prescription (Huston 1999). For example, a disturbance regime that optimizes understory diversity of vascular plants may be severely detrimental to more sensitive plant or animal species (e.g., liverworts, amphibians), unless sufficient heterogeneity in management is applied, and appropriate scales are considered. Rather, management must encompass sufficient spatial and temporal scale to ensure viability of taxa groups with potentially contrasting needs. This is consistent with maintaining an intermediate disturbance regime as described in Section 3.2. We discuss these topics in the next section on landscape scale approaches for biodiversity protection.

11.0 MULTISCALE MANAGEMENT APPROACHES FOR PROTECTING BIODIVERSITY

To preserve biodiversity, habitat must be maintained for the full complement of species through time and across space. This may be impossible from a stand-scale perspective because a) different species often have conflicting habitat needs; b) different species have different spatial requirements; and c) some species have habitat requirements that require large spatial scales (reviewed by Lindenmayer and Franklin 2002), or are too vulnerable to extinction in landscapes with even low levels of anthropogenic influences (Duffy 2003).

11.1 Modeling Management after Natural Disturbance

Adaptations to recovering from natural disturbances are common. In recent years, a number of books (see Kohm and Franklin 1997; Hunter 1999; Lindenmayer and Franklin 2002), papers (e.g., Atwill 1994), and special features in the journal *Ecological Applications* (Roberts and Gilliam 1995) have addressed the topic of biodiversity in forests managed for fiber production. They have all concluded that management should attempt to deviate less from the historic disturbance regimes in order to better maintain biodiversity. The premise is that organisms are not as well adapted to disturbance regimes that did not occur in the past and/or that substantially diminish various legacies upon which resilience apparently depends. Silviculture based on models of natural stand development following disturbance is now increasingly used where goals are both economic and ecological (Franklin et al. 2002). The general approaches complement and incorporate stand-level procedures such as retaining structures at the time of harvest, use of longer rotations, and active creation of structural complexity and habitat heterogeneity.

In addition, estimates of the historic range of variability in natural disturbances are used to provide general guidance in addressing management questions of how much disturbance and at what spatial and temporal scales. The scale dependency of historic range of variability can be a limitation, however. For example, at the scale of a small watershed, the proportion of old growth hemlock/Douglas fir may have historically ranged from 0-100%, but its range throughout the whole Douglas fir region probably varied from 30-75% (Lindenmayer and Franklin 2002). In cases where historical information is uncertain or too difficult and/or costly to obtain, managing for an intermediate disturbance regime may be a relatively low risk approach to maintaining biodiversity.

It is beyond the scope of this report to describe the natural range of variation in disturbance regimes that have operated in a region as dynamic as the Pacific Northwest. Appendix B provides a starting point, but managers must gather information on the disturbance regimes in the specific landscape with which they are concerned to use as a basis for aligning treatments. An excellent recent example of this process that can serve as a model for landowners wishing to integrate ecological and economic objectives is provided by Cissell, Swanson, and Weisberg (1999). This approach may be especially well suited for application over relatively large areas with a simple ownership pattern.

It may be easiest to align silviculture with natural disturbance regimes historically characterized by disturbances that created relatively fine-scale age patch mosaics as opposed to large-scale stand replacement or high frequency stand thinning disturbance (Franklin et al. 2002). Harvesting can approximate a patch mosaic in space and time, although patch size tends to be larger with group selection harvesting, and rotation interval for patch establishment tends to be shorter. For emulating large stand replacing disturbances, the challenge is that clearcutting, as traditionally practiced, is described by Franklin et al. (2002) as having little overlap except by creating a light environment suitable for shade-intolerant tree regeneration. Where large stand-replacing disturbances (fire, mass soil movements, large wind throw events) are dominant, they are typically infrequent (i.e., centuries), and they usually leave behind large quantities of wood, and other legacies (i.e., seed released from dormancy) are often present in significant amounts. These differences are also apparent in comparing naturally regenerating post-disturbance stands with plantations. Therefore, in forests where large, stand-replacing disturbances have prevailed, stand-level procedures can be used to help maintain biodiversity, but aligning harvest to natural disturbance may not be cost-effective due to rotation length and retention of wood that accompanies the natural disturbance regime. It may also not be cost-effective to emulate disturbance such as frequent surface fire (e.g., ponderosa pine forests of the eastern Cascades). These fires historically killed non-merchantable sized trees and few large trees.

11.2 Riparian Buffers

Establishing riparian buffers and restricting activities within them is widely regarded as a valuable approach for biodiversity protection, and is the current management paradigm underlying existing riparian management regulations (Section 5). The rationale for these buffers is that streamside vegetation a) prevents water temperatures from becoming too high for fish and amphibians (Beschta et al. 1987); b) provides the nutrient and energy base for streams in the form of allochthonous inputs; and c) is the source of large woody debris that has numerous roles in maintaining biodiversity. Ultimately, the effectiveness of riparian buffers for biodiversity protection depends upon careful consideration of both local site-specific and larger scale conservation goals.

There are both limitations with buffers and potential improvements to consider. As described in Section 5, appropriate widths and activities within buffers are controversial. They are often based on political compromise, rather than on ecological principles. In intensively managed watersheds with simplified stand structures and constraints on natural disturbance processes, buffers may not contain species, structural characteristics, or dynamic processes that provide the intended services. For example, it may be more desirable for woody debris from sources outside of buffers to be placed in streams rather than relying on recruitment from the buffer area. This may be especially true where large conifers are lacking. An additional concern where stream density is high is that management of the upland forest patches can be impractical or not economically viable. Depending on the condition of these patches, abandoning management may be detrimental to biodiversity. Finally, the functional effects of riparian areas extend to different distances from the stream leading to difficulties in determining appropriate buffer widths. Knutson and Naef (1997, Appendix C, http://www.wdfw.wa.gov/hab/ripfinal.pdf) suggest riparian habitat buffer widths for retaining various riparian habitat functions, and these widths vary considerably. Two important sources for this variation are in the research method used and in the ecosystem studied.

Riparian buffers are often uniquely important for terrestrial wildlife, as they often serve as travel corridors (Sections 7.4 and 7.5). Riparian areas at low- to mid-elevations appear to be used more as travel corridors than are riparian areas at high elevations (Kelsey and West 1998); however, the role of stream buffers functioning as travel corridors is not well studied. Using only riparian buffers to provide travel corridors or late-seral forest refugia may not be effective to maintain biodiversity for species that are primarily found in uplands, and perhaps others that use, but do not rely entirely on upslope forests. These tradeoffs need to be considered where upland harvest regimes are designed to compensate for lack of harvest in buffers.

Buffers may be most beneficial for stream amphibians because they are physiologically sensitive, both in streams and in the terrestrial environment near streams. Maximum shading capacity may be reached within a width of 25 m, and 90% of that capacity occurs at 17 m (see Budd et al. 1987; Beschta et al. 1987). However, widths of 30 m or more may be needed if the goal is to stabilize microclimates within some streamside riparian zones (Brosofske et al. 1997). Some experts believe that the minimum buffer width may need to be 60-80 m wide (Ledwith 1996a, 1996b; Welsh et al. 1998) or up to 100 m (McComb, McGarigal, and Anthony 1993) to minimize sediment flow, maintain other riparian functions, and protect the most sensitive organisms.

Moreover, as reported in Section 6.2, Welsh and Lind (1996) describe a problem with the misclassification of streams where the faulty assumption is made that aquatic life does not exist in particular channels and this results in inadequate protection for headwater streams. For example, there are concerns that riparian buffer rules for non-fish bearing streams may not be adequate to maintain required microclimatic conditions for the torrent salamander. As mentioned in Section 6.2, forest managers' collective goals may be better served by reducing reliance on a set of physical criteria such as buffer widths at 30 m and by placing more attention on the desired outcomes for streams: a) cool waters (temperature <15 C year round); b) little or no sedimentation; and c) input of large woody debris over the long haul. These issues point to the need to evaluate entire watersheds with regard to present conditions and natural range of variability when considering management that integrates both economic and ecological objectives.

11.3 Reserve-Based Management

Our concept of a reserve is simply a place where the primary goal is biodiversity conservation. Depending on the condition of a reserve, management activities may vary.

It is well known that biodiversity is nonrandomly distributed. Certain sites teem with life, whereas others are relatively species poor. The Nature Conservancy (TNC) and the closely related state Natural Heritage Programs have pioneered an approach to identify sites with exceptional biodiversity on private lands. The approach involves collection of site data and scoring of sites based on their global and state-level biodiversity values. As part of Ecoregional Planning, TNC develops a portfolio of sites that might be managed primarily for conservation via removal from commercial activities. The strength of this approach is that it recognizes the importance of certain sites and includes a mechanism to fund landowners for the financial values of private lands removed from commodity production. Society may be increasingly willing to provide incentives and compensation (http://www.wa.gov/dnr/htdocs/adm/comm/nr02-92.htm). Such an approach might have merit if a trusted nonprofit or government organization could be supported by multiple private forest owners. The conservation value of riparian forests needs to be assessed in this regard on a regional scale. This approach may be best for protecting localized populations of especially rare or sensitive taxa, or exceptionally valuable habitat (i.e., biodiversity “hot spots”, regionally rare habitats such as remnant multi-aged forests). It would be least effective for species with broader home ranges or wider distributions.

Effective biodiversity conservation via reserve networks is a broad planning issue. It needs to be evaluated within a larger landscape program of managed lands that also support some of the values of the reserves, of connectivity and proximity of reserves, of meta-population dynamics, and other issues are beyond the scope of the present analysis.

PART IV PAST, PRESENT, AND FUTURE RESEARCH

12.0 EXISTING LITERATURE AND ONGOING RESEARCH

We searched two databases to assess published and ongoing research as well as taxonomic and geographic patterns of emphasis. For published research, we used the University of Washington/Rocky Mountain Research Station Riparian Bibliography at http://riaprian.cfr.washington.edu/, a comprehensive database containing ~12,000 citations, as of March 2003. For ongoing research, we obtained the Forest Research Database, which contains information pertinent to issues in the Pacific Northwest and northern California. This database was created under a contract to the U.S. Environmental Protection Agency in Corvallis, Oregon for the interagency Regional Ecosystem Office (REO) in Portland, Oregon. This database contains descriptions of research projects supporting ecosystem management on both forested and non-forested lands in the region. The information in the database was acquired primarily from responses to a voluntary survey sent to researchers or research institutions in Washington, Oregon, and northern California.

Results of the database searches were not consistent in many respects (Figures 12.1 and 12.2). One reason for this is that the Forest Research Database is not focused solely on riparian research. It was found to contain 120 ongoing projects that involved riparian areas, however, and a number of the projects are oriented toward assessments of ecosystem processes. Many directly deal with components of biodiversity as illustrated in Figure 12.1.

There is clearly an emphasis on endangered species and the related topics of anadromous fish and wildlife in ongoing research (Figure 12.1). The number of studies concerning plants seems surprisingly low, especially considering the growing problem of non-native plants displacing native species and crop species. Considering the ecological importance and diversity of terrestrial invertebrates (Sections 6.3 and 7.5), they clearly lack sufficient attention. Likewise, no studies specifically focusing on non-vascular plants and lichens were found. Although this does not mean that none exist, there is a dearth of attention in terms of actual research devoted to these organisms as further evidenced by published literature. There were no published studies on non-vascular plants, and only two on lichens found in the Riparian Bibliography. The emphasis on endangered species is also not apparent in published literature compared to ongoing research, suggesting this is a relatively recent development. Most published research on elements of biodiversity in riparian areas focuses on plants and wildlife. There is also a considerable amount published on aquatic macroinvertebrates in riparian areas (Figure 12.2).

Beyond what we can say from these data from keyword searches is that another gap in published and ongoing research concerns the difficult issues of landscape-scale processes and planning (see Section 14.0 on future research needs).

Geographic emphasis of ongoing biological research in the Pacific Northwest is shown in Figure 12.3. The Oregon Coast range is receiving more attention than other regions. About twice as many research projects are located in the western Cascades of Washington and Oregon compared to the east side of the range in these states.

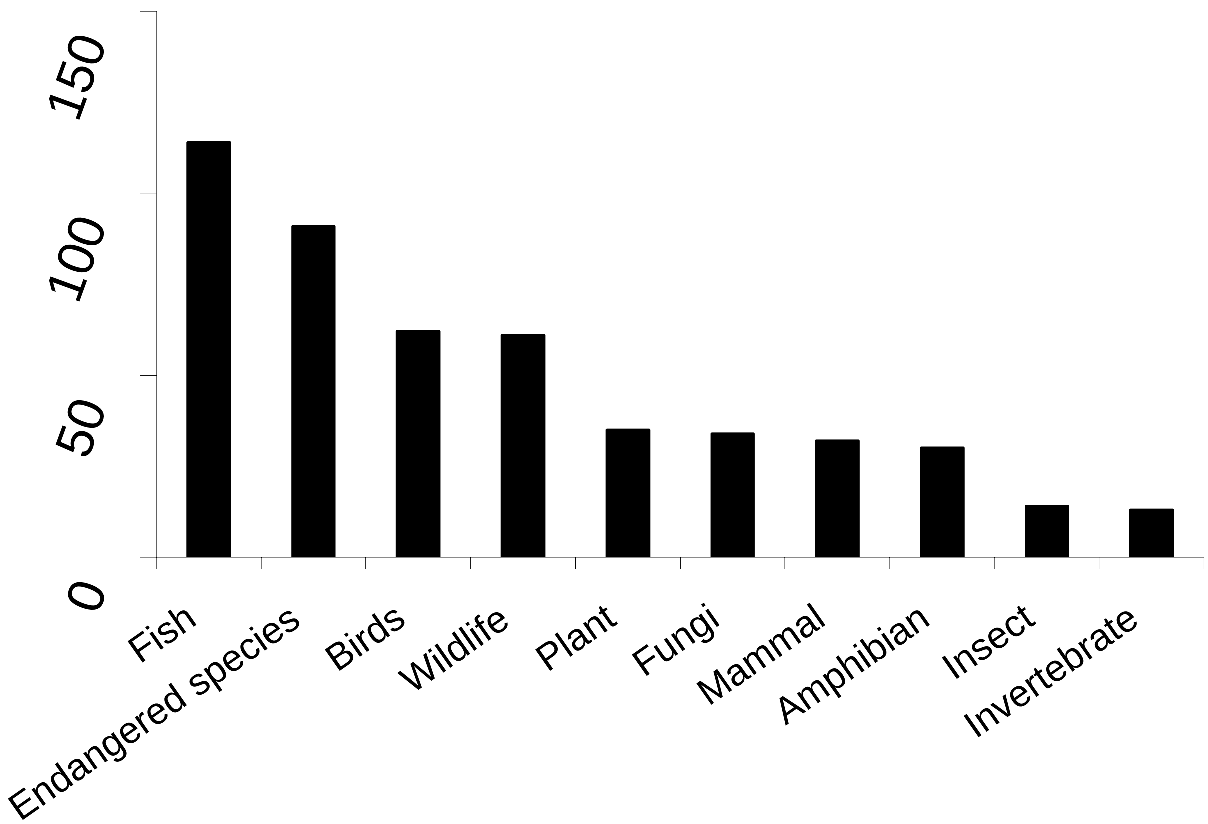

Figure 12.1 Number of Ongoing Studies by Taxonomic Group Identified in the Forest Research Database

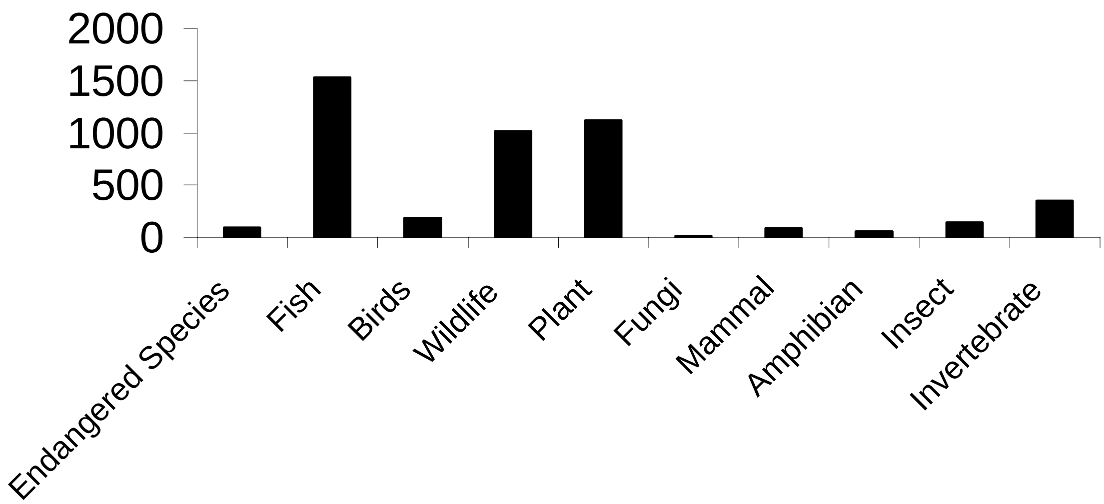

Figure 12.2 Number of Ongoing Studies by Taxonomic Group Identified in the University of Washington/Rocky Mountain Research Station Bibliography

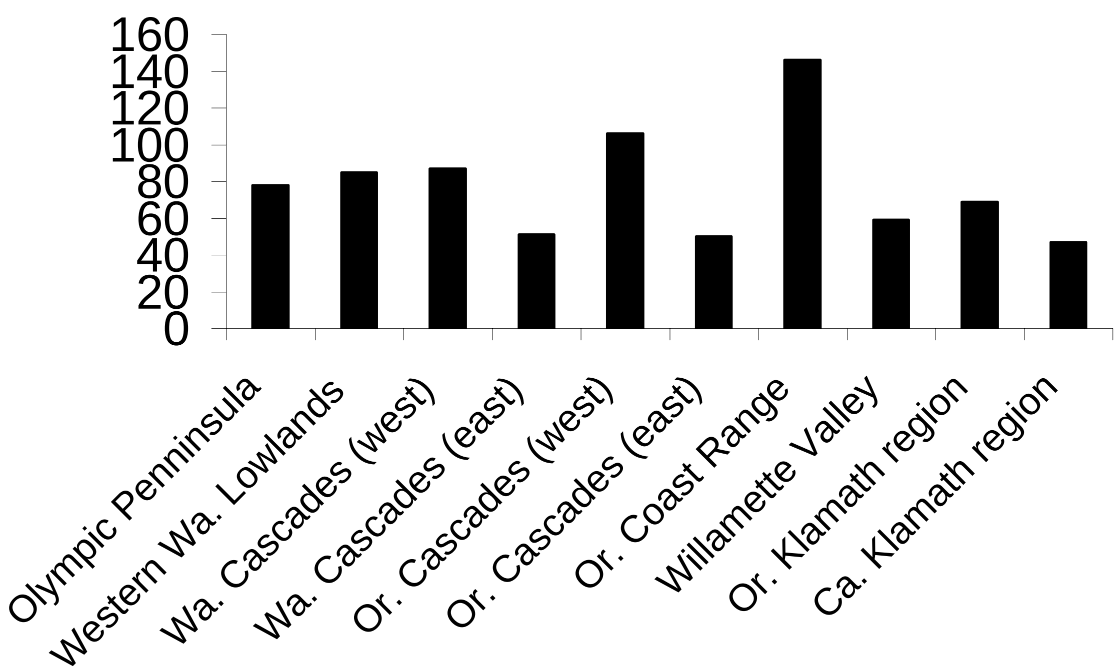

Figure 12.3 Number of Ongoing Studies by Geographic Area Identified in the Forest Research Database

13.0 A MEANS FOR ASSESSING RIPARIAN LITERATURE

Few issues draw sharper reactions from the public and scientific community than timber harvest in the Pacific Northwest. Tensions peak when discussing the management of riparian forests, which contain some of the most productive, biologically rich forests in the region. The perspective that riparian forests require complete protection contrasts with a growing appreciation that riparian forests are structured by disturbance. Although much has been written about riparian forests in the Pacific Northwest, easy answers about forest management effects on biodiversity are elusive. Those interested in determining the effects of forest management on biodiversity must weigh the published literature carefully.

Engeldinger (1991) provided a number of questions for developing annotations and assessing the validity of information sources that are especially useful for considering literature in such contentious fields:

1. Who is the author?
2. What is the author's purpose?
3. Who is the intended audience?
4. What is the author's likely bias, if any?
5. What is the primary information source for the article?
6. What are the primary conclusions?
7. Are the conclusions justified?
8. What is the article's relationship to other sources?

Additional questions might include:

9. Has the article been peer-reviewed?
10. Is the study design robust for addressing the stated objectives?

These and similar questions in critical thinking are dealt with in detail by a number of authors in Shirato (1991). These general principles are highly relevant to an objective assessment of biodiversity protection in riparian management zones. From our review of existing literature and many discussions on the topic, we note several sources of bias or uncertainty in the scientific literature that bear particular consideration.

Political bias. As residents of the region, scientists frequently have strong feelings about land management policies. These feelings run deep with regard to regional rivers and riparian environments. Although most scientists try to maintain neutrality in scientific discussions, advocacy is not unknown (Rykiel 2001). Scientists are increasingly asked to participate in policymaking decisions, especially where important ecological and social values are at stake (Lach et al. 2003). When such decisions impinge upon complex topics such as riparian biodiversity, distinctions between scientists' political perspectives and ecological judgment can become obscured. Some authors assume a protective stance with regard to riparian forests, and argue that any forest management will have a negative effect on biodiversity. Yet, full protection is likely to result in unintended consequences to biological diversity, such as loss of hardwoods over time. Other scientists may emphasize a minimalist perspective on riparian forest protection. Readers should take care to discern when policy perspectives or value judgments become intermixed with scientific discussion.

Taxonomic bias. Disciplinary perspectives strongly infuse the riparian management literature. Fisheries ecology is a particularly strong influence, as are plant ecology, wildlife ecology, and forestry. As noted above, less charismatic or economically valuable taxonomic groups (e.g., soil arthropods) have received much less study, yet these species undoubtedly comprise the majority of species occurring in riparian zones. The tendency to generalize findings from a single group to "biodiversity" as a whole is frequently made in project justification statements and in conclusions of primary papers. As noted above, such cross-taxa inferences are probably not justified in many cases. As we have argued throughout this report, inferences about forest management effects on biodiversity depend on a) the organism's life history; b) the geographic setting; c) previous disturbance regimes; and d) the management intensity. Much more basic research is needed to determine the specific responses of lesser known taxonomic groups and to determine if their responses can be estimated by surrogate species or other metrics.

Geographic bias. As noted above, our knowledge of rare species, riparian dynamics, and forest management research in general, is disproportionately weighted towards sites close to urban centers and research universities (i.e., western Washington, northwest Oregon). Areas east of the Cascades and in the Klamath Region have received less intensive study. More generally, it is probably important to closely consider where the study occurred and to consider whether conclusions can be extrapolated to other locations with sharply contrasting climates, hydrology, or other features.

Generalizations about disturbance effects. The disturbance ecology literature frequently includes clear value judgments that add confusion. In particular, natural disturbance regimes are frequently cited as essential for biodiversity, whereas anthropogenic disturbances are frequently described as ecologically harmful. The fire ecology literature is rife with value judgments such as "cleansing" vs. "catastrophic" wildfire. Until a more quantitative basis for evaluating disturbance mechanisms and taxonomic responses is developed and adhered to, discussions of disturbance ecology will remain imprecise. As we have described, quantifying functional heterogeneity and legacy retention (White and Jentsch 2001) will help explain general effects of disturbance.

14.0 A RESEARCH AGENDA TO SUPPORT RIPARIAN BIODIVERSITY PROTECTION

In this section, we present rationale and recommend approaches for further research in support of biodiversity protection in riparian forests where silviculture is practiced. The recommended research agenda is organized into several programs outlined below, and a number of more taxa-specific research needs. Research needed for specific taxa groups is provided in Sections 6 and 7.

14.1 Programmatic Recommendations

14.1.1 *Program One: Evaluation of Potential and Actual Diversity across Geographic Gradients*

Rationale. There is tremendous spatial variation in the diversity and abundance of native riparian species. We need much more complete information about the factors controlling potential diversity to both locate biodiversity "hot spots" and to assess how current levels of diversity compare with expected values. A research program is needed to more quantitatively define how biodiversity is being affected by forestry practices and to help identify sites of exceptional conservation interest.

Approach. Spatially explicit models could be used to determine distribution and diversity of focal riparian taxa from existing field inventories, augmented with targeted field inventories of poorly understood regions. The survey and manage program on federal lands has begun such an approach. A parallel effort on private lands may also yield valuable insights. Once a response surface of potential diversity had been established for focal groups (e.g., fungi, vascular, nonvascular plants, arthropods), additional field measurements could evaluate sites that are below their biodiversity potential, or that have regional significance. These insights would add a quantitative basis to statements about the status of biodiversity on private forestlands. Federal inventory and monitoring programs are working to develop this regional understanding of biodiversity to target monitoring efforts (Figure 14.1). Landscape and historical factors strongly govern the potential diversity of a given locale. Once a reliable quantitative estimate of the diversity of a site is obtained, it may be possible to evaluate the deviation between potential an actual diversity of a site.

14.1.2 *Program Two: Private Forestlands of the Pacific Northwest*

Rationale. Natural disturbance regimes in riparian forests are caused by a complex interplay of fluvial and hillslope disturbance processes. These varied disturbances are known to be important for maintaining the diversity of many types of organisms in riparian forests. Studies of the temporal dynamics of riparian forests have typically taken a single factor approach to the study of disturbance (i.e., fire, fluvial disturbance, gap dynamics). In reality, large landscape-scale disturbances and more localized or low intensity disturbances are both important in terms of their effects on dominance and maintaining habitat heterogeneity in riparian forests (Sakai et al. 1999). When applied uniformly, even-aged or uneven-aged silvicultural systems can both lead to simplified stand structures when contrasted with the patterns caused by stochastic disturbance events. A better understanding of the size, periodicity, and intensity of riparian forest disturbances is needed to inform riparian forest management.

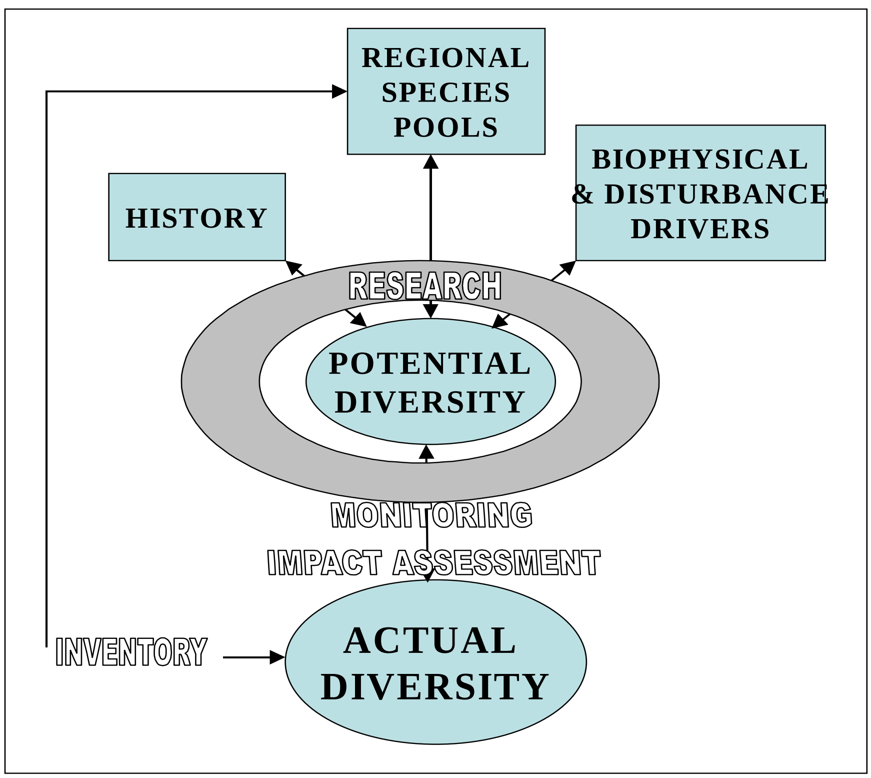

Figure 14.1 Conceptual Model of Controls on Biodiversity, and Roles of Inventory Monitoring, and Research

Approach. Retrospective analyses of radial growth patterns on cut stumps or live trees, hydrologic or climatic analyses, and comparative field studies in remaining reference riparian forest stands and watersheds could be employed to study the complex disturbance regimes and associated stand structures of natural riparian forests. A network of permanent plots added to existing riparian monitoring programs might also provide information on the characteristics of both natural and human-caused disturbances and biological response in these systems.

14.1.3 *Program Three: Stand-Scale Analyses of Physiological Stress, Resource Availability, and Species Response across Management Intensity and Macroclimate Gradients*

Rationale. Different life forms respond individualistically to gradients in habitat structure, resource availability, or physiological stress associated with forest management. Quantitative data are needed to determine important stress thresholds for different life form groups across gradients in management intensity. The magnitude of such management-associated changes also varies geographically due to changes in climate and riparian forest structure, such that a management technique (e.g., thinning) that might be relatively innocuous for an element of biodiversity in a mild, coastal climate may be considerably more detrimental in a more severe, interior environment. For example, the response of stream temperature to timber harvest is likely to be considerably greater in the eastern Cascades or Siskiyous than in the Coast Range or Olympic Mountains. A stronger quantitative basis for evaluating physiological stress responses of taxa and associated geographic variation would be a great source of information for private land managers.

Approach. A manipulative study employing a gradient of treatment intensities (e.g., no harvest, light thin, heavy thin, shelterwood, patch cut) replicated in riparian forests across a climate gradient (e.g., Coast Range, Western Cascades, Klamath Mountains, Eastern Cascades) and tracking selected abiotic parameters and elements of riparian biodiversity before and after treatment would provide a rigorous evaluation of such intensity/site interactions.

14.1.4. *Program Four: Landscape Integration-Evaluation/Comparison of Conservation Paradigms for Protection of Riparian Biodiversity*

Rationale. Although most current regulations target management activities at the stand scale, scientists and land use planners may be able to join forces to develop strategies for planning and integrating stand-scale activities (individual land owner activities) across larger, often multiple ownership landscapes, to evaluate whether the elements critical to landscape-scale biodiversity conservation are also met. There is an increasing realization that conditions of the larger landscape may be as important as local habitat in determining the biodiversity of a site. Multiscale analyses are needed to determine the effective scale of controls on diversity (see Saab 1999). A number of current conservation paradigms in the Pacific Northwest and elsewhere directly influence forest regulations in riparian zones on private lands. As a starting point, we propose three paradigms that warrant comparative study at the landscape scale: natural disturbance regime-based management, riparian buffer systems, and the selection of no harvest preserves. These paradigms are outlined briefly below.

1. <u>Natural disturbance-based management</u> This paradigm proposes to base harvest patterns on an assessment of natural return intervals derived from the landscape disturbance regime. This approach incorporates disturbance as a management tool and attempts to recreate natural spatial patterns of forest age and structure. All areas of the landscape are open to management, but only within the context of natural disturbance intervals and intensities specific to that landscape position.

2. <u>Riparian buffers</u> This paradigm assumes that exclusion or curtailment of management activities within riparian areas is the most prudent means to protect biodiversity in the landscape as a whole. The paradigm has its origins in intensively managed agricultural landscapes, or in landscapes with important coldwater fisheries, where protection of streams is considered paramount.

3. Selection of conservation reserves This paradigm, widely used by the Natural Heritage Program and The Nature Conservancy, assumes that sites with a high concentration of regionally rare or "imperiled" taxa warrant special protection regardless of ownership or landscape position. Protection is advocated through outright land purchase or through conservation easements that preclude future development or extractive management. The reserve system (TNC's conservation portfolio) so created is intended to conserve regionally and globally significant taxa whether they are riparian-dependent or not.

Approach. These conservation paradigms each involve a number of untested assumptions about the interactions between harvest activities, landscape structure, and species viability. All are well intentioned and actively used in forming riparian management policy or in conservation planning. An explicit comparison of the outcomes of these models in private forest landscapes developed through the use of forest growth and wildlife habitat models would allow us to better assess the strengths and weaknesses of these approaches for maintaining biodiversity in managed landscapes.

14.1.5 *Program Five: The Role of Biological Legacies*

Rationale. In even-aged, short-rotation forestry, elements of older forests (e.g., large trees, large logs, snags, lichen and fungi populations, seed banks, bud banks) do not have time to develop. The concept of biological legacies, or carryovers from previous stands, has gained increasing attention in recent years (see Lindenmayer and Franklin 2002), yet quantitative data are still needed to guide management and restoration.

Approach. Since retention of certain elements (standing green trees, logs) is costly, it may be valuable to quantify the potential relationships between legacy abundance and specific biodiversity elements. For example, the importance of residual green tree densities for post-harvest lichen diversity may show linear, unimodal, convex, or concave response functions (Figure 14.2). If the linear model applies, more is better. If the convex models applies, much of the positive benefit may occur at lower retention levels. If the concave model applies, high retention levels will be required to maintain lichen diversity. The unimodal model suggests that there is an optimal level of retention for lichen diversity. Understanding these relationships may allow prediction of what different levels of retention do for the maintenance of biodiversity. Direct placement of legacy elements could be a more manipulative approach to evaluating the role of biological legacies (e.g., large wood) for biodiversity (see Hayes and Waldien 2001) that merits further study.

14.1.6 *Program Six: Restoration of Riparian Biodiversity*

Rationale. Riparian restoration is an active field in the Pacific Northwest, with many projects focusing on salmon and their habitats. A complementary approach might be to explicitly define a number of biodiversity elements (structural heterogeneity, lichen; fungi, small mammal, amphibian, bird abundance and diversity) and attempt to develop trajectories of recovery under alternative active and passive management approaches. Trajectories of recovery are poorly understood for most degraded systems and explicit documentation of recovery is badly needed in impacted landscapes.

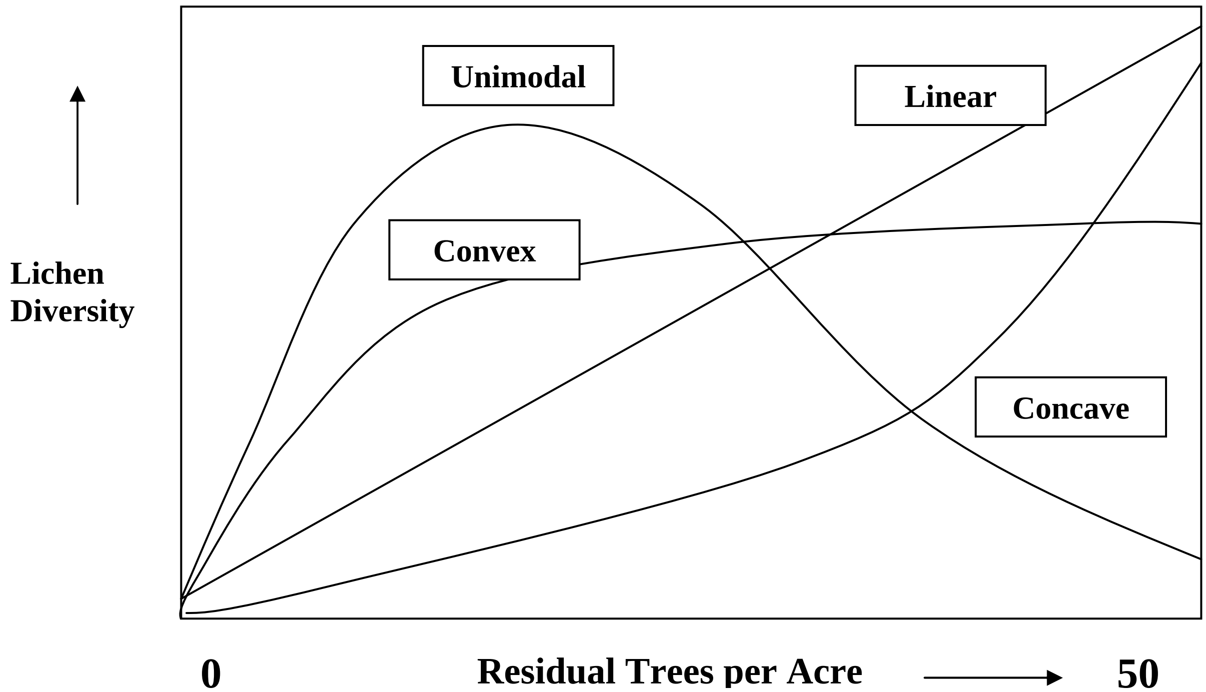

Figure 14.2 Three Potential Response Curves for Lichen Diversity as a Function of Green Tree Retention

The notion of hysteresis, or asymmetry in trajectories of degradation and recovery, is very important in restoration ecology and may provide insights into where to allocate limited restoration funds (Sarr 2002). Systems can have elastic responses to stress, where they recovery rapidly and predictably, or plastic response, in which recovery may either be exceedingly slow or impossible (Figure 14.3, Sarr 2002). In a system with an elastic response to stress, the system will probably recover rapidly once the stress is removed. For example, arboreal invertebrate communities would be strongly impacted by a streamside clearcut, but they would probably naturally converge toward the species composition of undisturbed forests with time. Similarly, aquatic invertebrate communities tend to recover rapidly from short pulses of sediment or pollutants. However, in a system with a plastic response, active management may be warranted to ensure or accelerate the recovery of desirable characteristics. This might include aquatic invertebrate or fish communities in a stream that has been eroded to bedrock after splash damming and/or beaver removal. Recovery of the populations will likely first require recovery of habitat structure and function, or at least an approximation of it.

Better understanding of essential species or ecosystem elements (“keystones”) and recovery responses of target taxa would provide a better ability to forecast where active or passive restoration approaches are needed. Finally, documentation of species’ responses to complete cycles of degradation and recovery (e.g., fire, debris flow cycles) may be essential to fully understand the life history requirements of certain species.

Approach. Once restoration goals are clearly defined, modeling or long-term manipulative field studies could be used to determine recovery trajectories for large wood (Beechie et al. 2000) or other target elements of biodiversity. The manipulative studies could use either silvicultural or other methods to augment recovery. For example, Hibbs and Chan (1997) noted that overstory thinning and understory shrub cutting both increased survival and growth of understory conifers. Placement of large wood on the riparian forest floor or creating artificial cavities in trees could also be evaluated as a means of rebuilding depressed wildlife or populations in riparian forests. Manipulative restoration experiments that add critical elements (beavers, large wood, mycorrhizal species) or remove non-native species (cowbirds, Himalayan blackberry, giant reed) could provide the means to unambiguously identify keystone species or processes that have an exception effect on the recovery of riparian biodiversity.

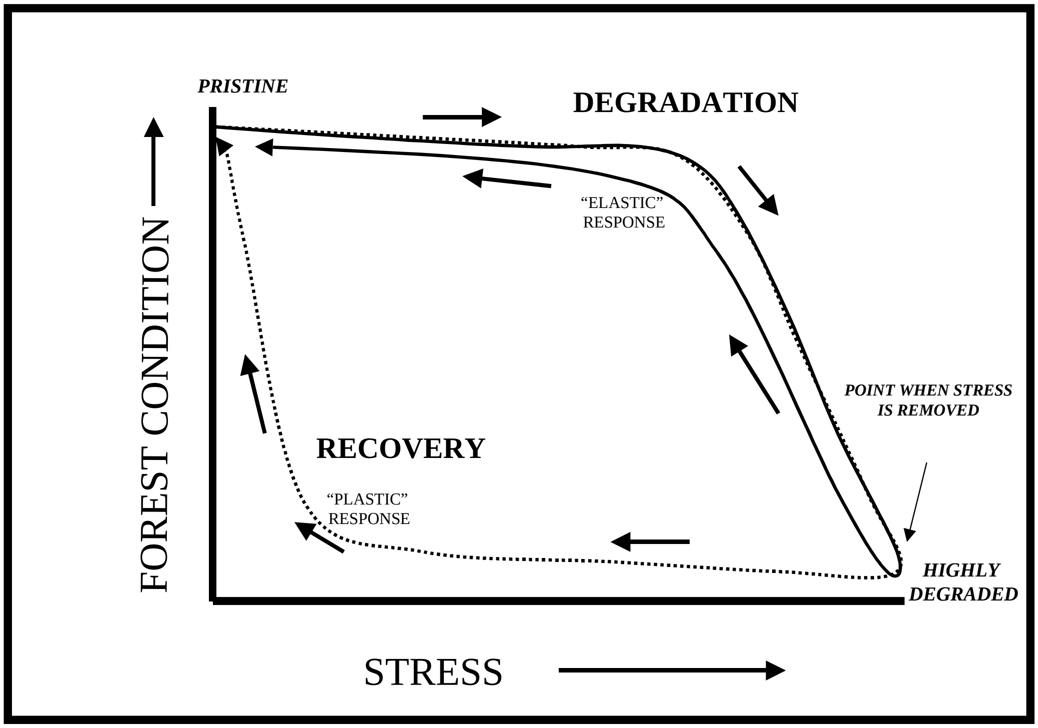

Figure 14.3 Conceptual Models of Physiological Stress and Recovery in a Riparian System (based on Sarr 2002)

In addition, the permanent plot monitoring system mentioned above would provide important insights into natural patterns of degradation and recovery that maintain biodiversity and could guide ecological restoration.

14.1.7 *Additional Taxa Specific Recommendations*

Additional taxa-specific recommendations can be found at the conclusions of each taxon subsection in Sections 6 and 7. A common information need for nearly all taxa groups is a better understanding of the individual species groups away from intensively studied locales, and more effort to determine the interactions between forest management and species diversity in a variety of settings. Program 3 above may provide a means to address some of these concerns. In addition to the need for more basic life history information, the subsection authors also highlighted the need to better determine the severity and duration of management effects, and the need to frame questions at the landscape scale to determine the factors maintaining the viability of species populations. Program 4 above may partially address these needs.

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APPENDIX A

FOREST ZONES IN THE PACIFIC NORTHWEST

Table A1 describes general riparian and associated upland vegetation in forestlands and how it varies across the study area. Included is the Willamette Valley, much of which is not forested, but which supports plantations in and next to riparian areas. Descriptions are based on Franklin and Dyrness (1973), with supplemental information for California based on Barbour and Major (1977). Herbaceous understory species and non-vascular plants are too numerous to detail here, but are described in the above treatments.

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Table A1 Summary of Riparian and Associated Upslope Vegetation (Excluding Lichens) in Forested Regions of the Pacific Northwest Where Private Lands Are Predominantly Located

Regional Vegetation Zone	Distribution	Dominant Riparian Trees*	Riparian Understory**	Upland Vegetation***
1. **West of Cascade Crest, Excluding SW Oregon and NW California**				
Sitka Spruce (*Picea sitchensis*) zone	Wet coastal strip from Alaska to N. Calif.	*Picea sitchensis, Tsuga heterophylla, Thuja plicata Pseudotsuga menziesii*[1], *Alnus rubra*[1], *Acer macrophyllum*[2], *Populus trichocarpa*[1,2]	Often well-developed, with woody species: *Rubus spectabilis*[1], *Vaccinium* spp., *Acer circinatum*, ferns, *Polystichum munitum*, and many forbs. Exceptional moss and liverwort abundance and diversity, many as epiphytes	Tree and shrub strata similar to riparian areas, but with much less hardwood (*Alnus*, *Acer*) cover.
Western Hemlock (*Tsuga heterophylla*) zone, **coast ranges**	Extensive at relatively low elevations (0-1200m) in Western Cascades and Coast Ranges of Washington, and Oregon.	*Alnus rubra*[1] , *Acer macrophyllum Fraxinus latifolia*[2],*Pseudotsuga menziesii*[1], *Tsuga heterophylla, Thuja plicata. Chamaecyparis lawsoniana*[3] , *Populus trichocarpa*[1,2]	Well-developed under hardwood overstory, with woody species (*Rubus spectabilis*[1,4], *Acer circinatum, Taxus brevifolia*, and ferns (*Polystichum munitum).* Many forbs. Exceptional moss and liver-wort abundance and diversity.	Tree and shrub strata similar to riparian areas. Less hardwood and understory cover, but *Alnus rubra* often follows stand-replacing fire or logging.
Western Hemlock (*Tsuga heterophylla*) zone, **Cascades**	Extensive at relatively low elevations (<1200 m) in western Cascades north of Rogue River.	*Psuedotsuga menziesii*[1], *Tsuga heterophylla, Thuja plicata, Alnus rubra,*[1,2] *Acer macrophyllum, Populus trichocarpa*[2], *Fraxinus latifolia*[2]	With conifer dominance, not as well-developed as in coast ranges. Woody species (*Acer circinatum, Taxus brevifolia*, ferns (*Polystichum munitum*), and many forbs present. Abundant mosses and liverworts.	Less hardwood and understory cover, *Alnus rubra* absent.
Subalpine (*Abies amabilis*) zone	East side of Olympic Mts. and west slopes Of Cascades south to Central Oregon. At 600-1500 m. depending on latitude.	*Abies amabilis, Tsuga heterophylla, Picea engelmanii, Psuedosuga menzeisii*[1], *Populus trichocarpa*[2]	Scattered woody species*: Oplopanax horridum, Vaccinium* spp. Herb-rich understory (e.g., *Tiarella, Smilacina, Clintonia*, and ferns). Mosses and liverworts common.	Tree and shrub strata similar to riparian areas. Less hardwood and understory cover.
Willamette Valley	Western Oregon	*Populus trichocarpa*[1,2], *Fraxinus latifolia, Salix spp.*[1,2,] *Acer macrophyllum, Alnus rubra*[1,2,] *Quercus garryana, Pinus ponderosa*[1], *Abies grandis*	Varies from sparse to lush. *Salix spp.*[2], *Carex spp*. Mosses and liverworts common.	Mainly grasslands, and oak (*Quercus garryana*) woodlands.

(Continued on next page. See notes at end of table.)

Table A1 Continued

Regional Vegetation Zone	Distribution	Dominant Riparian Trees*	Riparian Understory**	Upland Vegetation***
2. SW Oregon and NW California				
Redwood (*Sequoia sempervirens*)	Near coast in Northern California and Extreme Southwest Oregon.	*Sequoia sempervirens*[1], *Tsuga heterophylla, Umbellularia californica, Acer macrophyllum, Populus trichocarpa*[2], *Alnus spp.*[1,2]	Often not well developed. *Polystichum munitum*, *Oxalis oregana*, *Woodwardia fimbriata* and other ferns and perennial forbs. Considerable moss and liverwort abundance.	Similar to riparian, but lower tree layer dominated by *Lithocarpus densiflorus* and *Arbutus menzesii*[5].
Mixed evergreen *Pseudotsuga*/ Sclerophyll zone	700-1500 m in Klamath Mountain Ranges. Widespread away from coast.	*Alnus spp.* [1,2] , *Fraxinus latifolia*[2], *Chamaecyparis lawsoniana*[6], *Populus trichocarpa*[2], *Acer macrophyllum*[2],*Pseudotsuga menzeisii*[1]	Relatively well developed and rich in woody species (*Acer circinatum*, *Rubus* spp., *Corylus cornuta*, *Rosa gymnocarpa*, *Physocarpus capitatus*) and ferns (*Polystichum munitum*). Scattered herbs, mosses, and liverworts.	Much Pseudotsuga *douglasii*, and *Abies concolor*, with lower tree layer dominated by *Lithocarpus densiflorus* and *Arbutus menzesii*[5].
Mixed Conifer (*Psuedotsuga menzesii*) zone. (Upper elevation limit can be dist-inguished as the *Abies concolor* (zone).	700-2000 m, mainly in southern Cascades, but also found in the eastern Klamath Mountain Ranges.	*Alnus spp.*[1,2] , *Fraxinus latifolia*[2], *Populus trichocarpa*[2], *Acer macrophyllum*[2], *Abies concolor*, *Taxus brevifolia, Populus tremuloides, Tsuga heterophylla*[7],*Pseudotsuga menzesii*[1], *Thuja plicata*[7]	Well developed and relatively rich in woody species[8] (*Acer circinatum*, *Rubus* spp., *Corylus cornuta*, *Rosa gymnocarpa*, *Physocarpus capitatus) and ferns* (*Polystichum munitum*). Scattered herbs, mosses, and liverworts.	Much *Pseudotsuga menziesii*, *Abies Concolor*, with scattered *Pinus* spp[9]. and *Calocedrus decurrens*. Sub canopy of *Arbutus menzesi and Castanopsis chrysophylla*[5]. Fairly sparse understory.
3. Eastern Cascades				
Grand/white fir (*Abies grandis*/ *Concolor*) zone	1,000-2,000 m. Extensive in Oregon, less common northward.	*Abies grandis/concolor, Populus tremuloides, Pinus contorta.*	Scattered woody species (*Salix spp., Vaccinium spp., Ribes spp. and Symphoricarpos albus.* Non forest, mountain meadow vegetation of grasses and sedges common. Scattered herbs, mosses, and liverworts.	Wide variety of conifers Scattered with Abies, Most common are *Pinus Ponderosa, Pinus contorta, Larix occidentalis* and *Pseudotsuga menziesii.*

(Continued on next page. See notes at end of table.)

Table A1 Continued

Regional Vegetation Zone	Distribution	Dominant Riparian Trees*	Riparian Understory**	Upland Vegetation***
3. Eastern Cascades (cont'd)				
Western hemlock (*Tsuga heterophylla*) zone	Mainly in Washington and British Columbia at 800-1200m. Uncommon southward.	As described for Western Hemlock zone above, but *Thuja plicata* more common, and *Populus tremuloides* present.	As described for Western Hemlock zone above.	As described for Western Hemlock zone above, but *Thuja plicata and Pinus monticola* more common.
Eastside Pine (*Pinus ponderosa*)	Entire east Cascades in a 15 30 km wide band, 600-1200m elevation in the nort and 900-1500 (2000)m in the south.	*Pinus ponderosa*[1], *Populus tremuloides*, *Pseudotsuga menziesii*[10], *Abies grandis/ concolor*[10].	Well developed and relatively rich in woody species[8]. *Physocarpus malvaceus*, *Symphoricarpos albus*, *Holodiscus discolor*, *Ceanothus sanguineus*, *Ribes* spp. *Purshia tridentata*, perennial grasses, sedge (*Carex*) rushes (*Juncus*) and forbs.	Open Pinus ponderosa forest/ woodland with advanced *Abies* and *Pseudotsuga* regeneration. Understory of more drought adapted shrubs (*Artemisia* and bunch grasses (*Stipa*, *Agropyron*)

* In general order of dominance.
** Hydrophytic species such as willows (*Salix* spp.), sedges (*Carex* spp.), ferns (*Adiantum*, *Blechnum*), mosses and liverworts occur in all zones.
*** Compared to riparian areas within each zone. Hydrophytic vascular plant species described above (single asterisk) generally absent from upslope areas throughout the region.
[1] Shade intolerant, and consequently, regeneration into canopy occurs with opening of forest canopy through gap, fire, fluvial, etc. disturbance.
[2] Primarily or entirely found in riparian areas within the regional vegetation type.
[3] Not found north of Coos Bay area, coast range only (Zobel, Roth, and Hawk 1985).
[4] Mainly in coast range.
[5] Evergreen hardwoods.
[6] Sole tree dominant, or nearly so in riparian areas in serpentine region of SW Oregon, NW California (Hansen et al. 2000).
[7] Umpqua watershed and northward.
[8] Except in *Abies concolor* dominated areas at upper elevational limit of zone.
[9] Both ponderosa pine (*Pinus ponderosa*) and sugar pine (*Pinus lambertiana*), which were previously more common. They still distinguish these forests, however, along with occasional *P. monticola*.
[10] Reproduce here in the absence of fire, logging or other stand replacing disturbance.

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APPENDIX B

DISTURBANCE REGIMES IN RIPARIAN AREAS OF THE PACIFIC NORTHWEST

Table B1 Summary of Stream Characteristics and Regimes of Large Disturbance Processes (Fluvial/Geophysical and Fire) in Riparian Areas within Franklin and Dyrness Vegetation Zones and Other Areas Identified and Described in Table A1

Regional Vegetation Zone	Annual precipitation	Stream Characteristics	Fluvial/Geophysical and Other Disturbances	Historic Fire Regime* and Upland Disturbances
1. West of Cascade Crest, Excluding SW Oregon and NW California				
Sitka Spruce (*Picea sitchensis*) zone	200-300 cm. Mostly or entirely as rainfall.	Ranging from small streams in steep drainages to fairly large low gradient rivers.	Flooding/sediment deposition dominant process along rivers. Infrequent debris flows in small streams[1].	Stand-replacing fires at a frequency ranging from 100-500+ years. Riparian areas often missed (Quaye 1982). No fire in some areas of Vancouver Island and Alaska Panhandle. Wind the dominant disturbance factor (Agee 1993).
Western Hemlock (*Tsuga heterophylla*) zone, **coast ranges**.	150-300 cm. Predominantly rainfall.	Ranging from small streams in steep drainages to low gradient rivers. Many headwater/ higher-order streams.	Flooding/sediment deposition dominant process along rivers. Infrequent debris flows[1] important in lower order streams.	Stand-replacing fires at a frequency ranging from 100-500+ years. No fire in some areas of Vancouver Island and Alaska Panhandle.
Western Hemlock (*Tsuga heterophylla*) zone, **Cascades**.	150-300 cm. Mostly rainfall, some substantive snowfall most years.	Ranging from small streams in steep drainages to moderate and high gradient rivers. Many headwater/higher-order streams.	Flooding/sediment deposition dominant process along rivers, especially with rain on snow events in watershed. Infrequent debris flows[1] in lower order streams.	Fires of patchy mixed severity generally from 90-150 years.
Subalpine (*Abies amabilis*) zone	150-300 cm. A considerable amount as snowfall.	Mainly small to medium-sized high gradient small streams.	Flooding/mass transport often exacerbated by rain on snow events[2]. Infrequent debris flows[1].	Stand-replacing fires at a frequency ranging from 100-500+ years. Avalanches important, And maintain corridors or tracks of non-forest Vegetation.
Willamette Valley	~100 cm. on valley floor. Mostly or entirely as rainfall.	Relatively large, low gradient Rivers on valley floor with various sized moderate to low gradient tributaries.	Flooding/sediment deposition dominant processes.	Fire frequency highly variable depending on Native American burning practices. Predominantly low severity surface fire.
2. SW Oregon and NW California				
Redwood (*Sequoia sempervirens*)	150-300 cm. Mostly or entirely as rainfall.	Ranging from small streams in steep drainages to fairly large low gradient rivers.	Flooding/sediment deposition dominant process along rivers. Overbank deposition kills trees other than redwoods, generating pure stands on terraces	Variable past fire frequency (20-500Years) depending on proximity to coast and human ignitions. Severity usually low to moderate. Post-fire runoff produces overbank sediment deposition events

(Continued on next page. See notes at end of table.)

Table B1 Continued

Regional Vegetation Zone	Annual precipitation	Stream Characteristics	Fluvial/Geophysical and Other Disturbances	Historic Fire Regime* and Upland Disturbances
2. SW Oregon and NW California (cont'd)				
Mixed evergreen *Pseudotsuga*/ sclerophyll zone	60-170+ cm. Predominantly as rainfall.	Ranging from small, ephemeral streams in steep drainages to moderate and relatively high gradient rivers. Many headwater/higher-order streams	Flooding/sediment deposition dominant process along rivers and lower reaches of streams, especially following rain on snow events[2]. Infrequent debris flows[1] important in higher order streams.	Highly variable fire frequency, 3-90+ year Ranges reported, 90 year median in one study. Mixed fire severity with mostly low and moderate.
Mixed Conifer (*Pseudotsuga menziesii*) zone. (Upper elevation limit can be distinguished as the *Abies concolor* (zone).	90-130 cm. Much or most as snowfall.	Ranging from small streams in steep drainages to upper reaches of relatively high gradient rivers. Many headwater/higher-order streams.	Flooding/mass transport often exacerbated by rain on snow events. Infrequent debris flows[1] in small streams.	Fires of mixed severity at highly variable frequencies (3-150+ years). Frequency of stand replacing fire at a given location 150-350 years. Decreasing frequency with elevation.
3. Eastern Cascades				
Grand/white fir (*Abies grandis*/*concolor*) zone	60-120 cm. Much as snowfall.	Ranging from small streams in steep drainages to upper reaches of relatively high gradient rivers. Many headwater and lower-order streams.	Flooding/mass transport often exacerbated by rain on snow events. Infrequent debris flows[1] in small streams.	Few data. 9-100[3,4] years based on two studies. Both surface and stand-replacing fire, the latter at intervals from 140-340 years.
Western hemlock (*Tsuga heterophylla*) zone	56-170 cm. Much or most as snowfall.	Ranging from small streams in steep drainages to upper reaches of relatively high gradient rivers. Many headwater and lower-order streams	Flooding/mass transport often exacerbated by rain on snow events. Infrequent debris flows[1] in small streams.	Fires of patchy mixed severity generally from 90-150 years likely based on relatively dry west Cascade locations.

(Continued on next page. See notes at end of table.)

Table B1 Continued

Regional Vegetation Zone	Annual precipitation	Stream Characteristics	Fluvial/Geophysical and Other Disturbances	Historic Fire Regime* and Upland Disturbances
3. Eastern Cascades (cont'd)				
Eastside Pine (*Pinus ponderosa*)	35-80 cm. Much or most as snowfall.	Ranging from small streams in steep drainages to moderate and relatively high gradient rivers. Many headwater and lower-order streams.	Flooding/mass transport often exacerbated by rain on snow events. Infrequent debris flows[1] in small streams.	Few data. 3-33[3,4] years based on two studies. Much burning by Native Americans. Predominantly low severity surface fires. Patches of even aged cohorts from stand-replacing fires may occur (Daubenmire and Daubenmire 1968).

*Figures reported are for uplands; see Section 2.2 for explanation of differences in riparian areas, which have received little direct study. See Agee (1993), Frost and Sweeney (2000), and references therein for documentation. Fire frequencies are for recent centuries. Over longer time scales frequency varied, and there was no stationary fire frequency (Whitlock, Shafer, and Marlon 2003).

[1] Though infrequent (~1-2 times/millennium) extreme events, these have profound effects (see text).

[2] Important in generating exceptional streamflows, causing scouring and redistribution of organic and inorganic material and creating bare surfaces.

[3] Expected to be considerably longer at landscape scales that such studies that are based on locations of recorded trees, compositing samples, and that ignore the fire-free interval from tree establishment to the first fire scar (Baker and Ehle 2001).

[4] Fires less frequent since end of Native American ignitions, arrival of cattle, and with modern fire suppression.

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MONTANE RIPARIAN HABITAT AND WILLOW FLYCATCHERS: THREATS TO A SENSITIVE ENVIRONMENT AND SPECIES[1]

Susan D. Sanders and Mary Anne Flett[2]

Abstract: Mountain meadows provide critical habitat for California's dwindling population of Willow Flycatchers (Empidonax traillii) and for many other breeding birds. Most meadows in the western United States are managed for livestock production or other consumptive uses rather than for wildlife. The potential threats to Willow Flycatchers and their habitat are discussed, and suggestions to protect and enhance mountain meadow habitat for this and other riparian species are offered.

California's montane meadows have received relatively little attention from wildlife biologists and conservationists concerned with riparian habitat protection. For example, only two papers presented at the first California Riparian Systems Conference (Warner and Hendrix 1984) discussed mountain meadows, compared to 24 concerning Central Valley riparian systems. Mountain meadows deserve attention from riparian researchers because these wetlands support rich biological communities, and because they provide valuable scenic and recreational resources to California's expanding human population. Montane meadows also contribute a high proportion of the forage on forest grazing allotments and wilderness areas (Ratliff 1982). Land managers need information about the effects of grazing on biological resources in order to resolve these potentially conflicting uses of mountain meadows.

Montane meadow systems are the stronghold of California's population of Willow Flycatchers, an obligate riparian species whose range and numbers have dramatically diminished. Our particular concern is the status and habitat requirements of Willow Flycatchers in California, and the potential threats to Willow Flycatchers and other inhabitants of montane meadows from livestock grazing and Brown-headed Cowbird (*Molothrus ater*) parasitism. We make management recommendations to protect and enhance habitat for Willow Flycatchers and an assemblage of riparian bird species breeding in Sierra Nevada high elevation meadows in our conclusions.

We define meadows here as open wetlands characterized by hydrophytes, mesophytes, and dry herbland of the subalpine and alpine zone (Ratliff 1984). We focus on wildlife resources rather than floristic distinctions, and therefore we do not follow the finer meadow classifications delineated by Ratliff (1982) and Benedict (1984).

Perazzo Meadows and Lacey Valley, the sites at which we conducted most of our field research, occur along the Little Truckee River in Sierra County, California, approximately 32 km northwest of Truckee. These sites are at 2010 m on the east slope of the Sierra Nevada in Tahoe National Forest. Perazzo Meadows and Lacey Valley are very large, wet meadows dominated by grasses, rushes (*Juncus* spp.), and sedges (*Carex* spp.). The riparian zone consists of willow shrubs (*Salix lemmonii* and *S. jepsoni*) that parallel streams and old oxbows in the meadow. Lodgepole pine (*Pinus contorta* var. *murrayana*) forest surrounds the meadows.

Our discussion of Willow Flycatcher habitat requirements and potential threats to the species is based largely on field work conducted from mid June to late August in 1986 and 1987 at Perazzo Meadows and Lacey Valley. In addition to these studies, we surveyed meadows throughout the Sierra Nevada in June and July of 1986, searching for Willow Flycatchers and correlating their presence with habitat variables. The results of these surveys, discussed in detail by Harris and others (1987), also contribute to our analysis of Willow Flycatcher distribution, status, and habitat affinities.

Status of Willow Flycatchers in California

Willow Flycatchers have been extirpated as breeding birds from most of their former California range (Grinnell and Miller 1944; Flett and Sanders 1987; Harris and others 1987; Serena 1982). A few remaining populations inhabit isolated meadows of the Sierra Nevada. The largest of these mountain meadow populations occurs along the Little Truckee River drainage, which supports approximately 25 singing males. This species also occurs at lower elevations along the Kern, Santa Margarita, and San Luis Rey Rivers (Remsen 1978; Serena 1982; Unitt 1987). Recent surveys indicate a population of approximately 145 singing males in California (Harris and others 1987).

The loss of lowland riparian woodlands is probably the principal reason for the reduction of California's

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Willow Flycatcher population and the contraction of its range (Remsen 1978; Serena 1982). Nest parasitism by Brown-headed Cowbirds and livestock grazing may have also contributed significantly to population reduction (Gaines 1977, Serena 1982; Beedy and Granholm 1985; Sharp 1986; Taylor 1986; Taylor and Littlefield 1986). Other factors responsible for Willow Flycatcher declines in the Sierra Nevada may include loss of meadows due to reservoir and hydroelectric development, lodgepole pine encroachment on meadows, and habitat loss on wintering grounds (Serena 1982).

Habitat Requirements

Three features emerge as critical components of Willow Flycatcher habitat: large meadow size, water and willows. In the Sierra Nevada, Willow Flycatchers inhabit broad, flat meadows that are generally larger than 8 hectares, and that contain scattered clumps of willows (Harris and others 1987). They typically shun willow thickets on steep terrain, or narrow bands of willows bordered by conifer forest.

Water is an essential element on Willow Flycatcher territories. Twenty out of 22 territories at our study sites encompassed old oxbows, small secondary channels, or the Little Truckee River (Sanders and Flett 1988). All territories included areas with saturated soils, at least early in the season. Serena (1982) found that the portions of the meadows used by Willow Flycatchers were at least 40 percent wet. She also found that within meadows that contained dry areas, Willow Flycatchers occurred in the wettest sites.

In the Sierra Nevada, Willow Flycatchers are found only in meadows that contain willows (Harris and others 1987). All 22 Willow Flycatcher territories in our study site consisted of willow clumps separated by clearings. Willow cover on these territories averaged approximately 40 percent.

Willow Flycatchers build their nests in willows, and use these shrubs for foraging and singing perches, leaf and twig gleaning, and for cover. To provide suitable nesting habitat the willows should be at least 2 m in height, with a foliage density of approximately 50-70 percent. Nests generally are built at approximately 1 m in height, with about 1 m of willow cover above the nests (Sanders and Flett 1988).

Livestock Grazing

Direct Effects. Cattle can directly disturb Willow Flycatchers and other species nesting in montane meadows by knocking over nests in willow thickets or by crushing the eggs of ground-nesting birds. Stafford and Valentine (1985) and Valentine (1987) report that 4 of 20 nests monitored over a 4 year period were destroyed by cattle. Livestock also destroyed four nests shortly after the young fledged.

Cattle did not destroy any Willow Flycatcher nests in our study sites, although Perazzo Meadows contained approximately 150 cattle in 1986, and up to 360 in 1987. However, our data show that Willow Flycatchers invariably place their nests near the edge of willow clumps or along livestock trails, making them potentially vulnerable to disturbance by cattle (Flett and Sanders 1987).

In addition to Willow Flycatchers, at least 16 other bird species breeding in mountain meadows could be directly affected by cattle. Willow-nesting species include Yellow and Wilson's Warble (*Dendroica petechia and Wilsonia pusilla*), White-crowned Sparrow (*Zonotrichia leucophrys*), Song Sparrow (*Melospiza melodia*), and Red-winged Blackbirds (*Agelaius phoeniceus*). Ground nesting birds in mountain meadows are particularly vulnerable to trampling by livestock. These species include Canada Goose (*Branta canadensis*), Mallard (*Anas platyrhynchos*), Cinnamon Teal (*A. cyanoptera*), Virginia Rail (*Rallus limicola*), Sora (*Porzana carolina*), Killdeer (*Charadrius vociferus*), Spotted Sandpiper (*Actitis macularia*), Common Snipe (*Gallinago gallinago*), Wilson's Phalarope, (*Phalaropus tricolor*), Savannah Sparrow (*Passerculus sandwichensis*), and Lincoln's Sparrow (*Melospiza lincolnii*).

The potential for livestock to trample or upset bird nests depends on the overlap between the nesting season and presence of the livestock. Most species are incubating eggs or nestlings by late June, and are therefore particularly vulnerable to livestock disturbance from then until early July. Willow Flycatchers, however, are unusually late breeders. At our study sites they established territories around mid to late June. The first eggs were not laid until the second or third week of June. The latest of the young fledged by mid-August; most species fledged two weeks to one month earlier.

Indirect effects. Livestock indirectly affect Willow Flycatchers and other species nesting in willows by altering the vegetation and hydrology of montane meadows. Cattle and sheep consume the lower branches and shrub layers of streamside vegetation and consume or trample young riparian plants (Taylor 1986). Even grazing for only a few days or weeks has been observed to adversely affect regeneration of woody vegetation (Crumpacker 1984). Obligate riparian species are more affected by grazing than other bird species (Mosconi and Hutto 1982). Duff (1979) reports a large increase in the number of passerine birds after excluding cattle from a riparian area. This increase was due to the reestablishment of the middle story of willows.

Livestock grazing can also reduce water quality, compact soils, and accelerate streambank erosion (Thomas and others 1979; Platts 1984). Streambank erosion due to overgrazing can eventually result in incising and gullying of streambeds (Ratliff 1984). Gullying can lower the water table of formerly moist meadows (Van Haveren and Jackson 1986), thus drying the soils and altering the meadow's vegetative composition.

Taylor and Littlefield (1986) documented the adverse effects of cattle grazing on Willow Flycatchers and Yellow Warblers at Malheur National Wildlife Refuge in Oregon. They censused these two species along riparian transects with different grazing histories. Taylor and Littlefield found that transects that had been ungrazed for forty years supported significantly more Willow Flycatchers and Yellow Warblers than grazed transects. Willow foliage volume and density was significantly higher in the ungrazed transects. Heavily grazed transects had very few willows and no Willow Flycatchers or Yellow Warblers. Taylor and Littlefield also presented 12 years of U.S. Fish and Wildlife Service Breeding Bird Survey data, indicating a significant relationship between increased Willow Flycatcher numbers and decreased grazing intensity.

Cowbird Parasitism

Brown headed Cowbird nest parasitism has been suggested as a cause of the Willow Flycatcher's decline in California (Remsen 1978). Their decline in central and coastal California coincides roughly with the spread of cowbirds in the 1920's and 1930's (Gaines 1977, Garrett and Dunn 1981). Friedmann (1963) reported 150 instances of Brown-headed Cowbird parasitism of Willow Flycatchers, 41 of which were reports from southern California.

Studies by Harris (in prep.) in 1987, at The Nature Conservancy Kern River Preserve, revealed intense parasitism by Brown-headed Cowbirds on Willow Flycatcher nests. The Kern River Preserve is a willow-cottonwood riparian woodland at an altitude of 750 m. At least 13 and possibly 16 of 19 Willow Flycatcher nests at the Kern River Preserve were parasitized by cowbirds. The losses due to parasitism resulted in a low egg-to-fledgling success rate of 24 percent.

While cowbird parasitism seems to be a major contributor to nesting failures of lowland populations of Willow Flycatchers, there is less evidence of cowbird parasitism in the higher elevations of the Sierra Nevada. One out of 22 Willow Flycatcher nests at our study sites was parasitized by a Brown-headed Cowbird. The single cowbird fledged successfully, but its three Willow Flycatcher nestmates did not survive. The only other record of Willow Flycatcher nest parasitism in the mid to high elevation Sierra Nevada was from the Lake Tahoe region in 1960 (Gaines 1977).

Stafford and Valentine (1985) suggest that the peak of Willow Flycatcher egg-laying in the high-elevation Sierra Nevada often occurs after the peak of the cowbird breeding season. King (1954), studying parasitism in the state of Washington, also noted that the peak of egg deposition by Willow Flycatchers occurred after the height of the cowbird egg-laying season passed. He found only 2 of 44 Willow Flycatcher nests parasitized. On the other hand, studies of Willow Flycatcher populations living at high elevation (2,500 m) sites in northcentral Colorado documented high parasitism rates (Sedgewick and Knopf 1988). At least 40 percent (11 out of 27) of the Willow Flycather nests found during that study were parasitized by Brown-headed Cowbirds.

Cowbird parasitism on Willow Flycatcher nests is a potential threat at high elevations and clearly is a serious problem at lower elevations in California. Laymon (1987) suggests that reducing or eliminating livestock grazing in mountain meadows could increase the reproductive success of Willow Flycatchers. Elimination of grazing allows grass to grow too tall to be suitable cowbird foraging habitat and removes the large grazers with which cowbirds associate.

Conclusions and Management Recommendations

Wet meadows of the Sierra Nevada are critical resources for the rare Willow Flycatcher and for many other breeding birds. These meadows are typically managed for livestock production, often to the detriment of wildlife. The following recommendations provide guidelines for protecting and enhancing mountain meadows that support Willow Flycatchers. These management recommendations would also confer benefits to a diverse array of riparian birds breeding in montane meadows.

- Eliminate or Delay Grazing – To avoid the direct and indirect impacts associated with livestock, grazing should be reduced or eliminated in meadows and riparian areas that support Willow Flycatchers. One alternative to eliminating grazing entirely is to delay putting cattle on high elevation meadows until mid-August, after Willow Flycatchers have fledged. Another alternative is to exclude cattle from the vicinity of streams and riparian vegetation by fencing, providing an alternative source of water for livestock by means of stocktanks. These recommendations have the added benefit of protecting nests and habitat for at least 16 other species of birds that breed in mountain meadows.

- Acquire Habitat – Montane meadows and riparian areas that support Willow Flycatchers should be protected and managed as a primary resource on public lands. Occupied and potential sites on private lands should be protected by conservation easements with landowners or by land purchases. In particular, efforts should be made to permanently protect the meadow system along the Little Truckee River. These meadows support the second largest known Willow Flycatcher population in the state, and the largest Sierra Nevada population.
- Avoid Developments Adjacent to Montane Meadows – Cowbirds frequently feed in disturbed areas where high energy foods are concentrated, including residential housing with bird feeders, campgrounds, corrals, and garbage dumps (Airola 1986). Such developments should be kept away from riparian areas to minimize the impacts of the cowbirds on Willow Flycatchers and other species nesting in willow thickets of mountain meadows. Excluding residential and housing developments near meadows would also reduce the potential for disturbance from humans, dogs, cats, and off-road vehicles, all of which could have significant impacts on birds breeding in mountain meadows.
- Revegetate and Restore Montane Meadows – The response of Willow Flycatchers to revegetation and meadow restoration should be explored as part of a comprehensive plan of habitat protection and enhancement. Restoration of Willow Creek in Modoc County provides a promising model of such efforts (Clay 1984). In addition, Valentine (1987) makes some specific suggestions for restoring meadows that support Willow Flycatchers.

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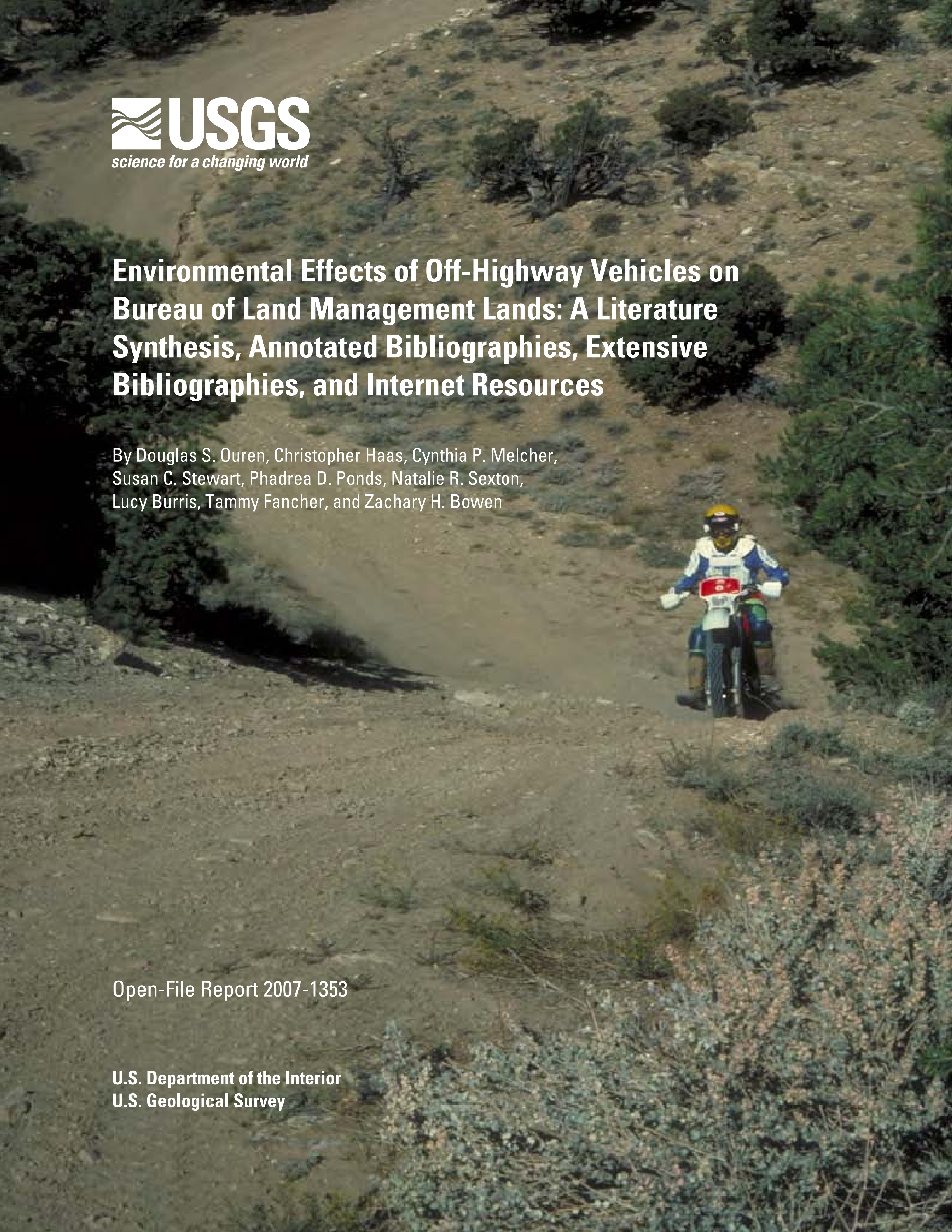

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Environmental Effects of Off-Highway Vehicles on Bureau of Land Management Lands: A Literature Synthesis, Annotated Bibliographies, Extensive Bibliographies, and Internet Resources

By Douglas S. Ouren, Christopher Haas, Cynthia P. Melcher, Susan C. Stewart, Phadrea D. Ponds, Natalie R. Sexton, Lucy Burris, Tammy Fancher, and Zachary H. Bowen

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Conversion Factors

Inch/Pound to SI (International System of Units)

Multiply	By	To obtain
Length		
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
yard (yd)	0.9144	meter (m)
Area		
square foot (ft^2)	929.0	square centimeter (cm^2)
square foot (ft^2)	0.09290	square meter (m^2)
square inch (in^2)	6.452	square centimeter (cm^2)
square mile (mi^2)	2.590	square kilometer (km^2)
Volume		
cubic inch (in^3)	16.39	cubic centimeter (cm^3)
cubic yard (yd^3)	0.7646	cubic meter (m^3)
Mass		
ounce, avoirdupois (oz)	28.35	gram (g)
pound, avoirdupois (lb)	0.4536	kilogram (kg)
ton, short (2,000 lb)	0.9072	megagram (Mg)
ton, long (2,240 lb)	1.016	megagram (Mg)

SI to Inch/Pound (English System of Units)

Multiply	By	To obtain
Length		
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
Area		
square centimeter (cm^2)	0.001076	square foot (ft^2)
square meter (m^2)	10.76	square foot (ft^2)
square centimeter (cm^2)	0.1550	square inch (in^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
Volume		
cubic centimeter (cm^3)	0.06102	cubic inch (in^3)
cubic meter (m^3)	1.308	cubic yard (yd^3)
Mass		
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	Pound, avoirdupois (lb)
megagram (Mg)	1.102	ton, short (2,000 lb)
megagram (Mg)	0.9842	ton, long (2,240 lb)

Glossary

ATV (All-Terrain Vehicle) Small, motorized 3- or 4-wheeled vehicles specifically designed for off-road use. The American National Standards Institute (ANSI) further defines an ATV as a vehicle that travels on low-pressure tires, with a seat that is straddled by the operator, and with handlebars for steering control. By the current ANSI definition, it is intended for use by a single operator, although a change to include 2-seaters (in tandem) is under consideration. Herein, the definition of ATV coincides with the description above and does not include passenger vehicles, including sport-utility vehicles or 4-wheel-drive jeeps.

fugitive dust Dust raised by mechanical (anthropogenic) disturbance of granular material exposed to and becoming suspended in the air, then carried by wind. Arises from "nonpoint" sources—such as unpaved roads, agricultural tilling operations, aggregate storage piles, and heavy construction—rather than "point" sources—such as confined flow streams discharged to the atmosphere from a stack, vent, or pipe.

indicator threshold For a given land health indicator (or set of indicators), the value(s) at or above which management action may be triggered or required.

land health The condition of natural resource attributes, including soils and site stability, hydrologic function, and biotic integrity.

OHV Defined herein as any civilian off-highway vehicle, including motorcycles, motorized dirt bikes, ATVs (see definition above), snowmobiles, dune buggies, 4-wheel-drive jeeps, sport-utility vehicles, and any other civilian vehicles capable of off-highway, terrestrial travel (including utility vehicles [UTVs] and ATVs with more than 4 wheels).

OHV route Defined herein as any unpaved route created for OHV travel, including single-track paths or trails, two-tracks, and unimproved or improved dirt/gravel roads. Herein, this term is also applied to "rogue" (undesignated or unauthorized) routes created by OHV users in closed or limited areas.

population dynamics Herein, used broadly to include wildlife or vegetation population size, density, and/or distribution (both spatial and temporal); rates of birth/germination, death, and/or survivorship; population gender/age-class structure; population genetics; and/or the rates/directions of change in all these parameters.

right-of-way habitat Habitat provided within the legal description of a given transportation corridor.

sink population For a given metapopulation, a population sink is a local area or habitat where the local population's reproductive rate is lower than the required replacement rate (in other words, a sink population is eventually extirpated without immigration of individuals from other areas). Population sinks often occur where there is excessive predation pressure and/or poor habitat quality.

source population For a given metapopulation, a population source is a local area or habitat where the local population's reproductive rate is greater than the required replacement rate. Excess individuals produced from a source population may emigrate to join sink populations, thereby keeping the sink populations from becoming extirpated.

stream order A stream's order is determined by its confluence with other streams: first-order streams are headwaters, the confluence of two first-order streams forms a second-order stream, the confluence of two second-order streams forms a third-order stream, and so on.

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Executive Summary

This report and its associated appendixes compile and synthesize the results of a comprehensive literature and Internet search conducted in May 2006. The literature search was undertaken to uncover information regarding the effects of off-highway vehicle (OHV) use on land health, or “natural resource attributes,” and included databases archiving information from before OHVs came into existence to May 2006. Information pertaining to socioeconomic implications of OHV activities is included as well. The literature and Internet searches yielded approximately 700 peer-reviewed papers, magazine articles, agency and non-governmental reports, and internet websites regarding effects of OHV use as they relate to the Bureau of Land Management’s (BLM) standards of land health. Discussions regarding OHV effects are followed by brief syntheses of potential indicators of OHV effects, as well as OHV-effects mitigation, site-restoration techniques, and research needs.

Terminology Used in This Report

The BLM has definitions for several road and trail types; however, the OHV literature often uses somewhat different definitions. Whereas all terms are useful within their own contexts, herein the general term “OHV routes” is used to simplify discussions concerning all types of unpaved roads and trails, whether designated or unauthorized, and “roads” or “highways” are used to simplify discussions concerning paved roads. The definition of OHV also varies by agency and author, and to simplify discussions herein, OHV may include off-highway motorbikes, ATVs, dune buggies, snowmobiles, 4-wheel drive jeeps, motorcycles, some types of 4-wheel drive automobiles (including sport-utility vehicles), and any other civilian vehicle specifically designed for off-road travel. OHV type or route/road type are specified if a given discussion warrants and if the literature cited in that discussion specified OHV type or route/road type.

How to Use This Report

Major sections of this document comprise a “manager’s report,” which includes a literature synthesis and related discussions of (1) OHV effects on natural resource attributes and socioeconomics; (2) indicators described in the literature to evaluate/monitor OHV effects on natural resource attributes and could serve as potential indicators in future research or monitoring programs; (3) mitigation and site-restoration techniques used for OHV-use areas; and (4) research and monitoring needs pertaining to OHV-effects. This document also includes extensive bibliographies pertaining to OHV effects on natural resources. It is recommended that readers focus first on the manager’s report, as it provides the basic understanding of OHV effects and potential approaches to researching, monitoring, and/or managing OHV effects. Reading the Executive Summary, the summaries provided in each section, and the conclusion may suffice for those seeking a quick overview. **To facilitate a rapid review, section summaries are placed at the beginning of their respective sections and do not contain in-text citations. For a more in-depth review (with in-text citations), the entire manager’s report should be read.** Appendix 1 provides the extensive bibliographies, and Appendix 2 details the literature/Internet search methods and summarizes the search results (including tables and graphs).

OHV Effects

OHV Effects on Soils and Watersheds

The primary effects of OHV activity on soils and overall watershed function include altered soil structure (soil compaction in particular), destruction of soil crusts (biotic and abiotic) and desert pavement (fine gravel surfaces) that would otherwise stabilize soils, and soil erosion. Indicators of soil compaction discussed in the OHV effects literature include soil bulk density (weight per unit of volume), soil strength (the soil's resistance to deforming forces), and soil permeability (the rate at which water or air infiltrate soil). Generally, soil bulk density and strength increase with compaction, whereas permeability decreases with compaction. As soil compaction increases, the soil's ability to support vegetation diminishes because the resulting increases in soil strength and changes in soil structure (loss of porosity) inhibit the growth of root systems and reduce infiltration of water. As vegetative cover, water infiltration, and soil stabilizing crusts are diminished or disrupted, the precipitation runoff rates increase, further accelerating rates of soil erosion.

OHV Effects on Vegetation

Plants are affected by OHV activities in several ways. As implied above, soil compaction affects plant growth by reducing moisture availability and precluding adequate taproot penetration to deeper soil horizons. In turn, the size and abundance of native plants may be reduced. Above-ground portions of plants also may be reduced through breakage or crushing, potentially leading to reductions in photosynthetic capacity, poor reproduction, and diminished litter cover. Likewise, blankets of fugitive dust raised by OHV traffic can disrupt photosynthetic processes, thereby suppressing plant growth and vigor, especially along OHV routes. In turn, reduced vegetation cover may permit invasive and/or non-native plants—particularly shallow-rooted annual grasses and early successional species capable of rapid establishment and growth—to spread and dominate the plant community, thus diminishing overall endemic biodiversity.

OHV Effects on Wildlife and Habitats: Native, Threatened, and Endangered Species

Habitats for native plants and animals, including endangered and threatened species, are impacted by OHVs in several ways. A salient effect is habitat fragmentation and reduced habitat connectivity as OHV roads and trails proliferate across the landscape. Reduced habitat connectivity may disrupt plant and animal movement and dispersal, resulting in altered population dynamics and reduced potential for recolonization if a species is extirpated from a given habitat fragment. Wildlife is also directly affected by excessive noise (decibel levels/noise durations well above those of typical background noise) and other perturbations associated with OHV activities. Disturbance effects range from physiological impacts—including stress and mortality due to breakage of nest-supporting vegetation, collapsed burrows, inner ear bleeding, and vehicle-animal collisions—to altered behaviors and population distribution/dispersal patterns, which can lead to declines in local population size, survivorship, and productivity.

OHV Effects on Water Quality

The effects of OHV activities on water quality can include sedimentation (deposited solids), turbidity (suspended solids), and pollutants within affected watersheds. Sedimentation increases because compacted soils, disrupted soil crusts, and reduced vegetation cover can lead to increased amounts and velocities of runoff; in turn, this accelerates the rates at which sediments and other debris are eroded from OHV-use areas and flushed to aquatic systems downslope. Pollutants associated with deposition of OHV emissions and spills of petroleum products may be adsorbed to sediments, absorbed by plant material, or dissolved in runoff; once mobilized, these contaminants may enter aquatic systems.

OHV Effects on Air Quality

Air quality is affected when OHV traffic raises fugitive dust and emits by-products of combustion. Because wind can disperse suspended particulates over long distances, dust raised by OHV traffic can blanket plant foliage and disperse dust-adsorbed contaminants well beyond a given OHV-use area. Primary combustion by-products potentially affecting air quality in OHV-use areas include (but are not limited to) polycyclic aromatic hydrocarbons, sulfur dioxide (SO_2), nitrogen oxides (NO_x), and ozone (O_3). Although leaded gasoline has not been used in the United States since 1996, lead emissions deposited prior to the ban on leaded gasoline may persist for decades and continue impacting ecosystems as wind and water erosion continue to mobilize lead and other contaminants downwind (or downslope) of contaminated soils.

Socioeconomic Implications of OHV Use

For the purposes of this document, the socioeconomics of OHV use include (1) OHV user demands, concerns, and attitudes; (2) the economic effects of OHV use on communities near OHV-use areas; (3) and the effects of OHV use on other land users. Although not one of BLM’s land health considerations, the socioeconomic implications of OHV use have significant direct and indirect effects on land health. As the popularity of OHV recreation increases, socioeconomic factors become increasingly important considerations in understanding and mitigating the overall effects of OHV use on land health. OHV recreation can have significant economic value to local communities where and when OHV use is popular; however, the economic costs to those communities remain unknown. OHV use also can lead to conflicts among different land users—both OHV users and people seeking non-motorized forms of recreation—within OHV-use areas and nearby areas. Crowding of designated OHV areas may encourage unauthorized use in closed areas, and adjacent or overlapping use types may cause dissatisfaction or discourage recreation altogether, which can diminish public support for land-management programs.

Potential Indicators (Both Direct and Indirect) for Evaluating and Monitoring OHV Effects

Soil Health and Watershed Condition

- Soil strength
- Soil bulk density
- Soil permeability (rates of air and water infiltration)
- Erosion rate
- Level of sedimentation or turbidity in wetlands
- Surface changes (for example, gully erosion)
- Presence/condition of soil crusts

Vegetation Health

- Plant community composition (including species and structural diversity, ratio of native to non-native or invasive species)
- Abundance of individuals and/or stem density

- Percent vegetation cover
- Plant size
- Growth rate
- Biomass

Habitat Condition and Health of Wildlife Populations (including indirect indicators)

- Habitat patch size and connectivity
- Community composition (including species diversity, ratio of native to non-native or invasive species)
- Population size, density, and trend
- Spatiotemporal distribution of populations
- Survivorship and mortality rates
- Productivity and body mass
- Age-class and gender structure
- Frequency of OHVs passing through a given area and associated wildlife mortalities rates
- Road or trail type and width
- Level (decibels), duration, and timing of traffic noise

Water Quality

- Sedimentation rate
- Levels of turbidity and suspended solids
- Contaminant levels, including petroleum-derived compounds from spills and emissions, such as benzene; ethylbenzene; m-, p-, and o-xylene; toluene; 1,3-butadiene; and lead

Air Quality

- Level of dust particulates
- Particulate levels of OHV emission by-products, such as polycyclic aromatic hydrocarbons, aldehydes, carbon monoxide, nitrogen oxides, ozone, and sulfur oxides

Socioeconomics

- User satisfaction with recreation experiences
- User compliance with OHV (or other) regulations
- User knowledge regarding effects of recreation activities on various aspects of land health
- Distribution and intensity of OHV versus non-motorized recreation and other land uses
- Extent to which unauthorized trails are created and damage to vegetation occurs
- Trends in local economic indicators associated with OHV and non-motorized recreation and other land uses, such as sales of camping equipment, gasoline, restaurants, lodging facilities

Mitigation and Site-Restoration Techniques

Balancing OHV-user preferences with protecting land health and the needs of other land users requires careful study and planning, as well as appropriate management strategies. Prior planning for locating OHV areas before they are opened to the public can preclude undesirable effects of OHV use and costly site restoration. Once a site has been used, however, trail/area closures, signage, and other visual cues, as well as enforcement and limiting visitor numbers through "rationing," are among the tools used to preclude additional effects.

Because habitat fragmentation is particularly difficult to repair, planning and management designed to maintain habitat connectivity are crucial to minimize fragmentation. Variation in impacts requires various restoration techniques. Whereas a single OHV pass on a xeric landscape may cause long-lasting damage, a similar single pass on a mesic landscape may require no treatment at all. Restoration approaches may include replacing native soil where erosion has removed the topsoil and exposed the underlying bedrock, seeding with indigenous plants, inoculating soils with native microbes and mycorrhizae, scarifying, and/or mulching. Ultimately, the success of such measures depends on the nature and intensity of the disturbance, topography, soil type, climate, and the ability of land managers to enforce closures and prevent the proliferation of new routes.

Monitoring and Research Needs to Support OHV Management Decisions

Elucidating OHV impacts on soils and watersheds, vegetation, wildlife and their habitats, water and air quality, and/or socioeconomics, whether through monitoring or experimental research studies, will require careful planning and appropriate, rigorous study design. Results of many past ecological studies on the effects of OHVs may be regarded as preliminary, particularly for those that lacked comparable treatment and control sites and site replication. Overall, the reliability and value of monitoring and research results would increase significantly by including a broad range of spatial and temporal scales—from microhabitat to landscape or ecosystem scale, and from short-term (seasonal) to long-term (decades)—within the full range of impacted habitat types represented on BLM lands. The full array of site types also needs evaluation, including designated OHV-use sites, undesignated (rogue) OHV-use sites, unused areas, and restoration sites. Multiple- and simultaneous-assessment techniques that take advantage (and push the advancement) of existing and emerging technologies are needed to fully represent the scale and diversity of OHV impacts on the abiotic and biotic components of affected lands and communities, and to ascertain indicator thresholds as they pertain to BLM's land health standards. More specifically, monitoring and research needs pertaining to OHV use and impacts include (but are not limited to)

- well-designed monitoring programs and experimental studies that incorporate planned comparisons of treatment (OHV-impacted) and control (unimpacted/reference) sites;
- before and after OHV-impact studies;
- studies at various spatial and temporal scales across all impacted habitat types;
- studies on habitat fragmentation and road-edge effects caused by OHV activities;
- studies on various gradients in OHV disturbance at various distances from OHV routes;
- studies to improve the understanding of the physical and chemical dynamics of soil compaction;
- studies evaluating the effects of erosion, sedimentation, and turbidity at both local (immediately downslope of OHV-affected sites) and landscape (throughout impacted watersheds) scales;
- studies of OHV effects on plant and animal population dynamics;
- simultaneous evaluations of wildlife responses and OHV route-specific variables;
- improvements in techniques for successful site restoration;

- improvements in techniques and technologies for assessing OHV impacts over large areas and long periods of time;
- effectiveness evaluations of various techniques to manage OHV use and its ecological and socioeconomic effects while simultaneously providing the greatest satisfaction among all land users; and
- studies that determine the economic and sociological costs of OHV use.

1.0 Introduction

1.1 Issue Context: Bureau of Land Management Land Health and Off-highway Vehicle Use

1.1.1 Bureau of Land Management Land Health Standards

In the mid-1990s, *Rangeland Health: New Methods to Classify, Inventory, and Monitor Rangelands* was published to examine the scientific basis and success of methods used by federal agencies to inventory, classify, and monitor rangelands, and to make recommendations for improvements to these methods (Committee on Rangeland Classification, Board on Agriculture, and National Research Council, 1994). Therein, rangeland health was defined as "…the degree to which the integrity of the soil and the ecological processes of rangeland ecosystems are sustained." In 2001, the BLM published *Rangeland Health Standards*, a process framework for assessing rangeland health via interdisciplinary teams that, at a minimum, would evaluate (1) watershed function; (2) nutrient cycling and energy flow; (3) water quality; (4) habitat for endangered, threatened, proposed, candidate, or special status species; and (5) habitat quality for native plant and animal populations (U.S. Bureau of Land Management, 2001). Subsequently, it was clarified that the term "rangeland" is interchangeable with "land" and that "…'the rangeland health standards' really apply to the condition of the land itself regardless of the uses that may influence the health of that land" (U.S. Bureau of Land Management, 2007). In other words, the standards for rangeland health apply to all BLM lands, whether used for grazing, off-highway vehicle (OHV) recreation, or any other use permitted to occur on BLM lands.

The standards of rangeland health establish minimum resource conditions that must be achieved and maintained to ensure the "proper functioning condition" (see Barrett and others, 1995)—both physical and biological—and sustainability of BLM lands. These standards, however, must be relative to the native conditions for a given "reference site," as native conditions vary widely among sites on BLM lands. For example, the BLM defines healthy soils in northwest California as "exhibit[ing] characteristics of infiltration, fertility, permeability rates, and other functional and physical characteristics that are appropriate to soil type, climate, desired plant community, and land form" (personal communication from M. Karl to V. Josupait, U.S. Bureau of Land Management, Denver, Colorado, May 2001).

The BLM's technical reference on *Interpreting Indicators of Rangeland Health* recognizes three broad categories of natural resource attributes for assessing land health: (1) soils and site stability, (2) hydrologic function, and (3) biotic integrity (Pellant and others, 2005). Relative to off-highway vehicle (OHV) impacts on ecosystem health, soil/site stability and hydrologic or watershed function pertain (but are not limited) to erosion and extent of surface changes and patterns of water flow, including infiltration. Biotic integrity pertains to community structure and functionality of plants. Within these three categories, overall land health is qualitatively assessed via 17 parameters (or indicators) that indicate the presence, number, extent, percent, and/or depth or height of (1) rills, (2) water flow patterns, (3) erosional pedestals/terracettes, (4) bare ground, (5) gullies and gully erosion, (6) wind scoured blowouts and/or depositional areas, (7) litter movement, (8) soil surface resistance to erosion, (9) soil surface structure and content of soil organic matter (SOM), (10) effect of plant community composition and spatial distribution on infiltration and runoff, (11) compaction layer, (12) dominance hierarchy of functional/structural groups in plant communities, (13) mortality and decadence among plant functional groups, (14) litter cover, (15) expected above-ground annual production, (16) potential invasive species, and (17) perennial plant reproductive capability (Pellant and others, 2005). Many of the indicators listed above may be assessed in terms of the condition and extent of abiotic (chemical) and biotic soil crusts, and soil surfaces of small stones known as "desert pavement," as they help stabilize soils and/or cycle nutrients through the system. Indicators 1-11 largely pertain to soil/site stability and hydrologic/watershed function, whereas parameters 12-17 are primarily indicators of biotic integrity, although there is overlap among the two groups.

1.1.2 Increasing OHV Use

An important factor affecting the health of BLM lands is the use of OHVs. In 1993, 2,920,000 all-terrain vehicles (ATVs) and off-highway motorcycles were estimated to be in use (Cordell and others, 2005). Between 1995 and 2003, sales of ATVs and off-highway motorcycles tripled, increasing the number of ATVs and off-highway motorcycles in use to 8,010,000 (Cordell and others, 2005). Because the popularity of OHV-based recreation is relatively recent and still increasing (see Matchett and others, 2004), the full range of short- and long-term impacts has yet to be fully realized or understood. Overall, it is clear that OHV use on public lands is and will continue to be an important management issue.

1.2 Objectives, Scope, Organization, and Use of This Report

1.2.1 Objectives

The objectives of this report are twofold. This first is to synthesize the results of a comprehensive literature search on what is currently known about the effects of OHV activities as they relate to the BLM's land health standards (U.S. Bureau of Land Management, 2001). These discussions include socioeconomic implications of OHV use—including preferences of OHV users, effects of OHV activities on other land users, and the economic impacts of OHV recreation on local economies—because understanding these factors and incorporating that knowledge into management plans and policies will be crucial to management success. The second objective is to discuss the indicators of land health and socioeconomics described in the OHV effects literature, as they have potential usefulness for evaluating or monitoring lands and land users affected by OHV use. This report also contains brief overviews of mitigation approaches, site-restoration techniques, and monitoring and research needs described in the OHV-effects literature.

1.2.2 Geographical Scope

Although the vast majority of literature and other sources consulted address OHV impacts on ecosystems in the western United States, there are a number of useful references that address OHV use or impacts of roads in other parts of the United States and in other countries. Some of these sources have been included to provide additional information not provided elsewhere and/or to broaden the scope and relevance of this document.

1.2.3 Organization

The main body of this document—referred to herein as the "manager's report" (Executive Summary and Sections 1-7)—includes the literature synthesis of OHV effects; annotated bibliographies that typify the body of research regarding effects of OHV activities on BLM's land health standards; sections regarding indicators of OHV effects, mitigation and site-restoration techniques, and monitoring and research needs; and a listing of all literature cited in the manager's report. To facilitate the logical flow of information, the natural resource attributes addressed in BLM's land health standards serve as the underlying organizational structure of all major sections in the report, as indicated by parallel subsection headings pertaining to (1) soils and watersheds, (2) vegetation, (3) habitat for native plants and wildlife, (4) water quality, and (5) air quality. Although not part of the BLM's land health agenda, a sixth subsection has been included in each major section to address the socioeconomic considerations of OHV use, because, ultimately, it is socioeconomic factors (including conflicts among land users, preferences of OHV users, and economics of OHV use) that drive changes in overall land health.

1.2.4 Tips on Navigating This Document

It is recommended that readers focus first on the manager's report, as it provides the basic understanding of OHV effects and possible indicators to use in research, monitoring, and/or management programs. Reading the Executive Summary, subsection summaries, and the overall conclusion may suffice for those seeking a quick overview. **For quick and easy reference, subsection summaries on OHV effects are placed at the top of their respective subsections. In-text citations were purposely left out of all summaries to enhance readability. For a more in-depth, fully cited review, reading the entire manager's report is recommended.**

Appendixes 1 and 2 provide further information and more extensive resources for those needing high levels of detail. The Extensive Bibliographies (Appendix 1) includes the approximately 700 publications and reports uncovered through the literature search (from 1960 to May 2006) that pertain to effects of OHV use. Appendix 2 details the literature- and Internet-search methods and provides tabular and graphical summaries of literature, websites, and associated resources, as they pertain to OHV effects on natural resources and related policies.

1.3 Definitions of OHV Routes/Roads, Vehicles, and Activities

1.3.1 Definitions of Roads and Trails Used in This Report

BLM's *Roads and Trails Terminology* document (U.S. Bureau of Land Management, 2006) defines (1) a road as "a linear route declared a road by the owner, managed for use by low-clearance vehicles having four or more wheels, and maintained for regular and continuous use;" (2) a primitive road as "a linear route managed for use by four-wheel drive or high-clearance vehicles;" and (3) a trail as "a linear route managed for human-powered, stock, or off-highway vehicle forms of transportation or for historical or heritage values." Bolling and Walker (2000) offer more detailed definitions of unpaved roads and trails: (1) graded, improved roads are those from which the topsoil has been removed by bulldozer and characterized by the presence of lateral berms; (2) unimproved roads are those not graded consistently; (3) jeep trails are four-wheel drive tracks impacted only by vehicular traffic and generally characterized by a center berm; and (4) single tracks are severely compacted trails generated by OHVs.

The literature includes numerous reviews on the ecological effects of paved as well as unpaved roads (see Andrews, 1990; Forman and Alexander, 1998; Spellerberg, 1998; Trombulak and Frissell, 2000). Ecosystems of the West, however, are especially vulnerable to OHV-related activities on unpaved (gravel or dirt) roads and trails due to the effects they impose on soils and vegetation, which may take centuries to recover (Webb, 1982; Lovich and Bainbridge, 1999). Furthermore, unpaved roads comprise the majority of OHV routes used throughout public lands in the western United States. Therefore, the primary considerations in this report are unpaved roads and trails. References and discussions regarding effects of paved roads are not entirely excluded, however, as they often inform the potential scope of OHV effects not otherwise addressed. When necessary to do so, the type of road is specified.

All of terms described above for roads and trails are useful within their own contexts, but to distinguish between them and those used in the OHV literature would unnecessarily complicate discussions herein. Therefore, except where there is a need to specify in more detail, unpaved roads, primitive roads, and unpaved trails, are referred to as “routes,” regardless of their intended purpose or how they are maintained. “Routes” also include unauthorized or “rogue” roads and trails created by OHV users traveling off officially designated roads, primitive roads, and trails. Paved roads are referred to as “roads.”

1.3.2 Definitions of OHVs and OHV Activities Used in This Report

BLM’s *Roads and Trails Terminology* document (U.S. Bureau of Land Management, 2006) defines an OHV as “any motorized vehicle capable of—or designated for—travel on or immediately over land, water, or other natural terrain” (excluding nonamphibious registered motorboats; military, fire, emergency, or law enforcement vehicles used for emergency purposes; official vehicles used expressly by an authorized officer; and military vehicles). Cordell and others (2005) further specify that OHVs may include motorcycles and off-highway motorbikes, ATVs, dune buggies, snowmobiles, most 4-wheel drive automobiles (jeeps, sport utility vehicles), and any other civilian vehicle specifically designed for off-road travel. For the purpose of this document, OHV is defined in accordance with BLM terminology and includes those vehicles listed by Cordell and others (2005), as well as utility vehicles (UTVs) and ATVs with more than 4 wheels.

There are numerous activities and outcomes directly and indirectly associated with OHV use. To simplify discussions of OHV activities herein, the term “OHV activities” largely refers to driving OHVs for recreation. In certain contexts, OHV activities also may include driving and parking vehicles that tow trailers carrying OHVs and loading and unloading OHVs from trailers. Whereas the use of 4-wheel drive jeeps, automobiles, and sport utility vehicles is largely restricted to unpaved roads and jeep trails, the effect of ATVs and off-highway motorcycles extends well beyond them to double- and single-track trails, as well as unauthorized roads and trails. Thus, “OHV activities” may include driving OHVs on authorized roads/routes and on (or creating) unauthorized routes.

2.0 Effects of OHV Travel on Natural Resource Attributes and Socioeconomics

2.1 Scale and Patterns of OHV Activities and Their Effects

Temporal and spatial scales are crucial considerations when evaluating or monitoring effects of any factor on ecosystems (Noon, 2003; Ringold and others, 2003). In discussing OHV effects on desert ecosystems, Brooks and Lair (2005) and Matchett and others (2004) describe the impacts of OHV activities at various spatial and temporal scales. At the highly localized spatial scale, one might find soil compaction taking place within the confines of a single OHV route, the effects of which might be limited to poor infiltration of water and reduced plant cover in the route itself. Brooks and Lair (2005) go on to explain, however, that the cumulative impacts of any one effect at many sites can result in impacts at much greater scales. For example, if networks of OHV routes criss-cross large areas, the habitat connectivity that previously facilitated animal movements within that landscape may be disrupted (Forman and others, 2003: p. 129-134). Similarly, a single pass by one OHV probably has negligible effects on animal distributions, but if OHV traffic is intense and chronic, animal densities may decline (Reijnen and others, 1995, 1997) as cumulative impacts of this one effect occurring at many sites across a landscape disrupt entire populations. Furthermore, any **direct** effect of OHV use is also likely to have **indirect** effects that go beyond the site of disturbance. For example, reduced plant cover (direct effect) can result in greater rates of erosion in and around an OHV route, which, in turn, might increase sedimentation and turbidity in wetlands downslope of the route (indirect effect). Overall, Brooks and Lair (2005) conclude that effects of OHV use need to be evaluated at appropriate scales, which must take into account the scale at which OHV activities and ecosystem responses occur (Brooks and Lair, 2005). Overall, most, if not all, effects of OHV activities described in sections 2.2 through 2.6 can occur from the very localized and/or ephemeral scale to the landscape and/or long-term scale. By the same token, any direct effect may have a number of indirect effects, the magnitude of which may depend on the spatial and temporal scales at which a direct effect occurs.

At an OHV site in California, Matchett and others (2004) classified OHV routes in terms that describe intensity and pattern of use. First, OHV lines (routes) were categorized as either *dirt* (lines most likely created for or by OHV use) or as *wash* (lines most likely created by water flow, but possibly used by OHVs). They went on to define levels of OHV use that also imply patterns of use: (1) densely tracked reticulate (OHV lines evident in a web-like pattern, but too dense and overlapping to distinguish individually); (2) densely tracked hill-climb (OHV lines evident on slopes, but too dense and overlapping to distinguish individually; (3) densely tracked intersection (OHV lines evident at intersections, but too dense and overlapping to distinguish individually; (4) densely tracked right-of-way (OHV lines evident near pipelines, transmission lines, and highway right-of-ways, but too dense and overlapping to distinguish individually; (5) densely tracked wash (OHV lines evident within washes, but too dense and overlapping to distinguish individually; (6) denuded hill-climb (OHV lines not readily evident on hill-climbs, but the preponderance of densely tracked areas in the vicinity indicate that OHV use was probably high within that area); and (7) denuded staging (OHV routes not readily evident in relatively flat camping areas, but the preponderance of densely tracked areas in the vicinity indicated that OHV use was probably high within that area) (Matchett and others, 2004). Overall, these categories indicate that OHV use often entails many criss-crossing routes as opposed to a single route. They also point out that OHV use may be heaviest on slopes, along right-of-ways, in washes, and in the vicinity of camping facilities.

2.2 OHV Effects on Soils and Watersheds

2.2.1 Section Summary

Important effects of OHV activities on soils and watershed function include soil compaction, diminished water infiltration, diminished presence and impaired function of soil stabilizers (biotic and abiotic crusts, desert pavement), and accelerated erosion rates. Compacted soil inhibits infiltration of precipitation. In turn, soil moisture available to vegetation is diminished, volumes and velocities of precipitation runoff increase, and soil erosion accelerates, leading to the formation of gullies and other surface changes. Additionally, soil compaction may inhibit root growth among plants, in which case organic matter, litter, soil fertility, and vegetative cover are diminished, further exacerbating the soil's susceptibility to erosion. Where biotic and chemical crusts or other soil stabilizers are disturbed or destroyed, soil erosion from water and wind may increase beyond rates found in undisturbed sites with similar soils and conditions; nutrient-cycling processes also are likely to be disrupted, potentially leading to declines in soil fertility.

2.2.2 Soil Compaction and Reduced Water Infiltration

One of the most common and important effects of OHV activities is soil compaction (Liddle, 1997), which diminishes water infiltration, destroys soil stabilizers (biotic and abiotic crusts, desert pavement), and promotes greater rates of erosion from water and wind. In turn, soil moisture available for plant growth is diminished, precipitation runoff increases in volume and velocity, and soil erosion accelerates, which leads to surface changes, including the formation of rills, gullies, terracettes, and pedestals (Webb and others, 1978; Iverson and others, 1981; Webb, 1982; Hinckley and others, 1983; Wilshire, 1983b). The extent of soil compaction may be measured in terms of soil bulk density, soil strength, and/or permeability. Soil bulk density, calculated as oven-dried soil weight per unit of volume, is typically expressed as g/cm^3 or g/cc. Soil strength, measured as the soil's resistance to deforming forces—or the amount of energy required to break apart aggregates or move implements through the soil—is typically expressed as kg/cm^2 or pounds per square inch (PSI). Soil permeability is the rate at which water (or air) infiltrates the soil, expressed as cm/hr or inches/hr (Leung and Meyer, 2004). Generally, soil bulk density and strength increase with increasing compaction, whereas permeability decreases with increasing compaction (Adams and others, 1982; Webb, 1982; Cole, 1990).

Important factors affecting a soil's susceptibility to compaction include its (1) texture (relative proportions of sand, silt, and clay); (2) structure (the grouping of sand, silt, and clay particles into aggregates), including its porosity (a measure of pore space, which affects the amount of air or water a soil can hold) and aggregate stability (the ability of soil aggregates to resist disruption from outside forces—water in particular); (3) type (series) and depth; and (4) antecedent moisture (the soil's water content prior to compaction). Sandy or clayey soils relatively uniform in texture and structure are less vulnerable to compaction than loamy sands or coarse-textured, gravelly soils characterized by variability in particle size (Lovich and Bainbridge, 1999). In addition, soils with greater water content are more susceptible to compaction than those containing less moisture (Webb, 1982), although even in semi-arid and arid lands soil compaction is problematic because the texture of these soils is slow to recover (Webb, 1982) through natural soil-loosening processes (including shrinking, swelling, drying, wetting, freezing, and thawing).

As the number of vehicle "passes" (one pass is the equivalent of one OHV passing over a given area one time) increases, soil bulk density and soil strength increase and permeability (as indicated by water infiltration rate) decreases (Lovich and Bainbridge, 1999). Soil compaction may become evident after only a few vehicle passes. In fact, Iverson and others (1981) found that soil bulk density increased logarithmically with the number of vehicle passes. Similarly, Adams and others (1982) report that soil strength on routes subjected to a single vehicle pass was 5.3 to 28.4 kg/cm^2 (75.366 to 403.848 PSI) greater (depending on the percent soil moisture) than that of nearby undisturbed soils; after 10 to 20 passes, soil strength was too great (impenetrable) to measure with a penetrometer, indicating that a few passes were enough to cause soil "cementation." After initial disturbance, the effects of soil compaction can persist for years, even centuries, before natural soil-loosening processes can restore the soil's texture (Webb and Wilshire, 1980; Webb, 1982; Froehlich and others, 1985; Prose, 1985; Lovich and Bainbridge, 1999). For example, one year after impact, a one-pass trail was still faintly visible, as indicated by slightly more surface gravel and growth of annual plants (the first to grow in disturbed sites) than on surrounding land, and trails impacted by 100 and 200 passes had notable side berms (Prose, 1985).

Other effects of soil compaction include changes to soil horizons and increased compaction in deeper strata. The OHV traffic associated with the annual Johnson Valley-Parker OHV race (1980-1983) near Joshua Tree National Park on the Colorado River compacted 2 to 5 cm (0.8 to 2.0 in) of the underlying vesicular soil horizon (composed of fined-grained, wind-blown material occurring about 20 cm [7.9 in] deep near surface soil horizons, often immediately under desert pavement, and characterized by small pores, or vesicles, of air space; typical of arid regions) and caused excavation (mechanical erosion) of the A and B soil horizons to depths of 20 cm (7.9 in) (Wilshire, 1983a). Prose (1985) found that resistance (to a penetrometer) of soil affected by military maneuvers (including tanks, tracked equipment and personnel carriers, and support vehicles) was 50 percent greater than that of undisturbed soils. Overall, traffic typically causes significant changes to soils, which may take years, if not decades, to recover.

2.2.3 Effects on Soil Stabilizers and Rates of Soil Erosion

A significant effect of soil compaction is the soil's inability to support vegetation after disturbance, thus increasing its susceptibility to erosion (Webb and others, 1978). Soil erosion resulting from soil compaction is caused by two main factors (Hinckley and others, 1983): reduced infiltration rates and destruction of soil stabilizers. Infiltration of water into soils depends, to a large extent, on the soil's porosity, which is reduced by compaction. Soil stabilizers, which are characteristic of undisturbed desert substrates, may include cryptobiotic crusts of lichen, fungi, bacteria, mosses, and/or algae; chemical or mechanical crusts (thin upper coating of clay particles oriented parallel to the surface); and desert pavements (closely packed, interlocking fragments of pebble- and/or cobble-sized rocks from which fine-grained materials have been removed by wind or water erosion) (Lovich and Bainbridge, 1999). Cryptobiotic organisms facilitate accumulation of organic materials and nutrients, including nitrogen and carbon, thereby increasing soil fertility (Johansen, 1993). Since they occur in the soil's upper layer, they also promote water infiltration and enhance retention of soil moisture (Belnap and Gardner, 1993). Their proximity to the surface, however, makes them susceptible to destruction by vehicular and foot traffic.

Cole (1990) documented destruction of cryptogamic soil crusts after only 15 passes by hikers wearing lug-soled boots. Traffic from the Johnson Valley-Parker OHV race mentioned in section 2.2.2 not only destroyed the vesicular soil horizon, it destroyed the overlying desert pavement (Wilshire, 1983a). Webb (1982), who evaluated soil surfaces (shape; another measure of soil compaction) after 1, 10, 100, and 200 motorcycle passes, found changes occurring after the first few passes, although the effects of subsequent passes were more severe due to their cumulative effects: routes subjected to 100 and 200 passes were characterized by berms and lateral edges, and route midlines were 10-30 mm below the level of surrounding undisturbed ground. Once damaged or destroyed, it may take 300-500 years per inch for soil stabilizers to recover or return to their original state (Hudson, 1971).

Typically, undisturbed soil surfaces are very important in controlling the soil's response to precipitation runoff, particularly where the soil surface is covered with fine gravel that overlays soils with large pores (Webb, 1982). In the Mojave Desert, surface runoff was typically five times greater and sediment yield (in runoff) was 10-20 times greater in OHV-impacted areas than in undisturbed areas (Iverson and others, 1981). For various reasons, certain portions of the desert, including dunes, playas, and areas covered with coarse surface material, are fairly resistant to erosion from runoff (Hinckley and others, 1983), whereas vulnerable areas are those where initial infiltration rates are low, slopes are high, and ratios of surface sand/gravel to smaller particles are low (Iverson and others, 1981). The character of precipitation also influences the susceptibility of denuded soil to erosion; erosion rates are typically greater when rainfall events are of long duration and high intensity (Iverson and others, 1981). Disturbed soils also increase the likelihood of debris eroding from areas disturbed by OHV activities (Lovich and Bainbridge, 1999). Indeed, debris flow has been documented to bury plants growing outside the area impacted (Nakata, 1983).

2.2.4 Annotated Bibliography for OHV Effects on Soils and Watersheds

Adams, J.A., Endo, A.S., Stolzy, L.H., Rowlands, P.G., and Johnson, H.B., 1982, Controlled experiments on soil compaction produced by off-road vehicles in the Mojave Desert, California: Journal of Applied Ecology, v. 19, no. 1, p. 167–175.

Under controlled conditions, soil crust properties were measured to determine how rapidly they were altered by passing vehicles. Routes impacted by a single vehicle pass had soil strengths 5.3 to 28.4 kg/cm^2 (75.366 to 403.848 lb/in^2 [or PSI], depending on the percent soil moisture) greater than undisturbed soil, indicating that just a single pass can begin to affect soil strength. Mean soil strength on routes exposed to 10 and 20 passes was too high to measure. Drying caused the soil in the slightly compacted track to become much harder (increased soil strength) than the undisturbed soil.

Belnap, Jayne, 1993, Recovery rates of cryptobiotic crusts—inoculant use and assessment methods: Great Basin Naturalist, v. 53, no. 1, p. 89–95.

Rates of recovery of cyanobacterial-lichen soil crusts from disturbance were examined. Plots were either undisturbed or scalped, and scalped plots were either inoculated with surrounding biological crust material or left to recover naturally. Natural recovery rates were found to be very slow. Inoculation significantly hastened recovery of the cyanobacterial/green algal component, lichen cover, lichen species richness, and moss cover; even with inoculation, however, lichen and moss recovery was minimal.

Belnap, Jayne, 2002, Impacts of off-road vehicles on nitrogen cycles in biological soil crusts—resistance in different U.S. deserts: Journal of Arid Environments, v. 52, no. 2, p. 155–165.

This study was conducted to evaluate short-term impacts of OHVs on lichen cover and the nitrogenase activity (NA) of biological soil crusts on various soil types in the Great Basin, Colorado Plateau, Sonoran, Chihuahuan, and Mojave deserts. Lichen cover was significantly correlated with percent silt in soil (and negatively correlated with percent sand and clay). Disturbance reduced NA at all 26 sites, but significantly at 12; declines were greatest in soils of cooler regions than hotter ones, possibly indicating that non-heterocystic cyanobacterial species are more susceptible to disturbance than heterocystic species. Sandy soils showed greater reduction of NA as sand content increased, while fine-textured soils showed a greater decline as sand content increased. At all sites, higher NA before the disturbance resulted in less impact to NA post-disturbance. These results may be useful in predicting the impacts of off-road vehicles in different regions and different soils.

Cole, D.N., 1990, Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park: Great Basin Naturalist, v. 50, no. 4, p. 321–325.

Under controlled conditions, cryptogamic soil crusts in Grand Canyon National Park were trampled by hikers to determine how rapidly they were pulverized and how rapidly they recovered. Only 15 passes were required to destroy the structure of the crusts; visual evidence of bacteria and cryptogam cover was reduced to near zero after 50 passes. It took soil crusts one to three years to redevelop, and after 5 years the extensive bacteria and cryptogam cover left little visual evidence of disturbance. Surface irregularity remained low after 5 years, however, suggesting that recovery was incomplete.

Eckert, R.E.J., Wood, M.K., Blackburn, W.H., and Peterson, F.F., 1979, Impacts of off-road vehicles on infiltration and sediment production of two desert soils: Journal of Range Management, v. 32, no. 5, p. 394–397.

This project staged a series of controlled motorcycle and 4-wheel drive vehicle passes, followed by simulated rainfall. Two sites were chosen to represent two different soil types. Infiltration rates were lower and sediment yield was higher after soil was disturbed by vehicular traffic. High sediment yield was attributed to reduced infiltration after 10 minutes; the remaining 20 minutes of the test period were characterized by particles being carried away in runoff water.

Folz, R.B., 2006, Erosion from all terrain vehicle (ATV) trails on National Forest lands (abs.): Proceedings of the 2006 American Society of Agricultural and Biological Engineers, Portland, Oregon, July 9–12, 2006: St. Joseph, Michigan, 2006 American Society of Agricultural and Biological Engineers, *http://asae.frymulti.com/abstract.asp?aid=21056&t=2*.

Concern about unmanaged use of all terrain vehicles (ATV) on U.S. Forest Service lands prompted an experimental study to test the relative effects of low-, medium-, and high-disturbance trails (based on traffic levels), as measured by reduced litter/vegetation and the width and wheel-rut depth of trails. Trail condition was assessed and then subjected to simulated rainfall. A negative relationship between levels of ATV traffic and rainfall infiltration was not statistically significant among disturbance levels; however, there were significant differences in infiltration and measures of erosion between undisturbed and disturbed conditions. Data from this study will be used to estimate ATV traffic-induced erosion and make decisions regarding management of ATV use.

Iverson, R.M., Hinckley, B.S., and Webb, R.M., 1981, Physical effects of vehicular disturbances on arid landscapes: Science, v. 212, no. 4497, p. 915–917.

In 50 rainfall simulation tests, vehicle-use plots had about five times more runoff and 10-20 times greater sediment yield than adjacent unused plots. In a desert environment, such effects may occur even when use of off-road vehicles is light. Recovery times from vehicular traffic were estimated to be nearly 100 years. Erosion rates were calculated from multivariate statistical analyses using 22 experimental factors. The character of the rainfall was identified as the most important variable in predicting increases in erosion.

Sparrow, S.D., Wooding, F.J., and Whiting, E.H., 1978, Effects of off-road vehicle traffic on soils and vegetation in the Denali Highway region of Alaska: Journal of Soil and Water Conservation, v. 33, no. 1, p. 20–27.

This study examined the effects of vehicles on trails. The surface layer of living material was killed on all main trails, although soil morphology was not generally altered except in the surface horizon. Varying amounts of organic matter were lost from the heavily used trails,

depending on slope and vehicle type. Soil depth and drainage were the most important factors influencing the condition of the trail. The greatest effects on soils occurred in poorly drained areas or on loose, gravel-free soils that were highly susceptible to erosion.

Tuttle, M., and Griggs, G., 1985, Accelerated soil erosion at three State Vehicular Recreation Areas: central and southern California, *in* Erosion control: A challenge in our time, proceedings of the 16th annual International Erosion Control Association, February 21–22, 1985, San Francisco, California: San Francisco, California, International Erosion Control Association, p. 105-115.

Soil erosion rates were evaluated at three State Vehicular Recreation Areas, with a particular focus on hillclimbs. The key factors contributing to erosion rates were slope, length of climb, soil type, and weather. Based on monitoring and catchment basin yield, erosion in open areas dedicated to OHV use was 10 to 25 times greater than in undisturbed areas.

Webb, R.H., 1982, Off-road motorcycle effects on a desert soil: Environmental Conservation, v. 9, no. 3, p. 197–208.

The effects of controlled motorcycle traffic on a Mojave Desert soil in California were studied in order to quantify soil compaction. Four experimental trails treated with 1, 10, 100, and 200 passes with an off-road motorcycle were established in loamy sand at 6.2 percent (by weight) moisture content. Soil penetration resistance, bulk density, infiltration rate, and response to rainfall were measured for undisturbed soil and the experimental trails immediately after the impact, and soil cores were measured in the laboratory to determine pore-size distributions. Soil bulk density was remeasured one year after the impact to ascertain the amount of recovery. The 1-pass trail had a slight surface indentation with knob imprints from the tires. Along the 100- and 200-pass trails, there were berms and lateral edges, and their centers were 10–30 mm below the level of undisturbed soil adjacent to the trail.

Wilshire, H.G., and Nakata, J.K., 1976, Off-road vehicle effects on California's Mojave Desert: California Geology, June 1976, p. 123–132.

This study was designed to evaluate long-term effects of an off-road vehicle race on the desert landscape, in particular the landscape condition after vehicular use (specifically motorcycles), whether or not effects were confined to the areas of direct impact, and how long the physical effects of such activities remained. Visual observations and penetrometer measurements were recorded in five ground types. Soil compaction was the dominant consequence of motorcycle use: penetrometer data revealed decreases in mean penetration depths. Combined with a notable reduction in plant cover, soil compaction significantly increased the potential for erosion. Initial vehicle impact resulted in substantial, immediate mechanical erosion, followed by wind erosion, culminating in the increased potential for water erosion over longer periods of time.

Wilshire, H.G., Nakata, J.K., Shipley, Susan, and Prestegaard, Karen, 1978, Impacts of vehicles on natural terrain at seven sites in the San Francisco Bay area: Environmental Geology, v. 2, no. 5, p. 295–319.

Vegetation and soil properties were measured at seven sites exposed to off-road vehicle activities. Impacts on loamy soils included greater soil surface strength and bulk density, lower infiltration rates and soil moisture, extended diurnal temperature ranges, and reduced organic

carbon. These effects, combined with the associated loss in vegetative cover, promoted erosion, the rates of which significantly exceeded Federal and local standards, and the increased sediment yield and runoff caused adverse effects on neighboring properties.

2.3 OHV Effects on Vegetation

2.3.1 Section Summary

Relative to plant communities in OHV-impacted areas, those in undisturbed sites are dominated by native plants, invasive species are not increasing, plant growth and reproduction are vigorous, age-classy structures are appropriate to the species, and canopy cover and vertical structure are adequate for dispersing the energy of precipitation runoff and promoting water infiltration. Direct impacts of OHV activities on vegetation include reduced vegetation cover and growth rates, and increased potential for non-native grasses and pioneering species to become established, thus altering vegetation communities. In certain instances, however, the impervious nature of compacted route and paved road surfaces could result in significant runoff that generates greater moisture availability immediately along OHV routes. In turn, this would promote increased vegetation cover and plant abundance than one might find in surrounding areas farther away from OHV routes.

Some important indirect effects of OHV activities on vegetation are tied to soil properties altered by OHV traffic, as soil properties typically influence vegetation growth. OHV roads and trails also create edge habitats, which can generate conditions that promote the encroachment of non-native and invasive plant species. Other indirect effects include increased amounts of airborne pollutants and dust raised by OHV traffic. A blanket of fugitive dust on plant foliage can inhibit plant growth rate, size, and survivorship.

2.3.2 Overall Effects on Vegetation Cover and Community Composition

When soils are severely disturbed, vegetation cover can be reduced significantly (Adams and others, 1982; Prose and others, 1987; Bolling and Walker, 2000) and growth can be impaired (Spencer and Port, 1988; Angold, 1997). As stated in the previous section, even a few passes by vehicles can cause significant changes in soil properties. Adams and others (1982) found reduced cover of desert annuals in tracks created by as few as 1 (on wet loamy sand) to 20 (on dry loamy sand) vehicle passes; the reduction in cover, however, was not due to fewer plants, but to smaller plant sizes. Similarly, Bolling and Walker (2000) found that in OHV routes there were many small individuals of creosote bush (*Larrea tridentata*), but larger plants were few or absent; in control plots, however, there were more large plants and fewer small ones.

Reduced plant sizes are typical where the extent of soil compaction inhibits their roots from penetrating to deeper soil levels. In fact, Adams and others (1982) determined that root growth is precluded at soil strengths of about 20 kg/cm^2 (284.4 lb/in^2). Within tracks made by 1, 3, 10, and 20 vehicle passes, Adams and others (1982) found that annuals with large taproots (for example, pincushion flower [*Chaenactis fremontii*]) decreased, whereas there was significantly greater cover of common Mediterranean grass (*Schismus barbatus*), a non-native grass with a fibrous root system. The fibrous root system of plants that characterized by single cotyledons, such as common Mediterranean grass, allows for easier germination and root growth than is possible for taprooted dicotyledons.

Soil compaction also increases the potential for invasive, non-native annuals and other early successional plants to establish rapidly in OHV routes, whereas native perennials may require at least 5 years to become established (Adams and others, 1982; Prose and others, 1987;

Lovich and Bainbridge, 1999). This is due, in part, to the increased surface moisture availability within the tracks of OHV routes after compaction has reduced the rate of water infiltration, which may favor the rapid germination and growth of non-native and invasive annuals (Adams and others, 1982). In disturbed areas, pioneering species, such as burrobush (*Ambrosia dumosa* and *Hymenoclea salsola*—now *Ambrosia salsola*; see *http://ucjeps.berkeley.edu/cgi-bin/get_cpn.pl?3578*), often dominate the plant community and typically their percent cover is similar to, or greater than, that of undisturbed areas (Prose and others, 1987). Davidson and Fox (1974) also found that non-native, early-successional species, such as redstem stork's bill (*Erodium cicutarium*) and common Mediterranean grass were common at sites disturbed by OHVs. When comparing vegetation in disturbed versus protected plots, Brooks (1995) found that common Mediterranean grass was the only species with greater biomass in the disturbed plots.

OHV traffic also causes direct impacts to vegetation structures (breakage, smashing), although population-level effects may be difficult to discern in the short term. Overall, the extent of immediate effects increases with the frequency of OHV passes. For example, Webb (1983) found that after a single pass, annual plants on an OHV route remained intact, but most were destroyed after 10 passes. Likewise, a series of studies to evaluate the impacts of OHV traffic on the Federally listed Peirson's milkvetch (*Astragalus magdalenae peirsonii*) indicated that this plant was more likely to occur at sites closed to OHV activity than at OHV sites that have been rested from OHV activity (Groom and others, 2005); however, additional study indicated that the number of reproducing plants (and the seedbank) was adequate to maintain the milkvetch population (Phillips and Kennedy, 2006; for more reports, go to *http://www.fws.gov/carlsbad/PMV_Docs.htm*). It remains unclear, however, whether research conducted over longer time scales would yield different results.

2.3.3 Edge Effects Along OHV Routes

Roads and trails also create edge habitats (Johnson and others, 1975; Vasek and others, 1975; Adams and Geis, 1983; Andrews, 1990; Holzapfel and Schmidt, 1990; Lightfoot and Whitford, 1991; Reed and others, 1996), resulting in a variety of effects, including changes in vegetation and encroachment of non-native and invasive species (Huey, 1941; Lovich and Bainbridge, 1999). As mentioned in section 2.3.2, the impermeable surfaces of roads and OHV routes shed precipitation, thereby increasing overall moisture availability in the immediate vicinity of the road or route. Additionally, the coarse-textured soils typically found in association with paved roads (roadbed materials laid down prior to paving) permit good water infiltration along road edges (Hillel and Tadmor, 1962); similar conditions may occur along improved gravel routes. The increased moisture availability may promote greater plant vigor along roadsides than in surrounding areas (Johnson and others, 1975), and Angold (1997) indicated that such effects may extend as far as 200 m from road edges. Indeed, several studies have shown that there can be more vegetation cover along roadsides and right-of-ways than in adjacent areas (Johnson and others, 1975; Vasek and others, 1975; Holzapfel and Schmidt, 1990; Lightfoot and Whitford, 1991). Perennial shrubs, in particular, may grow larger and attain greater vigor and density along road edges (Johnson and others, 1975; Lightfoot and Whitford, 1991). Likewise, Johnson and others (1975) found that the standing crop (a measure of primary productivity) was 6 times greater along unpaved roads (17 times greater along paved roads) than it was in nearby undisturbed areas.

The greater vegetation cover typically observed along roadsides also is often due, in part, to greater species richness in those areas (Holzapfel and Schmidt, 1990); however, much of this diversity may be represented by non-native species easily dispersed along roads and trails

(Wilcox, 1989; Tyser and Worley, 1992; Parendes and Jones, 2000). Furthermore, local-scale increases in species richness can be associated with decreases in species richness at the landscape scale, thus creating a relatively impoverished and anthropogenic vegetation community (Holzapfel and Schmidt, 1990). Interestingly, increased vegetation cover along roadsides may attract more invertebrates and other organisms. For example, Lightfoot and Whitford (1991) found that shrubs along a road supported greater numbers of foliage arthropods. What is not clear, however, is whether high densities of animals in roadside habitats represent improved conditions for native fauna or dominance by invasive and/or non-native organisms. Furthermore, high densities do not necessarily represent population sources (that is, where survivorship and productivity are high enough to contribute to the species' overall population); instead, high densities can indicate poor-quality habitat into which subordinate animals may crowd and experience poor survivorship if they cannot find or defend better habitat (population sinks). In other words, density can be a misleading indicator of habitat quality (Van Horne, 1983). It is important to note, however, that the greater vegetation cover along roadsides compared to plots away from roads may be a phenomenon found only in arid environments (Hillel and Tadmor, 1962; Holzapfel and Schmidt, 1990).

Fugitive dust raised by OHV traffic also affects vegetation in the vicinity of roads. Along Alaskan roads heavily traveled by various types of vehicles, Walker and Everett (1987) found significant dust impacts up to 10 m (10.9 yd) from the roadside and dust blankets up to 10 cm (3.9 in) thick on mosses and other vegetation of low stature. Several morphological factors contribute to plant susceptibility to heavy dust loads, including mat or prostrate growth form, lack of a protective stem cortex or leaf cuticle, and intricate branching or closely spaced leaves that tend to trap dust (Walker and Everett, 1987; Spellerberg and Morrison, 1998). Processes that may be affected by dust include photosynthesis, respiration, and transpiration due to blocked stomata and cell destruction (Spellerberg and Morrison, 1998), all of which could result in reduced plant growth, size, productivity, and/or survivorship.

2.3.4 Annotated Bibliography for OHV Effects on Vegetation

Adams, J.A., Stolzy, L.H., Endo, A.S., Rowlands, P.G., and Johnson, H.B., 1982, Desert soil compaction reduces annual plant cover: California Agriculture, v. 36, no. 9–10, p. 6–7.

Soil crust properties and associated changes in vegetation composition were measured under controlled conditions over two 6-month wet seasons to determine how rapidly they were altered by vehicle passes. Reductions in annual plant cover occurred in tracks created by as few as 1 (on wet loamy sand) to 20 vehicle passes (on dry loamy sand). This cover reduction, however, was not due to fewer plants; rather, the plants were smaller, and their size depended on the duration of drying periods, during which the soil strength intensified in impact/track areas. Cover of annuals with large taproots (for example, *Chaenactis fremontii*) decreased in vehicle tracks, whereas the cover of *Schismus barbatus*, a grass with a fibrous root system, was significantly greater in tracks generated by 1, 3, 10, and 20 vehicle passes. It was determined that root growth for plants stops at soil strengths of about 20 kg/cm^2 (284.4 lb/in^2 [or PSI]). Soil disturbance also increased the potential for grasses and pioneering annual species to become established, whereas perennial species would take at least 5 years to return. A possible reason for this may be greater water availability in the track.

Angold, P.G., 1997, The impact of a road upon adjacent heathland vegetation—Effects on plant species composition: Journal of Applied Ecology, v. 34, no. 2, p. 409–417.

The effect of a road on heathland vegetation was investigated at five sites adjacent to the main trunk road through the New Forest, Hampshire, United Kingdom, and nine supplementary sites adjacent to five minor roads. There was enhanced growth of vascular plants near the road, notably heather and grasses, which was probably due to nitrogen oxides from vehicle emissions. There was a decrease in the abundance and health of lichens near the road. There was an increase in the abundance of grasses in the heathland near roads, which may be due to the changes in relative competitive ability of plant species under conditions of eutrophication. The extent of the edge effect in the heath was closely correlated with traffic intensity, with a maximum edge effect of 200 m adjacent to a dual carriageway.

Benninger-Traux, M., Vankat, J.L., and Schaefer, R.L., 1992, Trail corridors as habitat and conduits for movement of plant species in Rocky Mountain National Park, Colorado, USA: Landscape Ecology, v. 6, no. 4, p. 269–278.

Ground-layer vegetation was sampled along selected trail corridors to determine whether corridors provide habitat for certain species and serve as conduits for species dispersal. Patterns of plant species composition were analyzed in relation to distance from trail edge, level of trail use, and distance from trailheads, junctions, and campgrounds. Species composition was significantly affected by distance from trail edge and level of trail use, as species were favored or inhibited by the corridor, depending upon their growth habits. Species composition also was affected by distance from trailheads. These findings, along with the presence of exotic species, indicate that trail corridors in Rocky Mountain National Park function as habitat and conduits for dispersal of plant species.

Bolling, J.D., and Walker, L.R., 2000, Plant and soil recovery along a series of abandoned desert roads: Journal of Arid Environments, v. 46, no. 11, p. 1–24.

To elucidate factors controlling desert succession, soil and vegetation dynamics were examined along roads abandoned for 5, 10, 21, 31, 55 and 88 years in southern Nevada. None of the measured soil or vegetation parameters varied significantly with road age. Differences were found, however, between soils and vegetation on roads compared to those on nearby control sites, and soils differed between roads created by surface vehicular traffic and those made by bulldozing. Studies of recovery following disturbance in deserts must take into account natural patterns of plant and soil heterogeneity and initial disturbance type.

Brooks, M.L., 1995, Benefits of protective fencing to plant and rodent communities of the western Mojave Desert, California: Environmental Management, v. 19, no. 1, p. 65–74.

This paper documents the response of plant and small mammal populations to fencing constructed to preclude OHV activities between 1978 and 1979 at the Desert Tortoise Research Natural Area, Kern County, California. Aboveground live annual plant biomass was generally greater inside than outside the fenced plots during April 1990, 1991, and 1992. The non-native grass, *Schismus barbatus*, was a notable exception, producing more biomass in the unprotected area. Forb biomass was greater than that of non-native annual grasses inside the fence during all 3 years of the study. Outside the fence, forb biomass was significantly greater than that of non-native grasses only during spring 1992. Percent cover of perennial shrubs was greater inside the fence than outside, while no significant trend in density was detected. There was also more seed biomass inside the fence, which may have contributed to the greater species diversity and density of Merriam's kangaroo rats (*Dipodomys merriami*), long-tailed pocket mice (*Chaetodipus* *formosus*), and southern grasshopper mice (*Onychomys torridus*) in the protected area. These results show that protection from OHV disturbance has many benefits, including greater overall community biomass and diversity.

Holzapfel, C., and Schmidt, W., 1990, Roadside vegetation along transects in the Judean Desert: Israel Journal of Botany, v. 39, p. 263–270.

Vegetation was studied on comparable plots along roadsides and in the surrounding area. The uniqueness of roadside vegetation was shown using indices and measurements that allowed comparison along a climatic gradient. Near roads, biomass and species diversity were notably greater than in surrounding areas, and the chorological composition was different, at least under arid conditions. The reasons for these differences are discussed based on investigations of site conditions. Increased water runoff and more favorable soil conditions seem to have had important influences on the vegetation community.

Kutiel, P., Eden, E., and Zhevelev, Y., 2000, Effect of experimental trampling and off-road motorcycle traffic on soil and vegetation of stabilized coastal dunes, Israel: Environmental Conservation, v. 27, no. 1, p. 14–23.

The aim of this study was to assess the response of soil and annual plants of stabilized Mediterranean coastal dunes in Israel to various intensities of short-duration pedestrian and motorcycle traffic. Experimental procedures entailed 0, 20, 50, 100, and 200 straight and 150 turn motorcycle passes. The response of annual plants was assessed by measuring ground cover, height, and species richness and diversity, and soil response was assessed by measuring penetrable depth, organic matter, and moisture content. Motorcycle passage had an immediate significant impact on annual plants at all traffic intensities. The maximum effect on plants was observed in the wheel tracks and in the turn lanes. Mean annual ground cover and height were less sensitive measures than species richness and diversity for determining the overall impact of motorcycles on the area.

Prose, D.V., Metzger, S.K., and Wilshire, H.G., 1987, Effects of substrate disturbance on secondary plant succession—Mojave Desert, California: Journal of Applied Ecology, v. 24, no. 1, p. 305–313.

The effects of substrate disturbance on perennial plant succession in the Mojave Desert were assessed at three military camps abandoned for 40 years. Soil compaction, removal of the top layer of soil, and altered drainage channel density caused significant changes in perennial plant cover, density, and relative species composition. Long-lived species, predominantly *Larrea tridentata*, were dominant in all control areas, but percent cover and density were greatly reduced in areas where substrate alterations were significant. At one camp where substrate alterations were insignificant in disturbed areas, *Larrea* was the dominant species (as it was in the control areas).

Schultink, G., 1977, Impact analysis of off-road-vehicle use on vegetation in the Grand Mere Dune environment: East Lansing, Michigan, Michigan State University, Report no. NASACR155764, 10 p.

A linear regression of percent unvegetated land in OHV-impacted areas versus time for two sample areas indicated that the areas underwent average declines of 1.9 and 5.9 percent per year in vegetation cover. Two factors were assumed to play roles in the difference: the difference in accessibility and the extent of vegetation fragmented during the first year of the study (one sample area was located closer to potential access points and was more fragmented initially).

Wilshire, H.G., Shipley, Susan, and Nakata, J.K., 1978, Impacts of off-road vehicles on vegetation: Transactions of the North American Wildlife and Natural Resources Conference, v. 43, p. 131–139.

Observations of the impacts of off-road vehicles on soils and vegetation were made at more than 400 sites in seven western states during 3 years. This type of land use had both direct and indirect effects on vegetation. Direct effects included crushing and uprooting plants. Indirect effects included modification of the soil, which affected plants beyond the areas directly impacted by vehicles, and restoration of the plant cover was inhibited. This paper covers the erosional effects on vegetation, depositional effects on vegetation, and the effects of physical and chemical modification of remnant soils on revegetation.

2.4 OHV Effects on Wildlife and Habitats: Native, Threatened, and Endangered Species

2.4.1 Section Summary

The impacts of OHV activities on wildlife and their habitats are numerous and well documented. Networks of roads and trails fragment habitat, reduce patch size, and increase the ratio of edge to interior. This may have serious consequences for area-sensitive species (those that cannot carry out certain aspects of their life cycles without large blocks of habitat or corridors linking habitat patches), predator-prey relationships, and overall population dynamics. In particular, fragmentation and edges created by OHV routes may have strong effects on animal movement patterns. Precluding or inhibiting animal movements effectively diminishes dispersal to and recolonization in other areas, thus increasing the likelihood of local extirpations. Overall, studies demonstrate that even narrow roads (paved and unpaved) and trails can represent significant barriers to the movements of animals. Reluctance to cross even narrow trails similar in width to routes created by OHV travel may alter or preclude the movements of various species. The cumulative effects of OHV-route networks proliferating across the landscape may have serious ecological consequences for species reluctant to cross OHV routes. Where threatened and endangered species are at risk, understanding their particular responses to roads of varying types, widths, use intensities, and habitat contexts is crucial.

OHV routes also generate conditions unlikely to occur in environments unaffected by OHV activity; in turn, these conditions can facilitate range extensions and invasions of non-native and/or opportunistic species. In addition, OHVs can contribute directly to mortality (and possible population declines) of wildlife species through collisions with vehicles, nest destruction, and collapsing burrows. Noise generated by OHVs also has been found to cause inner ear bleeding. In particular, noise may alter animal behaviors, breeding populations, the abilities of some species to detect predators (through auditory cues), and it can stimulate estivating animals to emerge from their underground burrows at inappropriate times. These factors may result in diminished body mass, reduced productivity, and/or poor survivorship.

2.4.2 Loss of Habitat Connectivity: Fragmentation and Barrier Effects

Creating roads and trails (of any kind) diminishes habitat connectivity, increases the proportion of edge to interior habitat, and decreases patch size of habitats (Reed and others, 1996; Forman and others, 2003). In fact, roads, including OHV routes, represent a principal factor contributing to habitat fragmentation at various scales (Meffe and Carroll, 1997). Furthermore, both paved roads and OHV routes—ranging from 4-lane paved highways to two-track routes less than 3 m (3.3 yards) wide—that separate once-continuous habitat can disrupt the movement and dispersal of many wildlife species between and within habitats (Swihart and Slade, 1984; Brody and Pelton, 1989; Yanes and others, 1995; Lovallo and Anderson, 1996; Clevenger, 1998; Forman and Alexander, 1998; Jackson and Griffen, 1998). In turn, these effects can have consequences for area-sensitive species and may encourage non-native and/or invasive species. Special-status wildlife species known to occur on BLM lands and whose long-term persistence is threatened by habitat fragmentation and diminished habitat connectivity include grizzly bear (*Ursus arctos horribilis*; Gibeau and Herrero, 1998; Servheen and others, 1998), black bear (*Ursus americanus*; Brody and Pelton, 1989), gray wolf (*Canis lupus*; Paquet and Callahan, 1996), mountain lion (*Felis concolor*; Beier, 1993), lynx (*Felis lynx*; Ruediger, 1998), ocelot (*Leopardus pardalis*; Tewes and Blanton, 1998), and desert tortoise (*Gopherus agassizii;* Boarman and Sazaki, 1996). The resulting isolation of subpopulations (Dobson and others, 1999) can promote increased inbreeding and a lack of genetic exchange with other subpopulations, ultimately leading to declines in the genetic diversity required for adaptation to variable conditions and possible founder effects (Hanski and Simberloff, 1997; Hanski, 1999). Another consequence of subpopulation isolation is the reduced potential for recolonization when extirpations occur as a result of localized population fluctuations and catastrophic events (Yanes and others, 1995).

Until recently, only wide, multi-lane, paved roads have been considered significant barriers to animal movements. More recent lines of evidence from fragmentation studies, however, indicate that the ability or willingness of an animal to cross a given road type varies widely by species (Brody and Pelton, 1989; Lovallo and Anderson, 1996). For example, rodents in a desert habitat were found to avoid crossing a 4-lane highway, although they lived alongside the road in the right-of-way vegetation (Garland and Bradley, 1984). Likewise, in forested habitats divided highways wider than 90 m (98.4 yd) served as total barriers to dispersal by small forest mammals (Oxley and others, 1974). However, improved gravel roads have been found to inhibit crossings by mountain lions (*Puma concolor*; van Dyke and others, 1986), and even infrequently traveled, single-lane dirt roads have been found to alter movements by some species (Andrews, 1990). For example, Swihart and Slade (1984) report that prairie voles (*Microtus ochrogaster*) and cotton rats (*Sigmodon hispidus*) were strongly inhibited from crossing a route less than 3 m (3.3 yd)wide and composed of two dirt tracks created by the passing of 10 to 20 vehicles per day. Oxley and others (1974) evaluated small mammal responses to roads and routes ranging from 4-lane paved highways to country gravel roads in forested systems of southeastern Canada and found that they were not willing to cross roads or other routes with a total clearance (the distance between forest margins, including road surfaces and immediately adjacent strips of vegetation kept very short via spraying and/or mowing) of 30 m (32.8 yd) or greater; road surface apparently was unimportant. Likewise in Germany, forest mice (*Apodemus flavicollis*) did not cross roads 6 m (6.6 yd) wide, and very few mice returned to the side of the road from which they were captured after being translocated to the opposite side within the same habitat type (Mader, 1984). Areas characterized by high densities of roads also are characterized by low probabilities that amphibian species will occupy breeding pools (Vos and Chardon, 1998), most likely because the edges were relatively impermeable (whether due to behavioral avoidance or direct mortality) to critical amphibian movements (dispersal, seasonal movements; Gibbs, 1998). On the other hand, some small mammals are known to cross paved and gravel roads (Bakowski and Kozakiewicz, 1988), particularly where vegetated highway right-of-ways resemble those of adjacent habitats (Wilkins, 1982). These studies indicate that road surface type is not always the critical inhibiting factor; however, it does influence traffic speed, which can directly affect mortality rates (Oxley and others, 1974; Bakowski and Kozakiewicz, 1988).

Invertebrates also may be precluded from crossing various road types, including those considered relatively narrow; again, however, there are species differences that may be influenced by their ecologies and physical capabilities. For example, Samways (1989) found that both "tarred" (paved) and "untarred" roads were almost complete or partial barriers to three species of bush crickets (*Decticus varrucivorus monspeliensis* and *Platycleis fedtschenkoi azami*, both wingless, and *P. tessellate*, the flight range of which is less than [<] 5m [5.5 yd]), but roads were only minor, very minor, or did not serve as barriers to the movements of six other bush cricket species, five of which can readily fly across roads (flight ranges from <30 to 150 m [32.8–164.0 yd]. On the other hand, Munguira and Thomas (1992) found that wide highways did not affect the movements of butterflies in open populations; movements of butterflies in closed populations, however, were slightly impeded by roads. Other butterfly species may not even attempt to fly across roads (described by authors as two-lane highways and secondary roads), possibly due to the extreme changes in microclimate over roads (including columns of warm air rising above roads; Boer Leffef, 1958, as interpreted and translated by van der Zande, 1980). Mader (1984) reported that in a five-year mark-recapture-release study involving 10,186 carabid beetles representing nine species, three species were never recaptured on the opposite side of study area roads (one- or two-lane paved roads) or parking loops, and the remainder were recaptured across the road only rarely. However, some individuals of a Swedish snail species (*Arianta arbustorum*) that were captured and translocated to the opposite sides of narrow paths or relatively wider roads did return to the capture sides of paths (Baur and Baur, 1990).

2.4.3 Edge Effects

Aside from fragmenting habitat, roads and trails of any kind also create habitat edges (Reed and others, 1996). In many instances, these edge effects extend well beyond the road's actual footprint and for some species the effects may extend well into the desert interior. Therefore, assessing edge effects of roads and trails on wildlife may entail determining distributions of wildlife in reference to the extent of any one edge effect (Yahner, 1988). Even then there may be an array of factors that vary the distances from roads/trails at which edge effects may be apparent. For example, Nicholson (1978) indicates that metapopulations of desert tortoises may be depleted within 0.8 km (0.5 mi) of highway edges, and von Seckendorff Hoff and Marlow (1997) indicate that this effect may extend as far as 3.5 km (2.2 mi) from the highway edge.

Given the frequent incidence of significant vegetation cover along road edges, many organisms may be attracted to right-of-way habitats. For example, Adams and Geis (1983) found greater small mammal density within interstate right-of-way habitats than in adjacent habitats. Density, however, can indicate habitats sinks to which animals retreat when more desirable habitats are occupied (Van Horne 1983). Alternatively, road edges may serve as ecological traps (Andrews, 1990) that are attractive and replete with necessary resources on the one hand, but impose unusually high mortality rates on the other hand. For example, birds may be attracted to lush roadside vegetation for breeding, nesting, or foraging (Clark and Karr, 1979), but they may be at great risk of mortality due to being hit by vehicles (Mumme and others, 2000). Similarly, avian eggs and nestlings can experience increased mortality due to high rates of predation (Yahner and others, 1989) in edge habitats. As mentioned in the section above, edge effects along roads can alter or preclude the seasonal movements of amphibians to their breeding pools (Gibbs, 1998; Vos and Chardon, 1998).

In the same ways that travel routes promote increased dispersal of non-native and invasive plant species, they also promote increased distributions of wildlife species otherwise unlikely to be common in a given area; in turn, this exerts additional competitive pressures on native species. Huey (1941) documented pocket gophers (*Thomomys umbrinus*) extending their ranges across the Mojave Desert via roads and canal systems. Although much of the surrounding desert landscape contained soils unsuitable for gophers, the attractive habitat (greater cover of vegetation resulting from increased moisture availability) along roadsides and canals facilitated the spread of these animals (Huey, 1941). An additional important edge effect associated with roads of many types is the presence of utility infrastructures, which can contribute to significantly altered predator-prey relationships along roads. For example, raven species (*Corvus* spp.) have increased their distribution throughout the Mojave Desert, primarily due to the fact that they can perch along utility structures to scan for carcasses on adjacent roads (paved and unpaved) (Knight and Kawashima, 1993), a significant concern in light of the fact that Berry and others (1986) reported ravens as being responsible for 68 and 75 percent of mortality among juvenile desert tortoises on two study plots.

2.4.4 OHV Disturbance and Noise

Vehicular traffic is also a source of noise and other stimuli that have the potential for disturbing wildlife along any type of road or trail (Singer, 1978; van der Zande, 1980; Brattstrom and Bondello, 1983; Bowles, 1995; Reijnen and others, 1995, 1996; Bowles, 1995; Kaseloo and Tyson, 2004). Veen (1973; as interpreted and translated by van der Zande, 1980) found that four shorebird species inhabiting open grassland areas were disturbed within 500–600 m of a “quiet rural road” and within 1600–1800 m of a “busy highway;” van der Zande (1980) reanalyzed Veen’s data and yielded similar results for three of the four species, and went on to conclude that populations of these birds were diminished by as much as 60 percent over those distances. Forman and Alexander (1998) found that noise levels generally increase with traffic intensity, and Reijnen and others (1995, 1997) concluded that traffic noise can lead to significant reductions in breeding bird densities. Larger animals also exhibit responses to the intensity of traffic and traffic noise. Lyren (2001) found that coyotes changed their road-crossing periods in response to changes in traffic intensity throughout the day, and Singer (1978) reported that, in response to the shifting of truck gears, mountain goats ran away from a road edge when the truck was 1 km (0.6 mi) away from them, and they ran away from a lick that was 400 m (437.4 yd) from the road.

Noise emitted from certain types of OHVs can be as high as 110 decibels, which is near the threshold of human pain (Lovich and Bainbridge, 1999). Although sounds from OHV motors are not the loudest anthropogenic sounds, in wildlife habitats they are emitted more frequently than other high-intensity sounds (Brattstrom and Bondello, 1983), and the effect on animals can be significant. For example, sand lizards (*Uma scoparia*) and kangaroo rats (*Dipodomys deserti*) experienced hearing loss that lasted for weeks after being exposed to less than 10 minutes of dune buggy playback recordings played intermittently at lower decibel levels than the animals would have been exposed to in the actual presence of a dune buggy (Brattstrom and Bondello, 1983); subsequently, both species were unresponsive to recordings of predator sounds. In two other studies, kangaroo rats (*Dipodomys spectabilis*) experienced inner ear bleeding when subjected to OHV noise (Berry, 1980b; Bury, 1980). Another issue is the way in which OHV noise (sound pressure) may simulate that of natural sounds (thunder, for example) to which many animals may be adapted to respond. For example, in response to 30 minutes of taped motorcycle sounds, Brattstrom and Bondello (1983) documented a spadefoot toad (*Scaphiopus couchii*) emerging prematurely (wrong season, absence of rain) from its burrow, most likely because the sound mimicked that of thunder, to which the species would normally respond.

Noise, lights, and other disturbances associated with OHV activities also have the potential for eliciting stress responses from a broad spectrum of wildlife taxa. Indeed, studies have shown that ungulates, birds, and reptiles all experience accelerated heart rates and metabolic function during disturbance events; in turn, animals may be displaced and experience reproductive failure and reduced survivorship (see review in Havlick, 2002). For example, radio-collared mule deer disturbed by ATVs altered their patterns of foraging and spatial use of habitat; deer in undisturbed areas, however, exhibited no such changes (Yarmoloy and others, 1988). In addition, Yarmoloy and others (1988) found that harassment of deer resulted in diminished reproductive output in the following fawning season, whereas deer that were not harassed experienced no change in reproduction.

2.4.5 Wildlife Mortality and Related Issues

Direct wildlife mortality can result from vehicular impact (Harris and Gallagher, 1989; Beier, 1993; Bruinderink and Hazebrook, 1996; Moore and Mangel, 1996), thus removing individuals from populations (Harris and Gallagher, 1989; Forman and Alexander, 1998); thus, habitats containing roads may represent population sinks for any species that commonly attempts to move from one habitat fragment to another by crossing roads (Kline and Swann, 1998). If mortality rates exceed rates of reproduction and immigration, wildlife populations decline (Beier, 1993; Bruinderink and Hazebrook, 1996; Moore and Mangel, 1996; Forman and Alexander, 1998). Previous studies indicate that mortality rates vary widely according to habitat and road or route characteristics (for example, road width, traffic density and speed, adjacent habitat) (Ward, 1982; Bashore and others, 1985; Foster and Humphrey, 1995; Evink and others, 1996, 1998), as well as taxa studied—invertebrates: Seibert and Conover (1991), Munguira and Thomas (1992); reptiles and amphibians: Rosen and Lowe (1994), Ashley and Robinson (1996), Boarman and others (1998), Rudolph and others (1998), Means (1999); birds: Dhindsa and others (1988), Moore and Mangel (1996), Mumme and others (2000); and mammals: Gilbert and Wooding (1996), Romin and Bissonette (1996), Lehnert and Bissonette (1997), Gunter and others (1998), Lyren (2001). Even where the frequency of wildlife mortality is relatively low most of the year, it may increase during certain seasons (Feldhammer and others, 1986; Bruinderink and Hazebrook, 1996) or when traffic frequency increases (McCaffery, 1973). Furthermore, population dynamics can be altered if low mortality rates nonetheless cause disproportionate mortality among specific sex and/or age classes (Beier, 1993; Moore and Mangel, 1996; Mumme and others, 2000).

Several researchers have conducted extensive monitoring at desert OHV sites and undisturbed sites to compare direct effects of OHV activity on mortality and abundance of certain reptile species (Bury and others, 1977; Berry, 1980a; Bury, 1980; Luckenbach and Bury, 1983; Brooks, 1999; Grant, 2005). Of important concern is the susceptibility of desert tortoises to mortality on all types of roads. Berry (1980a) found a link between OHV activity and population declines of the desert tortoise and Couch's spadefoot toad (*Scaphiopus couchii*); numbers of tortoises and active burrows in a 25-ha control plot were significantly greater than in a similar plot exposed to OHV activity, presumably the result of direct mortality from vehicles or the collapsing of burrows caused by OHV traffic (Lovich and Bainbridge, 1999). Additionally, the body masses of subadult and adult tortoises in the control plot were greater than those of tortoises in the OHV area (Bury and Luckenbach, 1986, cited *in* Lovich and Bainbridge, 1999). When comparing lizards in OHV-impacted plots to control plots, controls supported 1.8 times more species, 3.5 times more individuals, and 5.9 times more biomass (Luckenbach and Bury, 1983). Similarly, Bury and others (1977) found more reptile species (1.63 times more) and greater reptile abundance (182 percent more individuals) at control sites than at OHV sites. In another study, the remains of 39 tortoises were recorded during three surveys over a 2.5-year period along a 24-km (14.9-mi) section of paved highway in the western Mojave Desert (Boarman and others, 1993). Snakes also experience high rates of mortality in the Mojave Desert due to their strategy for thermoregulation (lying on warm surfaces, such as roads; Sullivan, 1981). Rosen and Lowe (1994), who conducted nighttime snake surveys along a 2-lane paved road in the Sonoran Desert (primarily within Organ Pipe Cactus National Monument), documented a 72 percent rate of snake mortality (104 live, 264 dead); mortality peaked in spring—when snake activity was moderately high and automobile traffic had not yet reached its summer minimum—and during rain events in the monsoon season (July through early September). Overall snake mortality during the entire 4-year study was estimated at 2,383 snakes (13.5 snakes/km/year; 8.1 snakes/mi/year), although actual numbers were likely closer to 4,000.

Densities and species diversity of desert birds and small mammals also have been reported to decrease in areas where OHV use was extensive (Busack and Bury, 1974; Bury and others, 1977; Luckenbach, 1978; Luckenbach and Bury, 1983; Brooks, 1999). Direct and indirect effects of OHVs on these species include breaking shrubs containing nests (nests, eggs, or nestlings destroyed) and diminished cover when shrubs are reduced or eliminated, mortality due to vehicle impact (especially ground-dwelling animals), and collapse of burrows due to OHV traffic (Bury and others, 1977). Bury and others (1977) found greater small mammal species richness (1.25 times greater) and abundance (500 percent more individuals) at control sites than OHV sites. Similarly, Luckenbach and Bury (1983) found 1.5 times more small mammal species, 5.1 times more individuals, and 2.2 times more biomass in control plots than in OHV-impacted plots; the number of desert kangaroo rats recorded in OHV plots was 53 percent lower than the number in control plots. Luckenbach and Bury (1983) found that overall animal activity—as measured by track frequencies—was greater in control areas than it was in OHV-use areas: arthropod tracks were 24 times more abundant, kangaroo rat tracks were 5 times more abundant, kit fox tracks were 2 times more abundant, and cottontail rabbit tracks were 10 times more abundant. Finally, Brooks (1999) found that protected areas in the Desert Tortoise Research Natural Area supported a greater abundance and species richness of birds and lizards than nearby portions of the desert subjected to intense OHV use and past sheep grazing. In one study, however, road mortality did not appear to have detrimental effects on densities of small mammals inhabiting highway right-of-ways, although the authors admit that they could not rule out confounding effects of immigration (Adams and Geis, 1983). In a study of 36 radio-marked flat-tailed horned lizards (*Phrynosome mcallii*) subjected to high (60 percent OHV track coverage in 60 minutes of riding time), low (30 percent in 20 minutes of riding time), and no (0 percent) impact by OHV traffic in 100 × 100 m (109.4 × 109.4 yd) plots, all survived. At the time of OHV treatment, however, 32 of the lizards were in their hibernation burrows 2–17 cm (0.8–6.7 in) underground (Grant, 2005), and it remains unclear whether soil substrates and vegetation growing above the burrows helped protect the animals from being crushed (21 of the 32 were under shrubs, and 8 burrows were known to have been run over directly by OHVs).

A major indirect effect of OHV activity on vertebrate survivorship is loss of vegetation cover. For all terrestrial vertebrates sampled, including species of conservation concern (desert kangaroo rat [*Dipodomys deserti*] and fringe-toed lizard [*Uma notata*]), Bury and others (1977) found a positive correlation between the percent canopy cover of creosote bush and species richness, abundance, and biomass. In a study of OHV effects on biota (including herbaceous and perennial plants, arthropods, lizards, and mammals) of the Algodones Dunes area in California, Luckenbach and Bury (1983) detected 9.4 times more cover, and 40 times more overall volume in control plots than in OHV-impacted plots, largely because shrubby perennial cover was greater in control plots. Another indirect effect of OHV activity on wildlife mortality is the proliferation of routes that provide greater access to remote places by hunters, poachers, and people seeking several forms of nonconsumptive recreation (Boyle and Samson, 1985; Andrews, 1990). Boyle and Samson (1985) also report a variety of nonconsumptive recreation impacts on wildlife, including flushing animals off nests; unnecessary energy expenditures; and displacement of animals from food, shelter, and other vital resources. Of particular concern was the increasing access that roads provide for tortoise collectors, which may explain declining trends in tortoise numbers along highways (Boarman and others, 1997).

2.4.6 Annotated Bibliography for OHV Effects on Wildlife and Habitats: Native, Threatened, and Endangered Species

Berry, K.H., 1980, A review of the effects of off-road vehicles on birds and other vertebrates, *in* DeGraaf, R.M., and Tilghman, N.G., eds., Management of western forests and grasslands for nongame birds—Workshop proceedings, Salt Lake City, Utah, February 11–14, 1980: Ogden Utah, U.S. Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT–86, p. 451–467.

A review of the literature on the effects of off-road vehicles revealed that OHV use has significant effects and can reduce numbers, diversity, and biomass of birds and other vertebrates. The degree of impact depends upon amount and intensity of OHV use, habitat type, and sensitivity of the species.

Boarman, W.I., and Sazaki, M., 2006, A highway's road-effect zone for desert tortoises (*Gopherus agassizii*): Journal of Arid Environments, v. 65, no. 1, p. 94–101.

Roads can affect populations of animals directly (vehicle-animal collisions) and indirectly (due to habitat fragmentation and dispersal/proliferation of non-native or predatory species). This study investigated the effect of a 2- to 4-lane highway (with a posted speed limit of 65 mi/hr [105 km/hr] and an average daily traffic intensity of 8500 vehicles) on threatened desert tortoise (*Gopherus agassizii*) populations in the Mojave Desert, California, and attempted to determine the width of the road-effect zone by counting signs of tortoises (shells, tracks, scats, burrows, and pallets) along transects at 0, 400, 800, and 1600 m from and parallel to the edge of a highway. Mean sign count was 0.2/km (0.32/mi) at 0 m (0 yd), 4.2/km (6.72/mi) at 400 m (437.4 yd), 5.7/km (9.12/mi) at 800 m (874.9 yd), and 5.4/km (8.64/mi) at 1600 m (yd) from the highway edge. The differences between all distances except 800 and 1600 m (874.9 and 1749.8 yd) were statistically significant, suggesting that tortoise populations in the study area were depressed within a zone extending at least 400 m (437.4 yd) from the highway.

Brattstrom, B.H., and Bondello, M.C., 1983, Effects of off-road vehicle noise on desert vertebrates, *in* Webb, R.H., and Wilshire, H.G., eds., Environmental effects of off-road vehicles—Impacts and management in arid regions: New York, Springer-Verlag, p. 167–206.

This study determined that sand lizards (*Uma scoparia*) and kangaroo rats (*Dipodomys deserti*) suffered hearing loss lasting for weeks after being exposed to less than 10 minutes of playback recordings of dune buggy sounds played intermittently at intensities lower than the average intensity levels actually emitted by OHVs. Such impacts led to the inability of both of these species to respond to recordings of predator sounds. A spadefoot toad (*Scaphiopus couchii*) emerged prematurely from its burrow when exposed to 30 minutes of taped motorcycle sounds.

Brooks, M.L., 1999, Effects of protective fencing on birds, lizards, and black-tailed hares in the western Mojave Desert: Environmental Management, v. 23, no. 3, p. 387–400.

Effects of a protective (fenced) area on birds, lizards, black-tailed hares (*Lepus californicus*), perennial plant cover, and structural diversity of perennial plants were evaluated from spring 1994 through winter 1995 at the Desert Tortoise Research Natural Area (DTNA), in the Mojave Desert, California. Abundance and species richness of birds were greater inside than outside the DTNA; these effects, however, were more pronounced during breeding season and a year of high rainfall than during winter and a year of low rainfall. Nesting activity was also more frequent inside the exclosure. Total abundance and species richness of lizards and individual abundances of western whiptail lizards (*Cnemidophorous tigris*) and desert spiny lizards (*Sceloporus magister*) were greater inside than outside the exclosure. Black-tailed hares generally prefer areas of low perennial plant cover, which may explain why they were more abundant outside than inside the DTNA. Habitat structure may not affect bird and lizard communities as much as availability of food at this desert site, and the greater abundance and species richness of vertebrates inside than outside the DTNA may correlate with abundances of seeds and invertebrate prey.

Bury, R.B., Luckenbach, R.A., and Busack, S.D., 1977, Effects of off-road vehicles on vertebrates in the California desert USA: Wildlife Research Report no. 8, U.S. Fish and Wildlife Service, Washington, D.C., p. 1–23.

This study compared differences in avian diversity, abundance, and biomass in unused and OHV-disturbed sites. Compared to OHV sites, reptile species richness was 1.63 times greater and there were 270 more individuals at control sites. Similarly, mammal species richness was 1.25 times greater and there were 115 more individuals at control sites than at OHV sites. The potential for ground nests of birds to be crushed and incubating birds to abandon nests was greater in areas of high OHV activity. Indirect effects of OHV activity on vertebrates were primarily caused by the loss of vegetation cover. There was a positive correlation between the cover of creosote bush and the total number of species, abundance, and biomass of all terrestrial vertebrates sampled.

Bury, R.B., and Luckenbach, R.A., 2002, Comparison of desert tortoise (*Gopherus agassizii*) populations in an unused and off-road vehicle area in the Mojave Desert: Chelonian Conservation and Biology, v. 4, no. 2, p. 457–463.

This study examined habitat, abundance, and life history features of desert tortoises (*Gopherus agassizii*) on two 25-ha plots in the western Mojave Desert: one unused and one used by OHVs. The unused plot had 1.7 times more live plants, 3.9 times more plant cover, 3.9 times more desert tortoises, and 4.0 times more active tortoise burrows than a nearby area used heavily by OHVs; these between-plot differences were all statistically significant. Furthermore, the few large-sized tortoises in the OHV plot had less body mass than those in the unused area. Although the scope of this study was limited to one paired-plot comparison, current data suggest that operation of OHVs in the western Mojave Desert results in major reductions in habitat and tortoise numbers, and possibly the body mass of surviving tortoises.

Lovich, J.E., and Bainbridge, D., 1999, Anthropogenic degradation of the southern California desert ecosystem and prospects for natural recovery and restoration: Environmental Management, v. 24, no. 3, p. 309–326.

Large areas of the southern California desert ecosystem have been affected by off-highway vehicle use, overgrazing by domestic livestock, agriculture, urbanization, construction of roads and utility corridors, air pollution, military training exercises, and other activities. Secondary contributions to degradation include the dispersal and proliferation of exotic plant species and a higher frequency of anthropogenic fire. Effects of these impacts include alteration or destruction of macro- and micro-vegetation elements, establishment of annual plant communities dominated by exotic species, destruction of soil stabilizers, soil compaction, and increased erosion. This paper provides a broad view of impacts on biota and cites several pertinent studies relative to OHV impacts on wildlife. The authors suggest that given the sensitivity of desert habitats to disturbance and the slow rate of natural recovery, the best management option is to limit the extent and intensity of impacts as much as possible.

Luckenbach, R.A., and Bury, R.B., 1983, Effects of off-road vehicles on the biota of the Algodones Dunes, Imperial County, California, USA: Journal of Applied Ecology, v. 20, no. 1, p. 265–286.

Algodones Dunes, the largest dune complex in California, contains many unique species; however, it also receives the greatest use by off-road vehicles in California. Studies of paired plots (unused versus OHV-impacted) and animal tracks along sand sweeps clearly demonstrated that OHV activities in the Algodones Dunes significantly reduced the biota. There were marked declines in herbaceous and perennial plants, arthropods, lizards, and mammals in OHV-used areas compared with nearby controls. All sand-adapted species, including several rare or threatened plants, were greatly reduced in habitats where OHVs operate; the biota was affected even by relatively low levels of OHV activity. Areas heavily used by OHVs had virtually no native plants or wildlife.

Rosen, P.C., and Lowe, C.H., 1994, Highway mortality of snakes in the Sonoran Desert of southern Arizona: Biological Conservation, v. 68, no. 22, p. 143–148.

A total of 368 snakes (104 live, 264 dead) were recorded over four years on a paved highway during 15,525 km (9,647.2 mi; mostly within Organ Pipe Cactus National Monument, Arizona) of driving along the road to detect amphibians and reptiles during rainfall events or while basking on the warm road surface. During 4 years, an estimated 2,383 snakes were killed on this stretch of pavement, although the actual number killed was probably closer to 4,000.

Webb, R.H., and Wilshire, H.G., 1983, Environmental effects of off-road vehicles—Impacts and management in arid regions: New York, Springer-Verlag, 534 p.

This book discusses the physical and biological effects of OHVs (recreational, mining, and military vehicles) on arid-land ecosystems, including effects on soils, vegetation, and wildlife. It also points out the loss of choices that OHV effects impose on future land users. Actual case studies are presented, complete with practical solutions, detailed planning measures that can be taken to reduce the adverse effects of OHVs, methods that can be used to rehabilitate the physical systems and vegetation communities of disturbed areas, and management concepts and practices that can be employed in protecting susceptible areas, including regulations and education.

2.5 OHV Effects on Water Quality

2.5.1 Section Summary

The direct effects of OHV activity on aquatic systems have received surprisingly little attention, due, in part, to the fact that OHV-impact research has focused on arid environments, where aquatic systems are seasonal or rare. Nonetheless, there is great potential for OHV activities to affect water quality in arid environs as well as well-watered regions. As described in Sections 2.2 and 2.3, soil properties and vegetation cover may be altered by OHV use; in turn, surface patterns of precipitation runoff (amount, velocity) may be altered, resulting in accelerated rates of erosion and sedimentation and elevated levels of turbidity in affected watersheds. Where slope is a factor, the extensive networks of OHV routes proliferating across landscapes can serve as conduits that direct or alter the direction of surface flows. These conduits may be eroded to form gullies that channel dislodged sediments and contaminants into aquatic ecosystems. Water quality also is adversely affected by OHV-raised dust that settles into aquatic systems.

OHV-dispersed chemicals also may be transported into aquatic systems. The operation of OHV engines, especially 2-stroke engines, can impact water quality through spills and emissions. These contaminants may enter aquatic systems via direct flushing, or they may be adsorbed to sediments and/or absorbed by plant materials, both of which are easily transported to aquatic systems by precipitation runoff or wind. Spill or emission contaminants may include 1,3-butadiene, benzene and ethylbenzene, xylenes, and toluene. Prior to the ban on leaded gasoline, lead levels were high in plants and animals near roads, and although the 1996 ban on leaded gasoline has resulted in dramatic declines in lead levels, it persists in the soil and may be mobilized when soils are eroded into wetlands.

2.5.2 Sedimentation and Turbidity

Areas naturally most susceptible to water-quality problems are those where infiltration rates are low, slopes are steep, the ratio of surface sand and gravel to finer particles is low, and where rainfall events are typically prolonged and intense (Iverson and others, 1981). Altering soil texture, disrupting soil crusts or desert pavement, and reducing vegetation cover can increase the soil’s susceptibility to erosion; in turn, rates of sedimentation and turbidity levels can increase and alter the water quality of a given watershed, including streams and rivers, lakes, and small, isolated wetlands, including vernal pools (Forman and others, 2003). Sediments can displace the water-holding volume of a wetland, thus diminishing or eliminating the wetland’s hydrological function (Luo and others, 1977). For example, where OHVs had traveled over the soil, Iverson and others (1981) found that surface runoff was 5 times greater and yielded 10-20 times more sediment than where soils were undisturbed.

Where OHV activity occurs, networks of OHV routes proliferate. Wheel cuts and tracks within these networks may serve as water conduits that channel and direct water flow containing sediments and contaminants into aquatic ecosystems (Wemple and others, 1996; Forman and others, 2003, p. 185–197). The generally impervious nature of soils compacted by OHV traffic enhances gully formation in these conduits, thus promoting additional flows of sediments and suspended solids into aquatic systems, effectively extending the drainage network of a given watershed, and potentially changing the timing of peak runoff flows (Wemple and others, 1996). The presence of OHV-route networks is an important factor in determining the severity of potential sedimentation in nearby aquatic systems. In particular, Wemple and others (1996) found that the drainage ditches along logging roads and the gullies that form below culvert outlets (where drainage flows pass under a road, or cross-drains) on steep slopes served as primary conduits linking surface flows to streams. The extent to which sediments might be carried along these conduits and into aquatic systems depends primarily on the presence of obstructions below cross drains and the spatial intervals between them (Haupt, 1959). In situations where cross drains were positioned at sufficient distances from streams, the drainage discharge infiltrated the soil and did not contribute to sedimentation in streams (Haupt, 1959). In areas characterized by soils with relatively low infiltration rates, such as those compacted by OHV use, transport of sediments over greater distances and into aquatic systems may be substantial.

Furniss and others (2000) describe similar effects of road and/or trail networks across a landscape. In particular, they discuss the continuous "hydrological connections" that facilitate sediment transport between surface flows and waterways. Furniss and others (2000) go on to list ways in which water and associated sediments enter stream systems from roads, including (1) inboard ditches (ditches perpendicular to the road footprint and that bisect the road) delivering runoff to a stream at a road-stream crossing, (2) inboard ditches delivering water to a cross-drain (culvert, dip, waterbar), (3) where sufficient discharge is available to create a gully or sediment plume that extends to the stream channel, (4) roads sufficiently close to streams so that the fillslope (road fill between the outside edge of the road and the base of the fill where it meets the natural ground surface) encroaches on the stream, and (5) landslide scars on the road fill. These connections provide direct routes for accelerated runoff transporting sediments and road-associated contaminants to natural drainage channels.

2.5.3 Dust and Contaminants

Water quality also is adversely affected when fugitive dust and contaminants enter aquatic systems. Emissions from OHVs, particularly those with 2-stroke engines, can include a variety of contaminants, which may settle directly in wetlands or they may be deposited in snow or directly on soils during rain events, from which they may be mobilized into wetlands. Arnold and Koel (2006), who tested snowmelt runoff exposed to significant snowmobile emissions in Yellowstone National Park, detected benzene, ethylbenzene, m- and p-xylene, o-xylene, and toluene, and although all compounds were within the limits set by the U.S. Environmental Protection Agency, it is not clear what the cumulative impacts of these chemicals may be in watersheds. Adams (1975) found that the stamina of brook trout experimentally exposed to elements commonly found in snowmobile emissions, as measured by their ability to swim against the water current, was significantly diminished compared to that of control fish.

Airborne dust—and contaminants adsorbed to dust particles—raised by OHV traffic may eventually settle directly into wetlands (Forman and others, 2003, p. 231–234). The potential for adsorbed contaminants to be carried along with precipitation runoff and into wetlands is also a concern, as are plant materials containing absorbed contaminants. Finally, contaminants may enter aquatic habitats by direct flushing of exposed contaminants (for example, petroleum puddles). Prior to the ban on leaded gasoline, lead levels were high in plants and animals near roads (Daines and others, 1970; Motto and others, 1970; Quarles and others, 1974; Wheeler and Rolfe, 1979). Although the 1996 ban on leaded gasoline has since resulted in dramatic declines in lead levels along roadsides and in organisms, it persists in the soil and may be mobilized when soils are eroded into wetlands.

2.5.4 Annotated Bibliography for OHV Effects on Water Quality

Adams, E.S., 1975, Effects of lead and hydrocarbons from snowmobile exhaust on brook trout (*Salvelinus fontinalis*): Transactions of the American Fisheries Society, v. 104, no. 2, p. 363–373.

Prior to snowmobiling season, hydrocarbon levels in the water of a pond in Maine were undetectable; by the time of ice-out in spring, hydrocarbon levels had reached 10 parts per million (ppm) in the water and 1 ppm in exposed fish. In addition, exposed brook trout fingerlings contained 9 to 16 times more lead than control trout. Brook trout (*Salvelinus fontinalis*) held in aquaria for 3 weeks in melted snow containing three different concentrations of snowmobile exhaust also showed hydrocarbon and lead uptake. Stamina, as measured by the ability to swim against current, was significantly less in trout exposed to snowmobile exhaust than in control fish.

Brabec, E., Schulte, S., and Richards, P.L., 2002, Impervious surfaces and water quality—A review of current literature and its implications for watershed planning: Journal of Planning Literature, v. 16, no. 4, p. 499–514.

This paper focuses on the effect of impervious surfaces on the health of nearby aquatic habitats. Although considerable research has been done to define watershed thresholds of impervious surfaces (beyond which water quality declines), there are numerous flaws in the assumptions and methodologies used. Given refinement of the methodologies, accurate and usable parameters for preventive watershed planning can be developed, including thresholds of impervious surfaces and balances between pervious/impervious surfaces within a watershed.

Brown, K.J., 1994, River-bed sedimentation caused by off-road vehicles at river fords in the Victorian Highlands, Australia: Water Resources Bulletin, v. 30, no. 2, p. 239–250.

This study investigated some of the effects occurring at OHV crossings on two rivers in eastern Australia, where many road crossings occur at low-level fords. It provides a method whereby the amount of sediment redeposited downstream of a ford can be measured. Attention is drawn to the fact that sediment is contributed to rivers by five major processes: the exposure of surfaces, the concentration of surface runoff in wheel ruts, soil compaction and subsequent reduction of water infiltration leading to increased surface runoff, backwash from the vehicle, and undercutting of banks by bow-wave action. The last two of these processes have not been reported previously. Sediment collection experiments in two upland rivers indicated a mean deposition rate at the stream bed of approximately 1,000 g/m^2 over a period of 30 days.

Furniss, M.J., Flanagan, S.A., and McFadin, B.A., 2000, Hydrologically connected roads—An indicator of the influence of roads on chronic sedimentation, surface water hydrology, and exposure to toxic chemicals: U.S. Forest Service, Stream Systems Technology Center, Rocky Mountain Research Station, Technical Report, 4 p., *http://www.stream.fs.fed.us/streamnt/jul100/jul100_2.htm*.

This study defines the concept of forest-road drainage as a transport system for sediment into streams and proposes design changes to road drainage that would prevent or minimize this movement. The proportion of road that is hydrologically connected to a stream network may be a useful indicator of the potential for several adverse effects, including (1) the delivery of road-derived sediments to streams; (2) hydrologic changes associated with subsurface flow interception, concentration, and diversion; (3) increased drainage density; (4) extension of the stream network; and (5) the potential for road-associated spills and chemicals to enter streams.

Hamilton, L.J., 2002, A study of the effects of ORV stream crossings on water quality of two streams located in the Angelina National Forest, Texas—A physicochemical and benthic macroinvertebrate analysis: Masters Abstracts International, v. 40, no. 3, p. 668.

A study was conducted for the U.S. Forest Service to determine whether OHV-based stream crossings affected water quality of two streams located in Texas. The sites differed most in turbidity, total solids, Shannon's diversity index, dissolved oxygen, nitrate, and ratios of Chironomidae:EPT (Ephemeroptera + Plecoptera + Trichoptera, a common indicator of taxonomic richness detected during stream surveys to assess water quality), although there were no significant differences in the physicochemical properties. At one site, however, the upstream and downstream plots differed significantly in terms of two benthic indices—Hilsenhoff's *Biotic Index* and *Ratio of Scrapers to Filtering Collectors*.

Katz, M., Legore, R.S., Weitkamp, D., Cummins, J.M., and Anderson, D., 1972, Effects on freshwater fish: Journal of the Water Pollution Control Federation, v. 44, no. 6, p. 1226–1250.

This is a literature review of the effects of water pollutants on freshwater fish. Topics include (1) tests to determine the lethality of estuarine and some polluted river waters to trout and cyprinids; (2) estimated degrees of river pollution based on bacterial and chemical analysis of water samples; (3) documentation of some effects of municipal wastewater effluents on the water quality, fish populations, and bottom-fauna characteristics of a receiving stream; and (4) observations of the environmental effects of pollutants such as synthetic detergents, industrial wastes, and pesticides.

Roy, A.H., Rosemond, A.D., Leigh, D.S., Paul, M.J., and Wallace, J.B., 2003, Habitat-specific responses of stream insects to land cover disturbance—Biological consequences and monitoring implications: Journal of the North American Benthological Society, v. 22, no. 2, p. 292–307.

This study analyzed the impact of a range of physical and chemical stressors on aquatic insects and tested whether the effects of these stressors differed in three habitat types: riffles, pools, and banks. Riffle assemblages were affected by both physical (for example, streambed mobility) and chemical (specific conductance, nutrient concentration) variables. The density of aquatic insects in pools also was correlated to physical and chemical variables, but there were few relationships with pool or bank richness or bank density. Because relative impacts of disturbance in riffles were greater than in banks, the authors found greater differences between riffle and bank richness in streams with greater sedimentation. The proportion of bank richness (bank richness/bank + riffle richness) increased with finer bed sediment and increased bed mobility. The study also compared richness of facultative taxa (found in multiple habitats) between sites characterized as minimally impacted and sediment-impacted. In riffles, richness of facultative taxa was lower in sediment-impacted than in minimally impacted sites, but was similar for both disturbance groups in banks.

Wheeler, A.P., Angermeier, P.L., and Rosenberger, A.E., 2005, Impacts of new highways and subsequent landscape urbanization on stream habitat and biota: Reviews in Fisheries Science, v. 13, no. 3, p. 141–164.

This paper emphasizes a more thorough consideration of highway impacts and, ultimately, better land-use decisions by conceptualizing road development in three stages: initial construction, road presence, and eventual landscape urbanization. Road construction is characterized by localized physical disturbances, which generally subside through time. In contrast, road presence and landscape urbanization are characterized by persistent physical and chemical impacts. Though not specific to OHV activity, this paper does focus on the fact that landscape urbanization is clearly the greatest threat to stream habitat and biota, as stream ecosystems are sensitive to even low levels (less than 10 percent of a given watershed) of urban development. Researchers know little about the occurrence, loading rates, and biotic responses to specific contaminants in runoff from roads. Also needed is a detailed understanding of how drainage crossings, especially culverts, affect fish populations via constraints on movement and how road networks alter natural regimes (streamflow, temperature).

2.6 OHV Effects on Air Quality

2.6.1 Section Summary

Fugitive dust raised by OHV traffic on unpaved roads/trails can contribute significantly to air-quality problems. Also problematic are OHV emissions, particularly from 2-stroke engines. Currently, many OHVs in use, including off-highway motorbikes and ATVs, run on 2-stroke engines, which do not burn fuel completely and produce significant amounts of airborne contaminants, including nitrogen oxides, carbon monoxide, ozone, aldehydes, and extremely persistent polycyclic aromatic hydrocarbons (PAH), including the suspected human carcinogen, methyl tert-butyl ether (MTBE). Some airborne contaminants settle onto plants or into soils and function as fertilizers, thus causing changes in plant community composition and altering growth rates. The accumulation of emissions contaminants is evident in the tissues of plants and animals exposed to them. Prior to the ban on leaded gasoline, lead also was prevalent in plants and animals near paved roads and other travel routes, and because it persists in the environment, it can still have impacts when contaminated soils are mobilized.

2.6.2 Fugitive Dust Raised by OHV Traffic

Fugitive dust (largely composed of lightweight soil particles, including silt and clay) suspended in the air may impact more total area than any other impact of roads (paved or unpaved; Forman and others, 2003), and it can have significant effects on ecosystems (Westec, 1979). Dust is created and raised into the air as OHVs disturb soil crusts, abrade and pulverize soils, and generate wind currents. Once soil surfaces are disturbed, wind erosion may increase the amount of debris flow (Lovich and Bainbridge, 1999). In 1973, satellite photos detected six dust plumes in the Mojave Desert covering more than 1,700 km^2 (656.2 mi^2); the plumes were attributed to destabilization of soil surfaces resulting from OHV activities (Nakata and others, 1976; Gill, 1996). Along roads in Alaska heavily traveled by various types of vehicles, Walker and Everett (1987) found that dust had buried mosses and very low-statured vegetation in the 10-m-wide area adjacent to each side of the road; dust blankets measured up to 10 cm (3.9 in) deep. Accumulations of dust on vegetation can disrupt photosynthetic and respiration processes, leading to reduced plant growth, reproduction, and survivorship.

2.6.3 Contaminants Associated with OHV Use

Before emissions controls on automobiles became significantly more effective, there was little concern about emissions from small engines; today, however, their relative contribution to air-quality problems is significant (see *http://www.egr.msu.edu/erl/Small%20Engine%20Emissions.html*). This is because small engines, especially 2-stroke models (many of which are being phased out), do not burn fuels completely; thus, their emissions contain the resulting by-products of incomplete combustion, including nitrogen oxides NO_x, sulfur dioxide (SO_2), carbon monoxide (CO), ozone (O_3), aldehydes, and extremely persistent polycyclic aromatic hydrocarbons (PAH). In fact, a very small, 2-stroke engine running for 2 hours emits the same amount of hydrocarbons as driving 10 cars (of the fuel-burning efficiency produced in 1995) for 250 miles each (*http://www.arb.ca.gov/msprog/offroad/sm_en_fs.pdf*).

Pollutants emitted from exhaust can cause a variety of impacts on vegetation. Carbon dioxide may function as a fertilizer and cause changes in plant species composition (Bazzaz and Garbutt, 1988; Hunt and others, 1991; Ferris and Taylor, 1995); nitrogen oxides also may function as fertilizers, producing similar effects along roadsides (Falkengren-Grerup, 1986; Holzapfel and Schmidt, 1990; Angold, 1997). Spencer and Port (1988) found that the soluble nitrogen content of perennial ryegrass (*Lolium perenne*) plants growing within 0-6 m (0-6.6 yd) of a paved road than in plants growing more than 6 m (6.6 yd) from the road, which contributed to greater growth rates and fecundity of aphids (*Rhopalosiphum padi*) inhabiting the plants closest to the road. Sulfur dioxide, which can be taken up by vegetation, may result in altered photosynthetic processes (Winner and Atkison, 1986; Mooney and others, 1988).

Several species of Mojave Desert perennials and annuals were fumigated in experimental chambers to determine their sensitivities to SO_2, nitrogen dioxide (NO_2), and O_3; Thompson and others, 1980; Thompson and others, 1984). Creosote bush (*Larrea* sp.), the only perennial species found to be sensitive to SO_2 and NO_2, exhibited leaf injury and reduced growth when exposed to SO_2 and NO_2; however, numerous annuals, including redstem stork's bill (*Erodium cicutarium*) and desert Indianwheat (*Plantago insularis*; extremely sensitive), cleftleaf wildheliotrope (*Phacelia crenulata*; very sensitive), and wooley desert marigold (*Baileya pleniradiata*), exhibited more dramatic effects, including extensive injury and death (Thompson and others, 1980). Another study by Thompson and others (1984) revealed several annual species that are extremely sensitive to SO_2 and O_3, including brown-eyed primrose (*Camissonia claviformes*), Santa Cruz Island suncup (*C. hirtella*), and Nevada cryptantha (*Cryptantha nevadensis*).

OHV emissions also contain a variety of heavy metals, including zinc, copper, nickel, chromium, and lead (National Research Council, 1986). In terms of overall quantity, lead was one of the most significant heavy metals emitted prior to the ban on leaded gasoline in 1996 (Daines and others, 1970; Motto and others, 1970; Quarles and others, 1974; Wheeler and Rolfe, 1979). At least in desert regions, concentrations of lead particulates along roads were positively correlated with traffic volume (Motto and others, 1970). Within 80 m (87.5 yd) of roadsides, Quarles and others (1974) found that lead concentrations diminished notably from road edges (543 and 190 ppm) to 10 m (47 and 5 ppm, respectively) away from the edge; beyond 80 m, accumulations of lead diminished at lower rates. The declining gradient in lead concentrations away from roadsides may have been due, in part, to the direction of surface water flow (Byrd and others, 1983) as soil and other debris to which lead adheres were flushed away by the volume of water that runs off road surfaces. Although lead emissions from gasoline have declined dramatically since control policies were implemented in the 1970s (Forman and others, 2003), it persists in soils and can continue to move through the environment when contaminated soils are dislodged.

2.6.4 Annotated Bibliography for OHV Effects on Air Quality

Agrawal, Y.K., Patel, M.P., and Merh, S.S., 1981, Lead in soils and plants—Its relationship to traffic volume and proximity to highway (Lalbag, Baroda City): International Journal of Environmental Studies, v. 16, no. 3–4, p. 222–224.

Accumulations of lead from motor-vehicle exhausts on soils and trees growing along a busy thoroughfare in the Lalbag area of Baroda City were studied. Analysis of soils and tree samples showed that the distribution of emitted lead was influenced by the direction of the prevailing wind. Lead concentrations in plants and soils near the roadside were greater than they were in soils and plants 4–6 m away from the roadside.

Bazzaz, F.A., and Garbutt, K., 1988, The response of annuals in competitive neighborhoods—Effects of elevated CO_2: Ecology, v. 69, no. 4, p. 937–946.

Four members of an annual plant community were used to investigate the effects of changing neighborhood complexity and increased carbon dioxide (CO_2) concentration on competitive outcome. Plants were grown in monoculture and in all possible combinations of two, three, and four species in CO_2-controlled growth chambers at CO_2 concentrations of 350, 500, and 700 microliters/liter (µL/L) (1 ppm), with ample moisture and light. Species responded differently to enhanced CO_2 level. The biomass of some species (*Abutilon theophrasti*, for example) increased with increasing CO_2, while that of others (*Amaranthus retroflexus*) decreased with increasing CO_2 concentration. The potential effects of CO_2 on community structure could be profound, particularly at the intermediate levels of CO_2 that are predicted for the first half of the 21st century.

Gish, C.D., and Christensen, R.E., 1973, Cadmium, nickel, lead, and zinc in earthworms from roadside soil: Environmental Science and Technology, v. 7, p. 1060–1062.

Cadmium (Cd), nickel (Ni), lead (Pb), and zinc (Zn) in soils and earthworms along two Maryland highways decreased with increasing distance (10, 20, 40, 80, and 160 ft) from the road. Along each highway, metal residues were greater where traffic volume was greater. Correlations between residues in earthworms and soil decreased with decreasing atomic weights (Pb, Cd, Zn, Ni). Metal residues in soils were positively correlated with quantities of soil organic matter. Earthworms accumulated up to 331.4 ppm of Pb and 670.0 ppm of Zn, concentrations that may be lethal to earthworm-eating animals.

Motto, H.L., Daines, R.H., Chilko, D.M., and Motto, C.K., 1970, Lead in soils and plants—Its relationship to traffic volume and proximity to highways: Environmental Science and Technology, v. 4, p. 231–237.

Lead concentrations increased with traffic volume and decreased with distance from highways. Much of the lead was present as removable surface contamination on plants, and major effects were limited to the soil surface within 100 ft (30.48 m) of the highway.

Nakata, J.K., Wilshire, H.G., and Barnes, G.C., 1976, Origin of Mojave Desert dust plumes photographed from space: Geology, v. 4, p. 644–648.

OHV-raised dust has been an enormous problem in the Mojave Desert, as illustrated by satellite photos that revealed six dust plumes covering more than 1,700 km^2 (656.4 mi^2) of the western Mojave region in January 1973; the dust plumes were attributed to destabilization of ground surfaces, primarily from OHV activity.

Quarles, H.D., Hanawalt, R.B., and Odum, W.E., 1974, Lead in small mammals, plants, and soil at varying distances from a highway: Journal of Applied Ecology, v. 11, no. 3, p. 937–949.

Lead particulates were measured at varying distances from three highways. Lead concentrations were greatest within 10 m (10.9 yd) of the highways. Lead concentrations in the soil along two transects dropped from 543 ppm and 190 ppm at the road edge to 47 ppm and 5 ppm 10 m from the road edge. Both plants and animals were susceptible to lead uptake.

Spencer, H.J., and Port, G.R., 1988, Effects of roadside conditions on plants and insects. II. Soil conditions: Journal of Applied Ecology, v. 25, no. 22, p. 709–715.

An experiment was done to investigate the performance of plants (*Lolium perenne*) grown in roadside soil. Significantly fewer plants germinated in soil taken 0 to 6 m from the road compared with soil taken 6 m from the road. For a given population size, however, plants grown in soil taken from beside the road attained significantly greater dry weight and significantly greater soluble nitrogen content. Nitrogen oxide emissions, identified as the probable cause of these effects, were absorbed by the roadside soil and subsequently assimilated by the plants.

Thompson, C.R., Olszyk, D.M., Kats, G., Bytnerowicz, A., Dawson, P.J., and Wolf, J.W., 1984, Effects of ozone or sulfur dioxide on annual plants of the Mojave Desert: Journal of the Air Pollution Control Association, v. 34, no. 10, p. 1017–1022.

Forty-seven species of annual plants from the Mojave Desert were grown in pots and exposed in open-top field chambers located at Riverside, California, to test their relative sensitivity to SO_2 and O_3. Species differed widely in their response to the pollutants. Three species, *Camissonia claviformis*, *Camissonia hirtella*, and *Cryptantha nevadensis*, were quite sensitive to both pollutants, exhibiting leaf injury when exposed to 0.1 ppm O_3 or 0.2 ppm SO_2. The other species were intermediate in sensitivity, and O_3 sensitivity did not always correspond to SO_2 sensitivity. For 8 of 11 species tested, total sulfur concentration was greater in plants exposed to 0.2 ppm SO_2 than in unexposed plants. *Baileya pleniradiata* and *Perityle emoryi* exhibited the greatest increases in sulfur concentration for exposed versus control plants.

Walker, D.A., and Everett, K.R., 1987, Road dust and its environmental impact on Alaskan taiga and tundra: Arctic and Alpine Research, v. 19, no. 4, p. 479–489.

The physical and chemical characteristics and ecological consequences of road dust in arctic regions were reviewed with emphasis on recent information gathered along the Dalton Highway and the Prudhoe Bay Spine in northern Alaska. Enhanced dust-control measures were considered, particularly where the road passes through scenic lichen woodlands, acidophilic tundra, and in calm valleys where dust commonly was a traffic-safety hazard.

Westec Services Inc., 1979, Fugitive dust impacts during off-road vehicle (ORV) events in the California desert: Tustin, California, WESTEC Services, Inc., Technical Report, 40 p.

Results and analysis of dust monitoring for five desert races demonstrated that factors such as distance from the point of generation, soil moisture, soil characteristics, wind, and relative humidity, as well as the type and number of vehicles in the race, had the largest effect on the amount and type of dust, particulate size, how quickly it settled, and the extent of the human health hazard present during the race. Long period (daily) dust-exposure levels were 10 times

greater than the standard, whereas short period (hourly) dust levels were 100 times greater than the standard under adverse conditions near the race activity.

2.7 Socioeconomic Implications of OHV Use

2.7.1 Section Summary

The socioeconomics of OHV use include OHV user demands, concerns, and attitudes; the economic effects of OHV use on communities near OHV-use areas; the economics of managing OHV activities; the effects of OHV use on non-motorized recreators; and the economics of losing ecosystem services. Although not currently addressed through BLM's indicators of rangeland health, natural resource attributes are heavily influenced by socioeconomic factors. Since the mid 1980s, the incidence of OHV use on public lands has increased substantially, and this trend is expected to continue. Moreover, the economic benefits from travel expenditures and the sales of supplies and equipment in communities bordering OHV-use areas generates significant pressure to maintain or increase current levels of OHV activity. As OHV activity increases, however, increasing stress is placed on natural resources, land managers who must monitor and regulate OHV activities, and visitors seeking non-motorized forms of recreation.

2.7.2 Trends in OHV Use and Technology

In a survey of Utah OHV users commissioned by the Utah Department of Natural Resources, Fisher and others (2001) found that public lands are primary destinations among most users; only one quarter of survey respondents took trips to private land. More specifically, BLM land was the primary destination for ATV, motorcycle, and 4 x 4 vehicle users; U.S. Forest Service land was the secondary destination among ATV and 4 x 4 users; and State land was the secondary destination among motorcycle users (Fisher and others, 2001). Increasing OHV use is likely to be accompanied by greater demand for places where OHVs can be used, particularly near urban areas and corridors; as urban populations increase, so do the numbers of recreators on nearby public lands, thereby putting more stress on the landscape (Brooks and Champ, 2006). The increasing demands also pose problems for land managers already balancing the needs of a dynamic land base, often with limited budgets and/or staffing (Brooks and Champ, 2006; Rocky Mountain Research Institute, 2002). These limitations constrain land managers but not OHV use; thus, OHV recreation is largely "unmanaged." In addition, technology advancements in outdoor recreation equipment have led to production of OHVs that easily access lands previously unimpacted by mechanized recreation (Meine, 1998; Ewert and Shultis, 1999). As a result, new problems have arisen for both previously unimpacted areas and backcountry users who now encounter OHVs. Problems potentially arising from a constrained ability to manage lands include resource degradation, displacement of wildlife, and conflict among users, both within and across user types.

2.7.3 Types, Sources, and Effects of OHV User Conflict

Much of the OHV literature addresses conflicts between OHV users and other land users, even those who are not directly affected by OHV users. Researchers have addressed conflict issues by using a variety of tools or models designed to help managers understand and reduce conflicts between or among user groups. Bury and others (1983, p. 401) describe conflict as existing "whenever incompatible activities occur" and offer three elements that contribute to the incompatibility of activities: spatial and temporal proximity, dominance over the environment, and dependence on technology. When the proximity of activities does not result in direct or indirect (seeing the effects of other uses) encounters among user types, then environmental dominance and technological dependence are more likely to come into play. Dominance over the environment refers to how much an individual feels the need to exert some kind of control over the environment. Dependence on technology can cause conflict when people who retreat to backcountry to seek solace from modern technology clash with those who use technology to enhance their outdoor experiences. Conflict also occurs between land users and land managers. Inconsistent management policies across different land management agencies can cause such conflict, particularly as OHV recreation is ushered from being "unmanaged" to "managed." On many public lands, trails are currently considered open unless posted as closed, and once a trail has been established by users, it is often considered open for use (Brooks and Champ, 2006).

Graefe and Thapa (2004) outline some of the traditional approaches to examining user conflicts through research, including studies of goal interference (first introduced by Jacob and Shreyer, 1980). Goal interference occurs when a user comes into direct (seeing the conflicting recreation type) or indirect (seeing the *effects* of a recreation type) contact with another user type and is impeded from accomplishing the desired purpose of his or her recreation (Badaracco, 1976). The factors that contribute to goal interference are activity style, resource specificity, mode of experience (whether individuals are focused or unfocused), and tolerance for lifestyle diversity. Another model classifies conflict as either interpersonal conflict or a conflict of social values (Vaske and others, 1995). Interpersonal conflict is similar to goal interference in that a user has a problem with another use type and encounters an individual participating in, or evidence of, that type (hearing OHV noise, for example). Social values conflict occurs regardless of whether or not differing user types encounter one another—just knowing that the other recreation type is permitted may be unacceptable.

In the literature on user conflict, conflict is more often characterized as one-sided than two-sided (Badaracco, 1976; Bury and others, 1983; Watson and others, 1997; Graefe and Thapa, 2004). For example, while backpackers may perceive OHV users as disruptive to their experience, it is less likely that OHV users will find backpackers disruptive to their experience (Jackson and Wong, 1982). Displacement is the most common personal coping mechanism by which conflict is abated (Watson et al, 1997; Graefe and Thapa, 2004). That is, if an individual feels negatively enough about certain recreational activities occurring in the area he/she wishes to use, there is a possibility that the individual will simply forgo recreating in the area altogether, thereby increasing the probability that area managers will gradually lose support from that user base (Watson and others, 1997; Graefe and Thapa, 2004).

2.7.4 OHV Users and Their Preferences

Overall, understanding the social effects of OHV use requires understanding the full array of recreational activities sought and the preferences of both OHV and non-OHV users alike. For example, people engaged in camping may include both OHV and non-OHV users, which can result in dissatisfaction among campers. In a survey of campers that included both OHV and non-OHV users, 66 percent indicated that having a regulated OHV riding area nearby would make their stay more enjoyable because it would reduce the number of riders in other areas and maintain a safer environment for both riders and campers (Bury and Fillmore, 1974). When given a choice between having (1) no motorcycle riding area but permission to ride on campground roads, (2) prohibition of all motorcycle riding, or (3) a nearby motorcycle area and no permission to ride on campground roads, 75 percent of riders and campers surveyed preferred the third alternative (Bury and Fillmore, 1974; riders and campers were socioeconomically similar).

Fisher and others (2001) reported that although 63.2 percent of motorcycle users surveyed did not stop to engage in any other type of recreational activity, almost 60 percent of ATV owners and 75 percent of 4 X 4 vehicle owners did engage in other recreational activities during their trips. Of those OHV users who did stop to engage in additional recreational activities, hiking was the most popular (>75 percent of motorcycle/4 X 4 vehicle users and 20 percent of ATV users). Hunting was the other most common recreational activity among ATV users and the second most common activity among 4 X 4 vehicle users; other recreational activities included fishing, camping, and sightseeing (Fisher and others, 2001).

Overall, the results of the user preference surveys discussed previously reveal a potentially conflicted OHV user base in that the quality of their associated recreational activities could be affected by OHV activities. For example, campers who wish to ride OHVs for additional recreation, but who feel strongly that OHV use should be restricted to designated areas, are likely to feel dissatisfied if other OHV users ride through the campground and/or on hiking trails. Similarly, if OHV use in preferred hunting or fishing areas—or other areas crucial to healthy populations of game and fish species—degrades habitat quality that results in diminished game and fish populations, then OHV riders who also hunt and fish may experience dissatisfaction.

Understanding the social effects of OHV activities (and potential outcomes of OHV activities) also requires determining where OHV users like to go and what their preferences are while riding. For example, in Colorado (where user attitudes are likely relatively moderate), Crimmins (1999) reported that

- 38.5 percent of OHV riders use U.S. National Forest Service land,
- 22.4 percent use private land,
- 18.6 percent use BLM land,
- 6.0 percent use State land,
- 3.4 percent use City or county land, and
- 2.3 percent use National Recreation Areas.

These data indicate that the use of public lands for OHV riding far outweighs that of private lands. Crimmins (1999) further reported OHV user preferences in terms of riding area attributes, which included

- no fee for use (if on public land),
- signs indicating all activities allowed on the trail, and
- locations removed from other human activity.

The least important attributes included

- patrolling by staff of land management agencies or local OHV clubs,
- restrooms, and
- loading ramps (Crimmins, 1999).

When presented with a list of priorities for uses of public funds, OHV users selected

- purchasing right-of-ways for OHV access,
- new OHV trail construction,
- erosion control, and
- OHV trail system planning and maintenance.

The low ranking of management patrols probably indicates that users desire more flexibility regarding where they may ride (Crimmins, 1999). Crimmins (1999) also pointed out that although the availability of facilities ranked low in terms of user preferences, management agencies nonetheless generally focus on providing facilities and generally report high user demand.

In terms of OHV user preferences for trail types and features, Bury and Fillmore (1974) reported that variation in terrain was the most important factor. The authors' recommendations for an effective OHV area included

- riding areas established near some, but not all, campgrounds;
- trails kept ≥600 feet (183 m) from the nearest campground;
- trails ≤6 feet (1.83 m) wide; and
- trails that traverse hillsides and include a variety of technical (obstacles, rugged terrain) and non-technical (no obstacles, smooth riding surface) features.

Fisher and others (2001) reported that motorcycle and ATV riders in Utah preferred

- riding off established trails (38.1 and 49.4 percent, respectively),
- double-track trails (12.7 and 17.1 percent),
- single-track trails (12.7 and 4.3 percent),
- moto-cross or ATV courses (9.5 and 15.1 percent), and
- roads (11.1 and 4.3 percent).

The issue of traveling off established trails is a serious concern with respect to natural resource management (Forman and others, 2003; Petersen, 2006); however, areas closed to OHVs and a shortage of designated OHV areas are common complaints among users (Achana, 2005; Fisher and others, 2001; Nelson and others, 2000). For example, in a survey commissioned by the Utah Department of Natural Resources to identify the most important issues affecting OHV use in Utah, 42.3 percent of respondents indicated that "Having enough places to ride" was most important; 8.4 percent indicated "Too many areas closed to OHV use;" and 5.6 percent indicated that "Resource management conservation" was the most important issue (Fisher and others, 2001). In a survey conducted by Nelson and others (2000), 44.6 percent of respondents selected "Do not reduce current trail/route system and OHV access" to indicate the most important thing that should not be changed, and 30.1 percent selected "Develop more trails/routes/area and connections to services" to indicate the most important thing that should be changed. When provided with several OHV-management statements with which to agree or disagree, Crimmins (1999) found that "Most trail closures have been done for good reason" received the highest level of disagreement.

Similar patterns in attitudes and beliefs were revealed through a survey of 336 ATV and motorbike users conducted by the Idaho Department of Parks and Recreation (Achana, 2005). On a scale of 0-7 (from least to most serious), respondents were asked to rank 23 issues of concern to them. Results indicated that the most serious issues of concern (in descending order of seriousness; scores greater than 4) were

- permanent closure of an area the recreator uses most,
- temporary closure of an area the recreator uses most,
- inattentive/careless recreators engaged in motorized recreation,
- litter,
- too many rules and regulations, and
- poor communication of rules and regulations.

Conversely, respondents felt that issues they were not were not concerned with (in ascending order of seriousness; scores less than 3) were

- too few rules and regulations,

- inadequate facilities at campsites,
- ATV impacts on water,
- motorcycle impacts on water,
- problems with parking availability for OHV-support vehicles,
- lack of suitable campsites,
- ATV impacts on wildlife, and
- some other (unlisted) issue of concern in OHV use areas.

Issues of concern that fell in the middle (in descending order of seriousness) were

- inattentive/careless non-motorized recreators,
- OHVs traveling too fast,
- motorcycle impacts on soil,
- motorcycle impacts on vegetation,
- ATV impacts on vegetation,
- hunters on OHVs off designated roadways and trails,
- ATV impacts on soil,
- motorcycle impacts on wildlife, and
- noise from OHVs.

When asked which of 16 possible factors contributed to creation of unauthorized trails in recreational regions of Idaho, survey respondents selected (from most to least frequently)

- belief that OHV users should be free to go anywhere,
- lack of enough designated places to ride,
- avoidance of crowded designated areas,
- lack of operator experience,
- riding motorcycles for fun,
- treeless terrain,
- riding ATVs for fun,
- using ATVs for hunting access,
- lack of enforcement regulations,
- using motorcycles for hunting access,
- using ATVs for camping,
- inadequate regulation,
- using motorcycles for camping,
- using motorcycles for fishing access,
- using ATVs for fishing access, and
- some other (unlisted) regional resource impact.

Combined, the top three possible factors contributing to creation of unauthorized trails indicate that closures of OHV areas could result in at least local increases in dispersed use. Finally, when presented with a list of four alternatives for creating uniform OHV access requirements to all recreation areas, trails, and roads on Idaho public lands, 53 percent of the respondents selected the alternative “Open to OHVs unless posted as closed by signing,” and 33 percent selected the alternative “Open to OHVs unless posted as closed by signing, designation, or description.” Only 6.1 and 1.0 percent felt that areas should be “Closed to OHVs unless open by signing, designation, or description” or “Closed to OHVs unless open by signing,” respectively (6.7 percent did not respond to this question). These results are consistent with the top possible factors contributing to creation of unauthorized trails: the belief that OHV users should be free to go anywhere unless posted as closed by signing, designation, or description.

2.7.5 Economic Benefits and Costs of OHV Use

The economic benefits resulting from OHV sales, operation and maintenance, and associated sales and activities have been well documented (American Motorcyclist Association, 1978; Dave Miller Associates, 1981; Reed and Hass, 1989; Dean Runyon Associates, 2000; Nelson and others, 2000). OHV recreation and camping, in particular, can generate significant revenues for local economies through campground fees, grocery sales, eating and drinking in restaurants, and sales associated with operating and maintaining OHVs and support vehicles. In 1999, camping at public campgrounds on local, State, BLM, and U.S. Forest Service lands in California generated $500 million; an additional $130 million was spent solely on going to and from the campground and/or home (Dean Runyon Associates, 2000). A study conducted in 1988 by Reed and Hass (1989) indicated that, during a 12-month period in 1987-1988, Colorado OHV users spent $488.7 million on OHV purchases, operation and maintenance, support equipment (tow trailers, storage sheds, and so on), and travel expenses associated with OHV trips. Nelson and others (2000) reported that, between July 1998 and June 1999, the average Michigan OHV licensee spent $1,944 on non-trip related purchases, 80 percent of which was for equipment. When extrapolated to the estimated number of licensees in Michigan, Nelson and others (2000) found that this amounted to $134 million in spending on equipment; a similar extrapolation indicated that $40 million was spent on local trips.

The literature search conducted for this report, as well as personal communications with experts working in the field of outdoor recreation socioeconomics, revealed no published studies on the socioeconomic costs generated by OHV use. These costs could include the degradation or loss of ecosystem services, the costs of restoring OHV sites, and the loss of revenues from non-motorized recreators who seek alternate areas for recreation where motorized recreation does not occur. Examples of degraded or lost ecosystem services would be the diminished capacity for a given watershed to provide high-quality water, diminished water infiltration into aquifers, and flooding resulting from increased runoff where soils become compacted. Lost constituencies (and associated revenues) could include not only non-motorized recreators, but also hunters and anglers whose primary recreational foci (wildlife and fish) may have undergone population declines due to the effects of OHV use. At this time, however, the true benefit:cost ratio of OHV use remains unknown.

2.7.6 Annotated Bibliography for Socioeconomic Implications of OHV Use

Badaracco, R.J., 1976, ORVs—Often rough on visitors: Parks and Recreation, v. 11, no. 9, p. 32-35, 68-75.

This paper first reviews relevant literature on user conflict and discusses the one-sidedness of conflicts between OHV and non-OHV users, as well as the spatial nature of conflicts that occur when non-OHV users seek solitude and quiet and OHV users seek places for challenge and adventure. The paper then describes the ISD (impairment, suppression, displacement) syndrome: impairment is the diminished enjoyment among non-OHV users when they come into direct or indirect contact with OHV impacts; suppression is reduced participation of the non-OHV group; and displacement is the abandonment of a site impacted by OHV activity. Land planners and managers often misinterpret displacement as disinterest in the abandoned activity and, in so doing, may focus management efforts and other resources on OHV user demands.

Bury, R.L., and Fillmore, E.R., 1974, Design of motorcycle areas near campgrounds—Effects on riders and non riders: College Station, Texas, Department of Recreation and Parks, Texas A & M University, Technical Report, 72 p.

This document analyzes some of the psychological and sociological effects of constructing motorcycle riding areas adjacent to fixed-site campgrounds. It describes rider and camper profiles, rider and camper perceptions of riders, and camper and rider preferences and satisfactions with respect to the proximity and design of riding areas.

Cordell, H.K., Betz, C.J., Green, G., and Owens, M., 2005, Off-highway vehicle recreation in the United States, regions, and states—A national report from the National Survey on Recreation and the Environment (NSRE): U.S. Forest Service, Southern Research Station, Technical Report, 90 p.

This report was prepared for the U.S. Forest Service's National OHV Policy and Implementation Teams. The data from the NSRE were collected between the fall of 1999 and late 2004. The focus of this report is off-highway driving of motor vehicles. The 15 July 2004, U.S. Forest Service draft rule regarding management of motorized vehicle use has increased attention on where and how OHV recreation occurs and is offered. As public land managers are tasked with the responsibility of examining and implementing clear and consistent agency policy, understanding who the OHV recreators are has become ever more important. The growing use of motor vehicles is prompting the Forest Service to revise its management of this use so that the agency can continue to provide opportunities desired by the public, while sustaining National Forest System lands.

Crimmins, T., 1999, Colorado off-highway vehicle user survey—Summary of results: Denver, Colorado, Colorado State Parks, Technical Report.

This report summarizes a State Parks user survey designed to elucidate OHV rider-use patterns, what riders want in a recreation area, enthusiast values and beliefs, use of OHVs in hunting, how the state OHV fund should use the funds collected, and rider perceptions of how OHV funds are used, lands are allocated, and routes are managed.

Dave Miller Associates, 1981, An economic/social assessment of snowmobiling in Maine: Windham, Maine, Dave Miller Associates, Technical Report, 52 p.

This summarizes a user survey covering economics (number of trips, distance traveled, duration, fuel, lodging, equipment) and analyzing the statewide impacts and trends indicated by the responses. (No information on demographics or user perception was gathered.)

Dean Runyan Associates, 2000, Campers in California—Travel patterns and economic impacts: Portland, Oregon, Dean Runyan Associates, Technical Report, 76 p.

This document charts the distribution of camping opportunity according to type of environment and land ownership, tallies the results of a questionnaire distributed to people using public campgrounds, and develops a comprehensive profile of camping travel patterns, demographics, and expenditures. The report provides significant detail on a wide range of camping patterns, such as how many trips, how long and where, a breakdown of the activities pursued by campers once on site, and the ethnic and income classifications of campers. Although not OHV-specific, it shows where OHV recreation fits into the big picture.

Decker, D.J., Krueger, R.A., Bauer, Jr., R.A., Knuth, B.A., and Richmond, M.E., 1996, From clients to stakeholders—A philosophical shift for fish and wildlife management: Human Dimensions of Wildlife, v. 1, no. 1, p. 70-82.

This paper begins with a call for wildlife professionals to "adopt and use the term stakeholder," the development of which they review and the definition of which they indicate as being any citizen potentially affected by or having a vested interest in an issue, program, action, or decision leading to an action. The authors maintain that successful natural resource management in today's society requires recognizing the array of stakeholders that demand a voice or involvement in decision-making about natural resource management. The authors describe taking a stakeholder approach to planning and decision-making in natural resource management by including all those who might be impacted by natural resource management decisions (the authors focus on fish and wildlife management, but the principle is applied throughout natural resource management). The process entails developing communication strategies for understanding and representing stakeholder concerns, attitudes, and conflicts. The authors maintain that today's successful professional resource managers need to "…seek a widely recognized image of giving unprejudiced consideration to all significant stakeholder interests in management decisions."

Fisher, A.L., Blahna, D.J., and Bahr, R., 2001, Off-highway vehicle uses and owner preferences in Utah: Logan, Utah, Institute for Outdoor Recreation and Tourism, Department of Forest Resources, Utah State University, Report no. IORT PR2001–02, 80 p.

This study entailed an OHV user survey to examine owner characteristics, attitudes, and preferences. Respondents were selected at random from Utah OHV registrations and interviewed by telephone. This was a very extensive questionnaire, including the verbatim responses to interviewers' open-ended questions. Other questions included demographics, vehicle type used, where ridden, distance traveled, types of riding preferred, attitudes toward OHV program fund use, attitudes toward training and safety, and much more.

Jim, C., 1989, Visitor management in recreation areas: Environmental Conservation, v. 16, no. 1, p. 19–32.

This paper discusses various visitor-management measures for diminishing or precluding the effects of visitor impacts on natural resources in recreation areas by employing existing recreation-management research on visitor decisions—such as trip duration, difficulty, and desired environment—to suggest ways of dispersing use into patterns that do not result in damage to natural resources. It also examines various management scenarios: signs and maps to direct users into a managed pattern, restricting admission, lotteries, and various rationing/pricing concepts.

Kockelman, W.J., 1983, Management Concepts, *in* Webb, R.H., and Wilshire, H.G., eds., Environmental effects of off-road vehicles—Impacts and management in arid regions: New York, Springer-Verlag, p. 399–446.

Noise and motorized intrusion were the major impacts of ORVs on non-OHV users. Permitting OHV activity on public land is described as "inefficient" in the goal to provide for multiple uses because the noise, dust, and speed of just one OHV can exclude all other recreators from an area. The author categorizes OHV users as work-related users, recreational users, or

“bad apples.” Work-related users are natural resource managers and utility workers, among others. Recreators are further categorized as casual (value aesthetics more than the challenges of riding) or endurance riders. “Bad apples” are characterized by a complete lack of concern about their impacts and are likely to be noncompliant with regulations.

Nelson, C.M., and Lynch, J.A., 2001, A usable pilot off-road vehicle project evaluation: East Lansing, Michigan, Department of Park, Recreation and Tourism Resources, Michigan State University, Technical Report, 50 p.

This report details the results of an interagency effort to increase compliance with OHV rules in a Michigan State forest. An OHV-rider survey asked for respondents’ perceptions of signs, maps, and trail systems in the pilot area, as well as rider perceptions of any law enforcement contact riders may have had during the study period. The survey also queried each respondent’s understanding of pilot area regulations and offered the opportunity to give open-ended comments. There is also a detailed discussion of the participating law enforcement agencies’ response to the pilot project, including officer concerns, jurisdiction conflicts, workload distribution vs. agency priorities, and an analysis of sign survival in the pilot project areas. Finally, interviews with park manager/grant recipients and discussion of the results in terms of park administration, funding, staffing, and resource protection are provided.

Nelson, C.M., Lynch, J.A., and Stynes, D.J., 2000, Michigan licensed off-road vehicle use and users 1998–99: East Lansing, Michigan, Department of Park, Recreation and Tourism Resources, Michigan State University, Technical Report, 49 p.

This details a survey of randomly selected OHV owners in 1999. In addition to questions about demographics, expenditures, type of OHVs owned, and preferred activities, respondents were queried about their perceptions of specific aspects of the State OHV program. One section is dedicated to comparing this survey with a similar survey from 1988.

Propst, D.B., Shomaker, J.H., and Mitchekkm, J.E., 1977, Attitudes of Idaho off-road vehicle users and mangers: Moscow, Idaho, College of Forestry, Wildlife and Range Sciences, University of Idaho, Technical Report, 30 p.

This report provides background information on, and an introduction to OHV use in, the era when it was new and poorly understood, and includes one of the earliest OHV/OSV (over-snow vehicle) user surveys. It compares user and land manager responses in the same survey; both groups were queried about their perceptions of environmental impacts, causes of conflicts, uses of public money for facilities, regulation enforcement, impacts on wildlife, and reasons for pursuing OHV/OSV activities.

3.0 Potential Indicators for Evaluating and Monitoring OHV Effects

3.1 Summary

There are numerous parameters that have the potential for serving as indicators of OHV effects in monitoring or research programs. Every attempt was made to provide an inclusive list of potential indicators of OHV effects described in the OHV effects literature (listed below). Of those listed, some correspond with BLM’s 17 indicators of rangeland health; others are quite different but could provide supplemental data for evaluating or monitoring OHV effects (for example, erosion and/or sedimentation rates would complement assessments of rill formation and other surface changes) or fill indicator voids (such as those pertaining to wildlife ecology).

(1) **Soil health** and **watershed condition**

- Soil strength
- Soil bulk density
- Water infiltration rate
- Permeability
- Erosion and sedimentation rate
- Sedimentation or turbidity in wetlands
- Surface changes (for example, formation of rills, gullies, and terracettes)
- Presence/condition of soil crusts (in some cases: depending on crust type)

(2) **Vegetation health**

- Plant community composition (including species diversity, ratio of native to non-native or invasive species, structural diversity)
- Abundance of individuals and/or stem density
- Percent vegetation cover
- Plant size
- Growth rate
- Biomass

(3) **Habitat condition** and **health of wildlife populations** (direct and indirect)

- Habitat patch size and connectivity
- Wildlife community composition (including species diversity, ratio of native to non-native or invasive species)
- Abundance, density, and distribution
- Population sizes and trends
- Survivorship, productivity, body mass, and roadkill rates
- Age-class and gender structure
- Frequency of OHVs passing through a given area
- Road or trail type and width
- Level (decibels), duration, and timing of traffic noise

(4) **Water quality**

- Sedimentation rate
- Levels of turbidity and suspended solids
- Contaminants levels, including levels of petroleum-derived compounds from spills (aromatic hydrocarbons in particular)

(5) **Air quality**

- Dust levels
- Levels of by-products of OHV emissions (including polycyclic aromatic hydrocarbons, carbon monoxide, nitrogen oxides, ozone, and sulfur dioxide)

(6) **Socioeconomics** (direct and indirect)

- Recreator satisfaction with their recreation (or other) experiences
- Compliance with OHV (or other) regulations
- Knowledge regarding effects of user activities on various aspects of land health

- Mapping the distribution and intensity of OHV versus non-motorized recreation and other land uses,
- Patterns of regulation compliance (as evidenced by creation of unauthorized trails, damage to vegetation, and so on)
- Trends in local economic indicators associated with OHV and non-motorized recreation and other land uses (for example, sales in camping equipment, gasoline, restaurants, lodging facilities)

Specific research questions and management goals—as well as sensitivity to OHV effects and the availability of funding and personnel—will determine the potential efficacy of using any one indicator to evaluate or monitor OHV effects on BLM lands. Qualitative indicators may be most useful for rapid assessments, whereas quantitative indicators may be needed for long-term monitoring. Ultimately, however, implementing an OHV effects monitoring program will require consultation with topical experts and additional research to identify or develop appropriate and efficient indicators and field methods for evaluating and monitoring OHV effects (personal communication from D.A. Pyke to Z.H. Bowen, U.S. Geological Survey, Fort Collins, Colorado, August 2007). Work on developing such indicators is currently underway by rangeland ecologist, D.A. Pyke, U.S. Geological Survey in Corvallis, Oregon.

3.2 BLM's Indicators of Land Health Compared to Indicators of OHV Effects Described in the Literature

In terms of the specific land health attributes assessed, there is some limited correspondence between several indicators of OHV effects described in the literature and some of BLM's 17 qualitative assessment indicators (see Pellant and others, 2005). The area of greatest overlap is that of soil health and watershed condition; there is somewhat less overlap in the area of vegetation health (table 3.1). Attributes addressed in the literature but not by BLM's 17 indicators include wildlife population and habitat health, water and air quality, and socioeconomics. Even indicators that measure the same or similar attributes, however, may differ notably with respect to the scale and scope to which they are or can be applied, or the precision and accuracy they can provide.

The differences between BLM's indicators and those described in the literature do not imply that BLM's indicators are inappropriate for assessing some attributes under some conditions. Rather, they underscore the need for a variety of indicators to meet equally variable needs. For example, qualitative indicators (such as those employed by the BLM) often entail making visual estimates, which may be suitable for rapid assessments by time-limited personnel operating with small budgets; qualitative measurements, however, are subject to observer bias. Research and monitoring studies, on the other hand, generally require quantitative indicators (or strict decision rules to guide data collection for qualitative parameters) that minimize observer bias and maximize statistical precision and accuracy to ensure defensible results and the detection of trends; quantitative measurements, however, can drive up the cost and time requirements of research and monitoring efforts. Therefore, the choice of indicators employed will depend on the specific goals, budgets, sites, and other factors. In some cases, BLM's indicators may be suitable; other cases may require more quantitative indicators. Co-opting indicators from other disciplines also may be extremely useful for revealing OHV effects on land health.

Table 3.1. Indicators emphasized in the literature reviewed for effects of off-highway vehicles (OHV) on land health compared to indicators of land health employed by the U.S. Bureau of Land Management (BLM) (Pellant and others, 2005).

Land health category	Indicators of OHV effects described in reviewed literature	BLM indicators[a]
Soils and watersheds	Soil strength	9,11*
	Soil bulk density	11
	Soil permeability	8,9*,11*
	Water infiltration rate	1,2,3,4,5,8*,9*, 10*,11*
	Erosion rate	1,2,3,4,5,6,7, 8*,10*
	Sedimentation rate	
	Presence/condition of biotic and abiotic soil crusts	2,4,6,7,8*
Vegetation	Plant species diversity	12*,13,16*
	Ratio of native plants to non-native and/or invasive plants	12,16
	Percent plant cover	4,13,14
	Plant size	11,13,14, 15*,17
	Plant growth rate	11*,13,14,15*, 17
Wildlife and habitats	Habitat patch size and connectivity (can be expressed as ratio of road edge:habitat area or as native:non-native habitat)	4,12*,16*
	Shape/scope of animal movements relative to roads	
	Wildlife diversity and/or species abundance	
	Ratio of native, endemic wildlife to non-native and/or invasive species	
	Population size and trend	
	Gender/age ratio trend	
	Productivity trend	
	Average body mass for a given age/gender	
	Average survivorship	
	Vehicle-caused mortality rate	
Water quality	Sedimentation rate or depth	
	Amount of suspended solids	
	Turbidity level	
	Level of atmospheric deposition associated with OHV emissions	
	Level of petroleum or its by-products (benzene, ethylbenzene, toluene, xylenes, 1,3-butadiene, lead)	
Air quality	Levels of OHV emission by-products (nitrogen oxides, carbon monoxide, sulfur dioxide, ozone, aldehyde, PAHs[b])	
	Level of suspended particulates	
	Plant-absorbed level of emissions by-products	

[a] 1 = Number and extent of rills.
2 = Presence of water flow patterns.
3 = Number and height of erosional pedestals or terracettes.

4 = Bare ground (excluding rock, litter, lichen, moss, plant canopy).
5 = Number of gullies and erosion associated with gullies.
6 = Extent of wind scoured blowouts and/or depositional areas.
7 = Amount of litter movement (description of size and distance expected to travel).
8 = Average soil surface (top few mm) resistance to erosion.
9 = Soil surface structure and content of soil organic matter (to include type of structure and A-horizon color and thickness).
10 = Effect of plant community composition (relative proportion of different functional groups) and spatial distribution on infiltration and runoff.
11 = Presence and thickness of compaction layer (usually none).
12 = Functional/structural groups (in descending order of dominance by above-ground production or live foliar cover).
13 = Amount of plant mortality and decadence (include which functional groups are expected to show mortality or decadence).
14 = Average percent and depth of litter cover.
15 = Expected annual production (this it total above-ground production, not just forage production).
16 = Potential invasive (including noxious) species (native and non-native).
17 = Perennial plant reproductive capability.

[b] Polycyclic aromatic hydrocarbons.

Ultimately, **any indicator used to evaluate or monitor OHV effects will need a standard value (or range of values) that represents the baseline condition or a threshold value above (or below) which management is triggered**. Ecosystem properties, however, can vary widely across (and within) **spatial scales**. Therefore, selecting an appropriate standard or threshold for any one indicator necessitates an evaluation of the spatial scale(s) at which the associated effect occurs or is likely to affect ecosystem function. For example, OHV-caused sedimentation in a first-order stream might have significant by very localized effects on that stream, but no significant effects downstream; in contrast, OHV-caused sedimentation in a first-order stream might not have significant effects on that stream, but sedimentation in multiple first-order streams may result in significant cumulative downstream effects in second- and third-order streams. Similarly, the **temporal scale** at which ecosystems respond to land uses can vary from short-term (hours) to long-term (decades or longer); thus, it is important to consider temporal scale(s) as indicator standards or thresholds are established. For example, wildlife behaviors may exhibit immediate responses to OHV traffic by moving away from OHV routes; population trends, however, may take several generations to show effects of OHV disturbance, in which case the time required to detect trends will depend on how long it takes for generations turn over. Where OHV effects are not an immediate concern, one could incorporate long-term, annual, qualitative assessments of OHV-use areas, but where effects are of immediate concern, short-term, quantitative assessments may be implemented.

Standards for a given land health indicator can be developed by ascertaining baseline values at control sites under a given set of conditions. Once the baseline values are established, it becomes important to standardize the conditions under which subsequent measurements of that indicator are made. For example, the freezing and thawing of soils tend to decompact soils, and precipitation tends to alter other soil properties; thus, it would be important to monitor soils at similar times of year and under similar soil-moisture conditions.

3.3 Some Potential Indicators for Evaluating and Monitoring OHV Effects

3.3.1 Potential Indicators of OHV Effects on Soils and Watersheds

A major effect of OHV traffic on soil health is compaction—the reduction of a soil's porosity. Potentially useful indicators for monitoring soil compaction include soil strength, soil bulk density, and infiltration rate (or permeability). **Soil strength** is typically measured in terms of the soil surface's resistance to a vertical force exerted by a penetrometer and is expressed as kg/cm^2 (see *http://cropsoil.psu.edu/extension/facts/agfacts63.cfm* for explanations and diagrams of this method). **Soil bulk density** is measured as the ratio of dry solid mass (after soil is over-dried) to bulk volume of soil and is expressed as g/cm^3 (or Mg/m^3). **Infiltration** or **permeability** is the rate at which the water infiltrates the soil and is expressed as cm/hr. Infiltration tests can be conducted in the field with relatively simple equipment (for a demonstration, see *http://www.grow.arizona.edu/Grow--GrowResources.php?ResourceId=181*); however, many replicates are needed to obtain adequate comparisons, and it takes time for infiltration to occur. Additional resources on the utility of devices and techniques for monitoring soil health include Leung and Meyer (2003) for soil compaction; O'Sullivan and Ball (1982), Hooks and Jansen (1986), Komatsu and others (1988), and Keener and others (1991) for soil strength; and Flint and Childs (1984), Isensee and Luth (1992), Miller and others (2001), and Lowery and Morrison (2002) for soil bulk density. McBrayer and others (1997) and Amezketa Lizarraga and others (2002) describe techniques used to measure infiltration rates. Overall, soil strength and bulk density are the most commonly used indicators of soil compaction in visitor impact studies (Liddle, 1997).

Soil strength, which increases with increasing soil compaction, depends on a number of inherent variables, such as particle size, type of clay mineral, the size/distribution of pores, and aggregate stability. For example, Adams and others (1982) found that soil strength in undisturbed areas of the Mojave Desert ranged between 5.1 kg/cm^2 at 6 percent water content and 21.1 kg/cm^2 at 1.8 percent water content, indicating that soil strength decreases with increasing moisture content. In general, a soil strength of less than 20 kg/cm^2 (284.4 lb/in^2) was indicative of undisturbed terrain, whereas trails intensely used by motorcycles had soil strengths (during wet conditions) that typically ranged from 20 to 60 kg/cm^2 (284.4 to 853.2 lb/in^2) (Adams and others, 1982). When soil strengths exceeded 20 kg/cm^2 (284.4 lb/in^2; measured at about field capacity) due to compaction, Grimes and others (1975, 1978) found that root extension of certain plants, including alfalfa (*Medicago sativa*), corn (*Zea mays*), and cotton (*Gossypium hirsutum*), was limited.

Soil bulk density also increases with increasing soil compaction. For example, in the Mojave Desert, Webb (2002) and Caldwell and others (2006) found that bulk densities of undisturbed surface soils ranged from 1.40 to 1.68 g/cm^3; however, bulk density increased significantly to 1.80 g/cm^3 at high-disturbance sites (Caldwell and others, 2006). Measuring soil bulk density, however, is time-consuming and difficult in gravelly soils (Webb, 2002), which are typical of many desert sites. Thus, its practicality may vary not only from region to region, but from soil to soil within a region. Infiltration, on the other hand, decreases with increasing soil compaction. In undisturbed desert habitats, Eckert and others (1979) found soil infiltration rates to be 3.2 cm/hr. In the same study, infiltration rates were 15 percent lower where motorcycle (2.7 cm/hr) and 33 percent where truck (2.1 cm/hr) traffic had occurred.

For each of the indicators identified above, it will be crucial to consider soil type and water content when setting standards, as these factors clearly influence overall inherent values of each parameter. It also is clear that variability in soil type may preclude using a single indicator—much less one standard—for monitoring soil health across all sites. Overall, monitoring soil health would entail measuring soil properties in tracked versus untracked areas of similar soil type and under similar conditions of soil water content. For areas previously unaffected by OHV activities but proposed to become OHV areas, managers may wish to collect predisturbance data on soil properties to serve as a baseline from which trends in soil properties may be assessed over time. An acceptable percent change in soil properties could be selected, and if soil properties were to exceed these threshold values then management actions could be implemented to bring the soil properties back to acceptable values.

A possible method of monitoring surface changes in watersheds might be to establish permanent monitoring sites for repeat photography studies. By standing in the same spot, orienting the camera in the same direction, and using the same focal length for each site visit, sequential photographs would provide a qualitative, relatively easy means of monitoring surface changes due to erosion and OHV tire cuts. Helpful websites that discuss repeat photography (both terrestrial and aerial) include *http://biology.usgs.gov/luhna/chap9.html*, *http://wwwpaztcn.wr.usgs.gov/wyoming/rpt_ground.html*, and *http://www.cpluhna.nau.edu/Tools/repeatphotog.htm*. Scientists with the National Science and Technology Center are using close-range (terrestrial or ground-based) photogrammetry techniques and associated tools for producing and interpreting close-range images (less than 300 m, as opposed to aerial photogrammetry distances of more than 300 m) that may be used to document soil crusts, soil erosion, vegetation, and other resources (see *http://www.blm.gov/nstc/prodserv/ST134/pdf/Handout3CloseRange.pdf*).

3.3.2 Potential Indicators of OHV Effects on Vegetation

Some plant responses to OHV activities are preceded by, and result from, OHV effects on soil properties. In other words, soil characteristics, particularly soil compaction, play important roles in the distribution, abundance, growth rate, reproduction, and size of plants. Furthermore, some plant responses are likely to lag behind changes in soil properties, and, by the time effects are detected in plants, site recovery could be more difficult and/or lengthy. As such, it would be important to implement management strategies for maintaining or improving soil condition before plants are affected and no longer provide enough cover to hold soils in place during restoration efforts. Several vegetation parameters, however, have potential value as direct indicators of OHV effects on plant communities.

As mentioned in section 3.3.1, soil compaction results in reduced water infiltration. As a result, overall plant productivity, as measured by **percent plant cover**, **abundance** or **stem density**, **growth rate**, **biomass**, **plant height** and **width**, **ratio of large:small species**, and/or **reproductive output** may be diminished (Johnson and others, 1975; Vasek and others, 1975; Adams and others, 1982; Webb, 1983; Prose and others, 1987; Holzapfel and Schmidt, 1990; Lightfoot and Whitford, 1991; Brooks, 1995; Bolling and Walker, 2000). It is important to consider, however, the differential growth habits and responses of plant species to conditions generated by OHV activities; they may be favored or inhibited by OHV effects (Holzapfel and Schmidt, 1990; Angold, 1997). For example, the productivity of some species may increase due to abnormal conditions of moisture availability from runoff near compacted areas and/or where water roadbed materials allow increased infiltration rates (Johnson and others, 1975; Vasek and others, 1975; Holzapfel and Schmidt, 1990; Lightfoot and Whitford, 1991).

To evaluate and monitor plant productivity, researchers often use transect-intercept methods for measuring larger- or site-scale indicators, such as percent cover, and quadrat-based sub-sampling at random locations along transects for measuring smaller-scale (individual plants) indicators, such as plant size or growth rate. Repeat photography is also potentially useful for monitoring vegetation cover when personnel budgets are limiting, although ground-based repeat photography methods may be more realistic than satellite imaging in terms of staff expertise and funding required (see methods and URLs provided in Section 3.3.1). Satellite imagery, however, can be very effective for ascertaining landscape-scale changes in vegetation cover during long-term studies at selected study sites (Johansen and others, 2007).

Plant community composition or diversity—a parameter that factors relative proportions of each species into species richness—is a commonly used indicator health of a vegetation community (Davidson and Fox, 1974; Adams and others, 1982; Prose and others, 1987; Wilcox, 1989; Tyser and Worley, 1992; Lovich and Bainbridge, 1999; Parendes and Jones, 2000). For example, a plant community comprising 5 individuals each of 10 species and 150 individuals of another (200 total individuals) would be considered depauperate compared to a community comprising 20 individuals each of 10 species (200 total). Native species diversity is further compromised when non-native and/or invasive plants dominate the plant community (Holzapfel and Schmidt, 1990). OHVs caked with mud acquired elsewhere potentially introduce or disperse seeds of non-native and invasive species; thus, OHV-route margins often become populated with exotics and invasives that eventually may spread and outcompete native species at the landscape level. Therefore, an important consideration when evaluating plant species diversity is the presence of non-native and/or invasive species. In other words, although species diversity can be useful for evaluating and monitoring the impacts of OHV activities on vegetation, the ratio of native to non-native and invasive plant species must be taken into account. Monitoring transects oriented perpendicular to OHV travel routes would help identify range expansions beyond the linear routes.

Similar to the precautions issued above for selecting standards of soil health, vegetation characteristics also vary widely according to soil type, slope, aspect, microclimate, and other factors. Ultimately, having baseline data before a site is disturbed, and/or having nearby reference sites not subjected to OHV activity would help differentiate between site-based variations and OHV impacts on vegetation.

3.3.3 Potential Indicators of OHV Effects on Wildlife and Habitats: Native, Threatened, and Endangered Species

The physical imprint of a road or OHV route creates barrier effects that may effectively alter **habitat patch size and connectivity**, which potentially alters or inhibits **animal movements** (Oxley and others, 1974; Mader, 1984; Swihart and Slade, 1984; Samways, 1989; Andrews, 1990; Baur and Baur, 1990; Forman and Alexander, 1998; Jackson and Griffen, 1998). Roads and trails also create edges that can alter wildlife habitat use and movements (Nicholson, 1987; Yahner, 1988; Reed and others, 1996; von Seckendorff Hoff and Marlow, 1997; Gibbs, 1998; Vos and Chardon, 1998), and in many instances these **edge effects** extend well beyond the road's actual footprint into habitat interiors. Unfortunately, both direct and indirect indicators of animal movements and habitat or home-range use can be difficult to measure, simply because populations are mobile and easily affected by many factors besides OHV activity if care is not taken to avoid, or control for, these additional effects. Furthermore, adequate sample sizes and accurate measurements of animal movements generally require capture-mark-release or capture-mark-resight studies, which can be costly in terms of funding for equipment and personnel time. Therefore, evaluating and monitoring movement responses of suitable indicator species or functional groups to OHV activities and routes might be more efficient than trying to monitor many species. For example, species least likely to cross OHV routes or networks of routes (beetles or small mammals, for example) would be more suitable for studying barrier effects of OHV routes than species inhibited from crossing only multi-lane, paved highways (large ungulates). Ultimately, animal movement studies might be most appropriate in comprehensive, long-term research projects at selected study sites representative of locations and habitats being impacted most by OHV activities. An appropriate measure of edge effects on wildlife may entail studies of spatial distribution relative to distance from OHV routes, as well as density and other population dynamics among target indicator species that require habitat interiors.

Traffic intensity and noise level of OHVs may be useful indirect measures of OHV effects on wildlife behavior and survivorship. If measured, temporal variation in animal activity or presence would need to be considered, as diurnal animals may be more affected by OHV traffic and noise than nocturnal animals if most OHV activity occurs during daylight hours (Ouren and Watts, 2005). Vehicle speed also may be an important parameter to measure, as it can affect mortality rates of animals as they attempt to navigate landscapes where OHV travel is significant. There are technologies available for field monitoring of noise levels and traffic volume and speed (see examples on the Internet at *http://www.jhuapl.edu/ott/technologies/technology/articles/P01254.asp* and *http://www.noisemeters.com/accessories/outdoorkits.asp*), although again it would be more cost-effective to use them in selected study sites for targeted research questions as opposed to broad-scale monitoring programs.

Ideally, any program for monitoring OHV effects on wildlife would include assessments of whether/how wildlife responds to OHV-related factors. If population dynamics were understood *before* the onset of OHV activities, then monitoring any changes that occur afterward could be straightforward if other conditions are held relatively constant. Climate, however, typically makes long-term monitoring of animal population dynamics very complex, and wide-ranging animals may be more difficult to monitor than those with smaller or more restricted movements. Similar to the dynamic ways in which wildlife populations use a given area, OHV use in a given area is also dynamic; if the area affected and the intensity of disturbance enlarges considerably, wildlife may respond as well. These and other factors typically drive up the funding and staffing needs associated with collecting population dynamics data. Thus, to determine thresholds of OHV activity potentially tolerated by wildlife, it may be more realistic to measure long-term changes relative to gradients and changes in OHV activity. Again, careful selection of suitable indicator species, or functional groups, may help increase the effectiveness of monitoring wildlife populations (Lindenmayer, 2000; Noon, 2003, p. 51-55).

To date, there have been few simultaneous studies of OHV use and wildlife responses to OHV activities, but such studies could provide more precise assessments of the relationship between patterns of OHV activity and wildlife responses, particularly behavioral responses to varying traffic patterns, intensity, and total area affected. GPS and satellite technologies could be employed to build Geographic Information System (GIS) data layers of OHV-associated variables, both static (for example, road width) and dynamic (vehicle speed and traffic volume). When overlaid with GPS-based telemetry data layers that map animal movements, one could relate changes in static and dynamic OHV variables to wildlife responses, making these particularly powerful tools for long-term studies aimed at evaluating OHV impacts on wildlife.

3.3.4 Potential Indicators of OHV Effects on Water Quality

The literature on OHV impacts offered little information specific to evaluating or monitoring OHV effects on water or air quality. Based on studies of water quality in other disciplines, however, potentially useful indicators highly relevant to OHV effects would be **sedimentation** and **turbidity**. Sedimentation can be measured in terms of deposition rate or total amount of solids deposited where surface and directed flows enter aquatic systems downslope of OHV-use areas. A useful reference on different methods for measuring sedimentation may be found in Lisle and Eads (1991). Turbidity, which indicates the level of **suspended solids** in water, can be measured easily in the field with a Secchi disk (see *http://www.noble.org/Ag/Wildlife/SecchiDisk/Index.htm*). With respect to monitoring levels of contaminants in water from OHV emissions and fuel or other chemical spills, water samples can be collected and analyzed in laboratory settings; however, this can be costly, and would probably be more suitable for selected, long-term research sites than in broad-scale monitoring programs. Potentially important contaminants to test for in OHV-impacted watersheds could include **benzene**; **ethylbenzene**; **m-, p-, and o-xylene**; **toluene**; **1,3-butadiene**; **and lead** (Forman and others, 2003: p. 205-213; Arnold and Koel, 2006). In addition, nitrogen deposition from nitrogen oxides can affect water quality if nitrogen loading alters the chemical balance of nutrients in aquatic organisms.

Although it has already been stated that, in general, it is important to compare OHV-impacted sites with similar reference (unaffected) sites when evaluating OHV effects, an additional consideration with respect to water quality is to make comparisons within the same drainage. This is because water quality variables can change depending on the geology, adjacent habitat, and hydrology of the area. Thus, an appropriate technique for assessing OHV effects on aquatic systems might be to compare water quality at replicate sites both upstream and downstream of where OHV activities are occurring.

3.3.5 Potential Indicators of OHV Effects on Air Quality

There are several measures of air quality that can be used to assess effects of OHVs, including levels of fugitive dust (**suspended particulates**) and/or **OHV emissions** (including **carbon monoxide**, **ozone**, **sulfur dioxide**, **aldehyde**, and **polycyclic aromatic hydrocarbons**). Due to the effects of humidity, precipitation, fallout rates of different particle sizes, and wind speed and direction, however, measuring dust levels specific to any one site or set of OHV activities can be difficult. A useful technique for assessing the amount of dust associated with OHV use is to collect PM_{10} (particulate matter less than 10 microns in diameter, which can pass through the nose and throat and get deep into the lungs) data, as dust is a common component of PM_{10}. Technological advancements continue to provide additional devices useful for monitoring dust and other suspended particulates (Sanders and Addo, 2000)—including satellite imagery (Nakata and others, 1976; Gill, 1996; Stefanov and others, 2001)—although cost becomes a greater factor with increasing sophistication of instruments used. Fox (1986) provides a list of appropriate procedures for measuring and evaluating various air-quality parameters, and the Environmental Protection Agency’s National Ambient Air Quality Standards include primary standards (those designed to protect overall public health and the health of “sensitive populations”) for carbon monoxide, lead, nitrogen dioxide, particulate matter, ozone, and sulfur oxide levels (see *http://www.epa.gov/air/criteria.html*).

3.3.6 Potential Indicators for OHV Effects on Socioeconomics

Resource planning has been known to take recreation and economic values of OHV use into greater consideration than biological considerations (Adams and Dove, 1989). **Human behaviors, attitudes, and economics**, however, are the ultimate drivers of OHV impacts on natural resources (Decker and others, 1996; Vaske and others, 2001); thus, understanding the socioeconomics of OHV use is crucial to the success of any program designed to address OHV effects, whether they impact natural resources or land users. This includes understanding the economic effects—**both benefits and costs**—of OHV management and use on OHV users, other land users, local businesses, land-management agencies, and ecosystem services.

Bight and others (2003) provide a framework and guidelines for conducting social assessments and identifying/organizing social science data, including measurable indicators (see Chapter 3, p. 21) for use in natural resource planning. Indicators suitable for monitoring the socioeconomic implications of OHV use (human behaviors, attitudes, and economics) may be identified through stakeholder interviews, focus groups, and surveys (mail or telephone); economic assessments; and developing maps depicting areas used for different forms and intensities of recreation (Massachusetts Department of Environmental Management, 1995; Decker and others, 1996; Stokowski and LaPointe, 2000; Nelson and Lynch, 2001; Dillman, 2007). For example, interviews can be designed to elucidate not only OHV impacts on all types of user experiences, but also ways in which those impacts might be mitigated. Trail openings or closures may affect levels of OHV user demand and satisfaction, which could be identified through well-designed survey questions that target OHV users (see Dillman 2007).

Likewise, understanding the effects of OHV site development and regulation compliance first requires monitoring where OHV users like to go and what their preferences are when riding. For example, Nelson and Lynch (2001) conducted a study to determine the effectiveness of OHV areas, the approaches for which included a survey of licensed OHV riders, interviews with key stakeholders about project management, and an assessment of the signs established to identify designated OHV trails. Monitored over time, these indicators could be used as guidelines for adaptive management. Finally, identifying the economic impacts associated with OHV activities and regulations potentially affecting nearby communities may entail economic analyses of businesses and services that cater to outdoor recreators (including OHV users and non-motorized recreators; English and others, 2001). Economic assessments also may be used to determine the financial effects associated with losing ecosystem services, regulation enforcement, resource restoration, and other potential costs of OHV use.

A potentially powerful tool for identifying socioeconomic effects of OHV use would be GIS applications (Massachusetts Department of Environmental Management, 1995; Stokowski and LaPointe, 2000; Kopperoinen and others, 2004). For example, areas considered by recreators as being most important for excluding motorized forms of recreation could be identified by users on a map and then used to develop time-series GIS data layers that illustrate long-term changes in preferences pertaining to land-management actions or other factors. Overlaid with maps that identify areas ecologically most suitable for OHV use, managers also could determine which areas are most likely to be resilient to OHV use and provide OHV user satisfaction.

4.0 Mitigation and Site-Restoration Techniques

4.1 Summary

Mitigation of OHV effects and restoration of OHV-impacted sites requires a range of approaches and techniques. Social science in particular has strong applicability for ameliorating the effects of OHV activities, not only in terms of their impacts on non-motorized recreators and other OHV users, but also in terms of their effects on natural resources, as ultimately human behavior is what drives OHV use and related behaviors. Important tools for managing OHV use, therefore, include not only interviews, surveys, and focus groups, but also strong educational campaigns. Once impacted by OHV use, however, ecosystems may need to be closed and rested, if not restored. Sites with severely compacted soils and/or bedrock exposures due to erosion may need restoration through importation of native soils, scarification, decompaction, stabilization, inoculation with microbes and mycorrhizae, and/or mulching before reseeding and/or planting can be done.

4.2 Mitigation and Site-Restoration Techniques

The OHV literature has addressed many effects of both motorized and non-motorized recreation on components of ecosystem health (Cole, 2004; Stokowski and LaPointe, 2000; Cline and others, 2007), including soils, vegetation and habitat, wildlife behavior and population dynamics, and the quality of water and air. Resource degradation and wildlife disturbance are not uniform, however. Factors influencing the extent and degree of impact include user types and behaviors, the environment’s resistance and resilience, and the timing, intensity, and distribution of use (Cole, 2004). Therefore, management and mitigation planning and implementation must take these factors into consideration.

4.2.1 Understanding Land User Preferences and Conflicts

Although addressing ecosystem degradation and user conflicts stemming from OHV use generally requires policy and management considerations (Vancini, 1989), there is the likelihood that one group or another will be dissatisfied with the outcome of any one management decision. Therefore, it is important to promote acceptance of, if not support for, management decisions prior to implementation, because those not accepted are likely to fail in the long run, regardless of how sound the reasoning is behind them (Shindler and others, 2004). One way to potentially improve policy acceptability is to identify, assess, and include all users who may be affected by management decisions and include them in the decision-making process; useful tools for accomplishing this goal include stakeholder and demographic analyses, and arranging stakeholder focus groups (Decker and others, 1996; National Oceanic and Atmospheric Administration, 2005). Basic discussions, such as defining different interpretations of OHV use or seeking consensus on the meaning of OHV use to disparate users, can potentially preclude feelings of marginalization by any given group that could lead to conflict between users and/or between users and managers (Stokowski and LaPointe, 2000).

A baseline understanding of OHV users (including recreators, livestock operators, and energy-development operators) can further help to alleviate conflicts among different users. If users can be classified, even broadly, managers can maximize their efforts by using that information to form relevant management plans and/or communication and education campaigns. For example, understanding the problems that users have with each other, as well as their motivations for recreating, can yield more effective management that placates most, if not all stakeholder groups, and does not lead to marginalization or displacement of any one group. Often, outdoor recreators of all types have fairly similar goals and reasons for participating in outdoor recreation (Schuett and Ostergren, 2003), including a need to “get away” from the pressures and commotion of everyday life, rejuvenation, and enjoyment of the natural environment. Even when the mechanisms by which those needs are met are not homogeneous among user groups, knowledge of stakeholder preferences can help to improve the management of OHV recreation on public lands.

Managing the social effects of OHV activities and promoting compliance with regulations also require determining where OHV users like to go and what their preferences are when riding (Nelson and Lynch, 2001). Likewise, management must consider the levels of environmental dominance and technology associated with different forms of land use. Once again, GIS data layers could be developed to identify spatial management needs and predict where conflicts may occur. For instance, nature study, an activity characterized by “low dependence on technology and low dominance over the environment,” and OHV touring, which is characterized by “high dependence on technology and high levels of environmental dominance,” need to be segregated spatially to help prevent user conflicts. If it is found on a map grid that two such activities are taking place in adjacent quadrats, there is a high likelihood that recreators who participate in these activities will come into conflict (Bury and others, 1983). Maps of sites suitable for a given activity could be overlaid with user preferences for certain locations or landscape features to further fine-tune planning and mitigation of user conflicts.

Finally, if recreators are made aware of their impacts and understand the implications of those impacts, they may be more willing to take steps that lessen those effects, thereby diminishing the necessary level of managerial monitoring (Vancini, 1989; Anderson and others, 1998). Communication and education campaigns can be difficult, however, due in part to (1) modern-day information overload, increasing the likelihood that recreators will disregard information pertaining to natural resources and recreation, and (2) the fact that managers and designers of education campaigns rarely have a formal knowledge and understanding of persuasion techniques (Absher and Bright, 2004). Assuming that these factors can be surmounted by well-designed educational resources and manager training, education is a crucial first step in alleviating negative perceptions of trail closures and other regulations or management actions. For example, management agencies need to explain the reasons and rationale for closures so that they do not appear arbitrary (Crimmins, 1999). Such educational campaigns may be more successful if they target the many OHV users participating in some other form of outdoor recreation that depends on a healthy ecosystem. Hunting and fishing are two such forms of recreation, and if an OHV area is closed seasonally to protect elk during calving season or fish during spawning season, communicating this to OHV users is likely to elicit more compliance with the closures. When OHV users understand that their hunting and fishing activities may be at stake, they are more likely to respect closures. In closed areas already subjected to high levels of illegal riding, education would be crucial for communicating to users the reasons for the closures and related management actions.

4.2.2 Mitigating OHV Use Effects

Lands managed by BLM are placed into one of three broad types pertaining to OHV-use designation: “open,” “limited,” or “closed.” Within the “limited” category there can be several types of limitations: OHVs can be “limited to existing routes,” “limited to designated routes,” or “limited seasonally” (see *http://www.blm.gov/nhp/news/releases/pages/2000/pr000110_ohv_qa.html*). Indeed, trail/area closures are among the management options available for allowing soils and vegetation to recover from OHV effects or to help preclude localized impacts on air and water quality. However, in areas where OHV effects would be notable with the first few uses and/or generate significant, long-lasting impacts, it would be prudent to consider whether such places are appropriate for OHV use in the first place (Cole and Landres, 1995; Cole, 2004). A well-placed, concentrated system of trails could alleviate the difficulties of enforcement that extend across a large territory (Major, 1987). Spatial models, including GIS data layers, developed for identifying areas most suitable for resisting or recovering rapidly from OHV impacts, or those most suited for concentrated and/or self-monitored OHV use, would be effective tools for establishing OHV sites in appropriate areas and avoiding the need for frequent, long-term closures, expensive restoration actions, and/or the high cost or enforcement monitoring. For example, erosion, sedimentation, and rill or gully formation are much more likely to occur in, and downslope of, OHV-use areas located on or at the top of a slope than in/from a flat site or depressional area lacking watershed outlets. The GIS data layers could, therefore, identify areas to exclude from development for OHV use.

Although trail/area closures may be among the easiest management actions to implement, they may prove difficult to enforce, particularly under a policy of "closed unless posted open." The enforcement difficulty will depend on the number of trails involved, their locations relative to one another, and the number of enforcement personnel available. In the absence of funding for adequately monitoring regulation compliance, educating the public about the effects of their recreational pursuits may prove more economical and yield self-monitored trail users. In areas affected heavily by recreational activities, however, visitor management may be required (Jim, 1989). "Rationing" is a visitor management strategy that can be used to control the number of visitors over a given area where the available recreational resources are finite and/or unique. If the number of visitors is restricted, their quality of experience may be greater; a drawback of this approach is that the benefit is realized by fewer users (see Dimara and Skuras, 1998). Other strategies may include reserving a permit in advance, a permit lottery system, or implementing user fees that reflect the quality of experience (for example, it may cost more to use a high-use area that offers a high-quality experience than a low-use area that offers a low-quality experience). In many cases, these strategies could help alleviate pressures on law enforcement.

Wildlife is affected by OHV recreation in numerous ways, including displacement caused by human disturbance or direct mortality caused by vehicle-animal collisions (Cole and Landres, 1995; Knight and Cole, 1995; Miller, 1998; Stokowski and LaPointe, 2000; Cline and others, 2007). As OHV-related landscape fragmentation increases, habitat area and required juxtapositions of habitat types that meet the different needs of a given species, as well as adequate cover from disturbance, are diminished. Because the resulting effects may be detrimental to individuals and/or local populations (Knight and Cole, 1995), mitigation and management may be required to protect wildlife. One approach for mitigating the effects of wildlife displacement and disturbance may be to control the spatial and/or temporal proximity of OHV activities to wildlife, especially during critical nesting and breeding times (Gutzwiller, 1995). Again, GIS data layers could be very useful for identifying crucial wildlife areas and ensuring that they are not overlapped, fragmented, or otherwise disturbed by OHV-use areas.

4.2.3 Restoration of OHV-Impacted Areas

Revegetation of natural communities in arid environments is particularly difficult (Wallace and others, 1980) and studies evaluating revegetation have shown varying degrees of success (Graves and others, 1975, 1978; Kay and Graves, 1983; Grantz and others, 1998); thus, restoration of sites significantly degraded by OHV activity may be needed before sufficient levels of revegetation can take place, particularly if underlying bedrock has become exposed. Webb and others (1978) recommend that soil be imported and stabilized to replace the displaced soil where bedrock has become exposed. Generally, restoring soil horizons for re-establishing microbial communities can be achieved by inoculating soils with native microbes and mycorrhizae (Belnap, 1993; Bolling and Walker, 2000). Bolling and Walker (2000) suggest that decompacting OHV tracks and flattening out the lateral and center berms associated with them may increase the probability of community redevelopment with a more natural surface shape. Recovery of cryptobiotic soils, however, is more complex and may require long periods of time (Wilshire, 1983b; Lovich and Bainbridge, 1999); ultimately, their recovery rate will depend on the degree of soil compaction and the nature and intensity of the initial disturbance (Bolling and Walker, 2000).

To reduce the potential for erosion at restoration sites, it is important to use mulch, stabilization techniques, and/or establish vegetation. Rasor (1976, cited *in* Webb and others, 1978) suggests that ground cover (such as wire netting) be applied across the restoration site to stabilize soils. Kay and Graves (1983) recommend that seeding with local seed stock begin as the disturbance desists. Revegetation techniques also may include container planting (Grantz and others, 1998), hand seeding (Lovich and Bainbridge, 1999), drill seeding (Kay, 1988), and establishment of visually dominant species, such as creosote bush (*Larrea tridentata*; Kay and Graves, 1983).

5.0 Monitoring and Research Needs

5.1 Summary

More information is needed to help support policy making and land management as they pertain to the natural resources and people affected by OHV policies and management. Research needed to help support policy makers and land mangers includes (but is not limited to)

- well-designed studies that incorporate planned comparisons of treatment (OHV-impacted) and control (unimpacted/reference) sites, as well as “before and after OHV-impact” studies;
- studies at various spatial and temporal scales across all impacted habitat types;
- studies on habitat fragmentation and road-edge effects caused by OHV activities;
- studies on effects of various gradients in OHV disturbance and at varying distances from OHV routes;
- studies of OHV effects on plant and animal population dynamics;
- simultaneous evaluations of wildlife responses and OHV route-specific variables;
- studies to improve knowledge about the physical and chemical dynamics of soil compaction;
- studies evaluating the effects of erosion, sedimentation, and turbidity downslope of OHV-affected sites;
- improvements in techniques for successful site restoration;
- improvements in techniques and technologies for assessing OHV impacts over large areas and long time periods;

- studies that evaluate the effectiveness of various techniques to manage OHV use and its ecological and socioeconomic effects while simultaneously providing the greatest satisfaction among all land users; and
- studies that determine the economic and sociological costs of OHV use.

The experimental design of past studies on the ecological effects of OHVs often proved inadequate for providing reliable, defensible results. In particular, use of comparable treatment and control sites with adequate replication has been minimal. To better elucidate OHV effects on wildlife, habitats, and vegetation, there is a need for well-replicated research based on treatments and controls ranging across various spatial and temporal scales within the full range of habitat types represented on BLM lands. In desert ecosystems, for example, the impacts of OHV activities can occur at several spatial and temporal scales (Forman and others, 2003: p. 129-134; Matchett and others, 2004; Brooks and Lair, 2005; see discussion in section 2.1); thus, research conducted across the scale(s) at which OHV activities and ecosystem responses are likely to occur will produce the most reliable information about OHV effects on land health and users of the land (Brooks and Lair, 2005). There also is a need for long-term monitoring of OHV effects at both designated OHV sites and undesignated (rogue use) sites, as well as revegetation sites.

Monitoring and research approaches that take advantage, and push the advancement, of existing and emerging technologies are needed to fully represent the scale and diversity of OHV impacts likely affecting plant and animal populations and communities, and to ascertain indicator thresholds as they pertain to BLM's land health standards. Current technologies, including satellite imagery, GPS, and GIS, among others, would be extremely useful in broadening the scope of OHV-impact research from site-based effects to ecosystems and landscapes. Technology also provides opportunities for better assessing OHV effects on wildlife, vegetation, and other natural resources. Finally, multiple assessments and the simultaneous recording of independent and dependent variables would improve overall results and better inform management decisions.

5.2 Monitoring and Research Needs

Remaining questions about effects of OHVs on ecosystems and people are numerous and varied. Information regarding management approaches for sustaining or restoring resources—from the level of single OHV routes to entire landscapes—to pre-disturbance conditions while still providing for quality OHV experiences is especially sparse. Therefore, the need for solid, well-designed research for supporting management decisions cannot be understated. Based on major unresolved issues and questions raised in the OHV impacts literature, current research needs include (but are not limited to)

- well-designed research capable of producing scientifically sound results by incorporating planned comparisons of treatment (OHV-impacted) and control (unimpacted/reference) sites, as well as studies that take advantage of opportunities to compare "before and after OHV-impacts" at sites that may be slated for—but are not yet impacted by—OHV use;
- studies to evaluate OHV effects on natural resource attributes at various spatial and temporal scales, particularly those appropriate for evaluating and understanding effects occurring at watershed, landscape, and plant and animal population levels;
- studies to improve the overall understanding of habitat fragmentation and road-edge effects caused by OHV activities in OHV-impacted habitat types;

- studies that evaluate effects of OHV activities at various gradients in OHV disturbance levels and at varying distances from OHV routes;
- studies to evaluate how OHV activities and habitat fragmentation affect plant and animal population dynamics;
- simultaneous evaluations of wildlife responses (from individual- to population-level scales) and route-specific variables, including route type and width, and the intensity, noise levels, and speed of traffic;
- studies to improve the understanding of the physical and chemical dynamics, as well as the consequences, of soil compaction, sedimentation, and turbidity downslope of OHV-affected sites;
- improvements in techniques and technologies for assessing OHV impacts over large areas and long periods of time;
- improvements in techniques for successful site restoration;
- studies to determine the economic value of ecosystem goods and services provided by natural resources on or in BLM (and affected) lands and waters; and
- studies to determine the costs of OHV use, including degradation or loss of ecosystem goods and services, loss of supportive constituencies, managing/enforcing regulations of OHV use, and restoring OHV-impacted sites.

5.2.1 Scientifically Rigorous Research Projects

Several studies discussed in this document compared areas impacted (treatment) to areas unimpacted (control) by OHVs. Many studies, however, did not compare treatment and control sites, or the control and treatment sites differed with regard to some major factor (for example, other recreation activities, livestock grazing, logging, or energy-development activities) that could have masked true differences pertaining to OHV effects. In other words, controls in research and monitoring programs provide the necessary frame of reference for identifying true effects of the variable(s) of interest; in turn, identifying true effects will better inform management actions. Therefore, among the greatest research needs pertaining to OHV effects on natural resources are scientifically defensible studies based on planned comparisons of OHV-impacted and unimpacted sites that are otherwise similar in terms of soils, topographies, climatic patterns, plant and animal communities, non-motorized activities, and other potentially confounding factors. This is particularly important for developing appropriate threshold indicator values for sustaining current resource conditions or triggering management actions.

An additional approach to research that can provide informative results is to conduct before-and-after comparisons of sites previously unimpacted by OHV use but slated for future OHV use. Although year-to-year variations in climate and plant or animal population cycles can introduce too much variation in ecological before-and-after data to provide statistically meaningful results, this research approach can provide valuable information when conditions are reasonably similar during each phase of the project. Thus, it would be prudent to take advantage of such opportunities as they come up.

Technologies and tools that could prove very useful in study design and site selection pertaining to OHV effects include satellite imaging and GIS. Although they are still advancing rapidly in terms of their capabilities and utilities, they are nonetheless already very helpful in studies ranging from the site to the landscape scale. Satellite imagery can help locate sites with differing extents of OHV activity, and, when rendered into GIS database layers, they can provide opportunities for repeating imagery over time and evaluating landscape-scale changes. U.S.

Geological Survey scientists, for example, are currently using these tools and technologies to study the effects of energy-development activities in sage-steppe systems on BLM lands in Wyoming (C. Aldridge, pers. comm.).

5.2.2 OHV Effects at Various Spatial and Temporal Scales, Across Habitat Types

Past studies regarding OHV impacts on natural resources have focused primarily on effects at the single route or site level; thus, the overall understanding of landscape-, watershed-, and population-scale effects, including habitat/population fragmentation, is inadequate for managing OHV effects at larger spatial and longer temporal (Boyle and Samson, 1985) scales. Most of what is known pertains to very localized and short-term effects on sub-populations. Even then, past studies evaluating changes in animal densities of sub-populations may have violated basic assumptions of closed-population status. That is, studies that "failed to detect OHV effects on animal densities" may have been confounded by the immigration of new individuals after original individuals experienced poor survivorship due to direct or indirect effects of OHV activities. Mark-recapture studies that take advantage of radio-marking technologies, both in and away from the affected site, can help evaluate the extent to which assumptions of closed versus open populations are upheld or violated.

Research is also needed to better understand the edge effects of roads on different habitats and populations of different species or taxonomic groups. Here again, scale is important. Many past studies have evaluated the effects of traffic in immediately adjacent habitats (road rights-of-way, for example), whereas the effects may be realized well into the habitat interior quite far from travel routes. Thus, study designs that incorporate gradients of distance away from OHV routes or route networks would be crucial.

5.2.3 Research Regarding Effects of OHVs on Animal Populations

Many prior studies of OHV impacts on wildlife have focused on indirect indicators of animal population health, such as distribution and behavior, at the expense of more direct indicators, such as population trend/size, gender and age ratios, and productivity. In part, this is because such data can be significantly more difficult and expensive to collect than behavioral and distribution data. Studies that address possible changes in plant and animal genetics in ecosystems fragmented by OHV roads and trails also are needed. For species that naturally occur at low densities, such as the desert tortoise, bighorn sheep (*Ovis canadensis*), and mountain lion, this is particularly important, as isolation of their populations could more easily lead to localized extinctions.

Understanding direct effects of OHV activities on wildlife populations would be further enhanced by research that evaluates wildlife population dynamics in different habitat types (Bury and others, 1977), at different spatial and temporal scales, and under different conditions (levels) of OHV use. Likewise, research programs that incorporate representative and disparate taxonomic groups would help identify ecosystem-level effects. For example, a given habitat-fragmentation factor (a network of OHV routes) might not have any significant impact on large ungulates, but it might lead to complete loss of genetic diversity among flightless invertebrates. In conjunction with this type of research, it would be important to identify which specific environmental and anthropogenic factors promote or limit the exchange of individuals across OHV-impacted habitat types or landscapes.

Few past studies have employed multiple-assessment or simultaneous survey methods to detect rare or sensitive species that may not be detected through standard monitoring techniques, resulting in biases towards more common and easily detected species and an incomplete understanding of the community at risk. For example, multiple survey techniques can be used to determine whether changes in species composition are due to changes in detectability across habitat types, times of day or season, geographical area, and abundance. For detecting reptiles and amphibians, researchers could combine noosing, pitfall sampling, night surveys, and road driving surveys. For mammal studies, track, scat, and camera surveys might enhance species detections. Many avian-ecology researchers now use combinations of road driving, point sampling, and mist netting, in addition to double-observer methods, for improving overall survey results.

Studies of OHV effects on wildlife also could be improved through simultaneous recording of independent and dependent variables. For example, many studies relating effects of OHVs on animal populations also lack any measurements of static and dynamic road- and OHV-related variables—such as width and traffic intensity/noise levels (Andrews, 1990)—that may strongly influence species behavior, distribution, abundance, survivorship, and productivity. Given the importance of relationships between these independent and dependent variables, employing long-term monitoring that incorporates their simultaneous measurement would be very helpful (see Andrews, 1990). This type of information would be particularly useful for identifying indicator thresholds that might trigger area closures, re-routing of OHV routes, and/or implementing a restoration project. A variety of technologies and tools now available would be useful in these endeavors, including satellite telemetry, Global Positioning Systems (GPS), infrared photography, pneumatic vehicle counters, and so on.

Finally, there are a number of specific questions that researchers could ask with respect to OHV effects on animal populations. For example, the discussion in section 2.3.3 regarding edge effects of OHV routes raises the question of whether high densities of animals in roadside habitats represent favorable conditions for native fauna or dominance by invasive or non-native organisms and, if the former, whether these habitats are population sources or sinks (see Van Horne, 1983).

5.2.4 Research to Determine Socioeconomic Costs Associated with OHV Use

The literature search conducted for this report yielded no published studies of the economic costs associated with OHV use. Costs could include the degradation or loss of crucial ecosystem goods and services (such as a decline in water quality due to accelerated sedimentation and increased turbidity in wetlands caused by OHV traffic, or the loss of livestock and wildlife forage due to soil compaction caused by OHV traffic), the loss of economic and political support from both OHV users and non-motorized constituencies whose recreation experiences (or other land uses) are degraded by OHV effects, and the costs of managing OHV recreation and restoring sites impacted by OHV traffic. Although the costs of OHV effects may be challenging to assess, there are efforts underway to first identify the economic values of ecosystem goods and services and factor them into economic analyses of human activities. For example, Ducks Unlimited Canada and The Nature Conservancy have co-published a report that calls for the immediate acceleration of “efforts to measure, protect, and enhance the natural capital of Canada,” including the need to “invest in science to measure, value, and monitor ecological goods and services, and develop economic instruments that recognize and protect natural capital, rather than continue to reward its destruction” (Olewiler, 2004). A companion report details the values of goods and services provided by wetlands (Gabor and others, 2004). Similar efforts to place a value on the goods and services provided by lands affected by OHV activities would provide the necessary basis for balancing economic equations of OHV use.

5.2.5 Research to Improve Site Restoration

The success of site restoration ultimately depends on the ability to return soils, including abiotic crusts and desert pavement, vegetation, and biotic crusts to their original condition. Therefore, research is needed to study the mechanisms behind, and mitigation that could diminish, changes in soils. Adams and others (1982) called for research on the disproportionate hardening of only slightly compacted desert soil and whether soil hardening is caused by chemical cementation or by a greater number of interstitial water bonds remaining between soil particles after drying. Webb and others (1978) called for studies that explore the amount of time required for soils to respond to revegetation of OHV sites and develop ways of mitigating changes in soil properties during OHV use. More recently, Bolling and Walker (2000) expressed the need for long-term monitoring and analysis of soil microbial populations. Answers to these and other important questions that elucidate the ways in which systems recover from the effects of OHVs are crucial to the possibility of long-term OHV use without incurring irreparable damage to ecosystems.

6.0 Conclusion

It is apparent from the literature identified and discussed herein that the effects of OHV activities on ecosystems are diverse and potentially profound, if poorly understood. Studies have revealed a variety of effects on soil properties, watersheds, and vegetation resulting from one to multiple passes by OHV vehicles. Likewise, research has shown a variety of OHV effects on both OHV users and non-motorized recreators. Considerably less is known about impacts to wildlife or air and water quality. Whereas the results of past OHV-effects research have been reasonably consistent in demonstrating the nature of OHV effects in the immediate vicinity of single trails and OHV sites, there is a need for stronger emphasis on the cumulative effects—both spatial and temporal—of OHV use. For example, the effects of a single OHV route on a watershed may be greater when it is part of a route network than when it is the only route within the watershed. Furthermore, a network of OHV routes is likely to accommodate more users than a single route. Therefore, route density is an important consideration when evaluating and monitoring OHV effects across a given landscape.

The Council on Environmental Quality’s (CEQ) regulations for implementing the National Environmental Policy Act define a cumulative impact as “…the impact on the environment that results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions” (Council on Environmental Quality, 1997). In other words, cumulative impacts can result from what may appear on small spatial or temporal scales to be minor, but which become collectively significant when such actions take place over large areas and long periods of time. Moreover, one effect may interact with other effects to generate additional effects that are not apparent when evaluating effects individually.

The concept of cumulative impacts as they relate to OHV activity, therefore, must be applied in a landscape context, as these impacts are not site-specific and may affect adjacent or even more remote habitats and landscapes. For example, dust created from OHV activities can be dispersed to areas far away from habitats directly impacted by OHV activities. Likewise, erosion of soils during heavy rain events may increase sedimentation far downstream of areas directly subjected to OHV activities, and edge or corridor effects of OHV routes may promote widespread dispersal of non-native and invasive species. Thus, there is a need for greater monitoring and research emphasis on the effects of OHV activities not only in the areas directly subjected to those activities, but across impacted habitat types, watersheds, and landscapes. Overall, monitoring of cumulative impacts is needed at a scale larger than the physical imprint of the OHV-use area. By the same token, economic analyses of OHV use are needed to account for not only the immediate and apparent economic benefits, but also the long-term, large-scale, and ongoing costs associated with OHV use. Without factoring these variables into models of economic impacts, true cost:benefit ratios of OHV use will remain unknown.

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Appendix 1. Extensive Bibliographies

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1.6 Socioeconomic Implications of OHV Use

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Appendix 2. Search Methods and Results of Off-Highway Vehicle Effects Literature and Internet Resources

2.1 Methods

2.1.1 Literature Search

From May 10–26, 2006, a comprehensive literature search was conducted to encapsulate the current knowledge on effects of off-highway vehicle (OHV) activities as it pertains to natural resource attributes addressed by the Bureau of Land Management's (BLM) land health standards (U.S. Bureau of Land Management, 2001; Pellant and others, 2005). The search was conducted through 33 electronic literature databases available at Colorado State University's (CSU) library (table 2.1) and search engines available on the Internet. The databases searched encompassed all major, and some minor, sources of relevant literature, including professional, peer-reviewed journal papers and technical reports/articles published in magazines representing the industrial sector and non-governmental conservation organizations.

Search terms used included "OHV," "off-highway vehicle," "ORV," or "off-road vehicle" combined with terms representing the BLM's land health standards, including soil health and watershed condition, nutrient cycling, wildlife health and habitat quality for native plants and animals (especially for species of special status), water quality, and air quality (table 2.1). In addition, searches were conducted on the socioeconomics of OHV use. Within each database searched, all search terms were applied to the 26 topic areas (see footnote 1 associated with table 2.1) listed in CSU's library database subject list. Relevant citations also were gleaned from highly relevant reports and journal articles. All relevant citations identified were grouped by land-health categories to build extensive bibliographies for each land-health category. If a given citation was relevant to more than one land-health category, it was listed in each of the related bibliographies.

2.1.2 Internet Search

Specific goals of the Internet search were to (1) identify websites and other Internet resources provided by the BLM and U.S. Forest Service (FS) regions 2 (time constraints limited this search to Colorado), 3, and 4; (2) classify OHV-related Internet websites by focus/intent and resource type; and (3) report on highly relevant websites. Two primary products were subsequently developed: (1) a thesaurus (table 2.2) of search topics and terms related to OHV effects, and (2) a list of significant BLM and FS resources and corresponding Internet websites. The thesaurus was developed to identify Internet-based resources regarding OHVs and the types of OHV effects. All possible combinations of search terms were used, but any one search string depended on the search engine used. Although *FirstGov* is a good search engine for finding government publications and reports, it limits search results to 100. In contrast, *Google* returns up to 1000 results, and its filter may be used to constrain searches to a specific type of website (for example, ".gov"); however, *Google* is unable to employ multiple exclusion criteria to exclude irrelevant websites (for example, it cannot specify "NOT FS" and NOT BLM") whereas *FirstGov* does handle multiple exclusion criteria. In other words, *Google* was searched with an OHV term combined with an impact term while limiting any one search to "BLM," "EPA" (U.S. Environmental Protection Agency), "FS" (U.S. Forest Service), "NPS" (U.S. National Park Service), or "USGS" (U.S. Geological Survey) sites. Then *FirstGov* was searched with the same

Table 2.1. Search terms used and publication years included when using search engines and 33 electronic literature databases at Colorado State University's library to assemble an extensive bibliography of literature on effects of off-highway vehicles.

Database searched (for publications years)[1]	Search term										
	Air quality	**Benefits**	**Domestic livestock grazing**	**Erosion**	**Noise**	**Soil impacts**	**Travel and transportation management**	**Visual impacts**	**Vegetation impacts**	**Water quality**	**Wildlife impacts**
AGRICOLA EBSCO (1970-2006)	X		X	X		X			X	X	X
AGRICOLA National Agricultural Library (1970-2006)	X		X	X		X			X	X	X
Agricultural and Environmental Biotechnology Abstracts (1993-2006)	X			X	X	X	X		X	X	X
Agricultural Sciences (in Cambridge Scientific Abstracts) (compilation of multiple databases[2])	X		X	X		X			X	X	X
Animal Behavior Abstracts (1982-2006)		X	X		X						X
Applied Science and Technology Abstracts (1983-2006)	X			X	X	X	X	X		X	
ASFA 1: Biological Sciences and Living Resources (1971-2006)					X				X		X

Database searched (for publications years)[1]	Search term										
	Air quality	Benefits	Domestic livestock grazing	Erosion	Noise	Soil impacts	Travel and transportation management	Visual impacts	Vegetation impacts	Water quality	Wildlife impacts
ASFA 2: Ocean Technology, Policy & Non-Living Resources (1971-2006)	X			X		X	X	X			
ASFA 3: Aquatic Pollution and Environmental Quality (1990-2006)	X			X	X	X	X	X		X	
BioEngineering Abstracts (1993-2006)				X		X	X				
Biological Abstracts (1969-2006)			X						X		X
Biological Sciences (1982-2006)			X						X		X
Biological Sciences (in Cambridge Scientific Abstracts) (compilation of multiple databases[2])			X						X		X
Biology Digest (1989-2006)									X		X
Biotechnology and Bioengineering Abstracts (1982-2006)				X		X	X				
CAB Abstracts Archive (1900-1973)											X
Ecology Abstracts (1969-2006)		X	X						X	X	X

Database searched (for publications years)[1]	Search term: Air quality	Benefits	Domestic livestock grazing	Erosion	Noise	Soil impacts	Travel and transportation management	Visual impacts	Vegetation impacts	Water quality	Wildlife impacts
EIS: Digests of environmental impact statements (1985-2006)							X				
Engineering Index (1884-2006)				X		X	X				
Environmental Engineering Abstracts (1990-2006)				X		X	X				
Environmental Sciences & Pollution Management (compilation of multiple databases[2])	X			X	X	X	X				
Forest Service database (dates not indicated)			X	X		X	X		X	X	X
Geobase (1980-2006)				X		X					
Human Population & Natural Resource Management (1995-2006)			X				X				
Plant Management Network (compilation of multiple databases[2])									X		
Plant Science (1994-2006)									X		
Pollution Abstracts (1981-2006)	X				X		X				

	Search term										
Database searched (for publications years)[1]	Air quality	Benefits	Domestic livestock grazing	Erosion	Noise	Soil impacts	Travel and transportation management	Visual impacts	Vegetation impacts	Water quality	Wildlife impacts
Pollution Management & Environmental Sciences (in Cambridge Scientific Abstracts) (compilation of multiple databases[2])	X				X		X				
Threatened and Endangered Species System (dates not indicated)									X	X	X
Toxicology Abstracts (1981-2006)	X					X					
Water Resources Abstracts (1967-2006)										X	
Wildlife & Ecology Studies Worldwide (1935-2006)											X
Zoological Record (1978-2006)											X

[1]Topic areas searched in databases included biology, botany, civil engineering, construction management, earth resources, ecology, engineering, environment, environmental health, fishery, fisheries, forestry, forest science, geology, hydrology, life sciences, natural resource recreation, tourism, natural sciences, physical sciences, plant science, plant pathology, rangeland ecosystem science, soil and crop sciences, toxicology, water resources, weed science, wildlife, wildlife biology, zoology.

[2]Dates vary by database.

Table 2.2. Search topics and their associated search terms used in searching the Internet for publications and documents pertaining to off-highway vehicle effects and policies.

Search topic	Search term(s)
Off-highway vehicle	OHV
	Off-road vehicle
	Off-road vehicles
Air quality	Air quality
Benefits	Benefits
Domestic livestock grazing	Domestic livestock grazing
Erosion	Erosion
Human dimension	Human dimension
	Human dimensions
Noise	Noise
Soil impact	Soil impact
	Soil impacts
Transportation management	Transportation management
Travel	Travel
Vegetation impact	Vegetation impact
	Vegetation impacts
Visual impact	Visual impact
	Visual impacts
Water quality	Water quality
Wildlife impact	Wildlife impact
	Wildlife impacts

terms while excluding “BLM,” “EPA,” “FS,” “NPS,” “USGS,” and “.com” domains, which captured websites representing individual States, military agencies, and other non-commercial entities.

After the Internet searches were conducted, each website identified were reviewed (limited to 2 minutes per website) and assigned a relevance-class code (relevance to OHV effects and policies; table 2.3). Websites classified as H (highly relevant), R (relevant), or S (slightly relevant) were further categorized by focus area (table 2.4). All BLM and FS regions 2 (Colorado only), 3, and 4 websites were included in the results tallies; websites containing news releases, specific flora assessments, and BLM Resource Advisory Council meetings were omitted. Highly relevant websites were selected for presentation.

2.2 Results

2.2.1 Literature Resources

The literature search, and additional sources uncovered outside of the formal search, yielded approximately 700 citations, a number of which overlap in terms of their relevance to categories of land health (table 2.5).

Table 2.3. Relevance class codes and definitions pertaining to Internet websites found to contain information regarding off-highway vehicle effects and policies.

Relevance class	Definition
H	Highly relevant—high-quality resources related directly to OHV impacts; generally included number of OHV trail miles and visitor days, or reasons why OHVs excluded from specific locations; includes OHV strategy/plan documents or public communication sites on OHV recreation areas
R	Relevant—targeted mention of OHV but not as specific or detailed as a highly relevant site
S	Slightly relevant—mention of OHV but with few supporting statements or details
U	Unrelated—no or minimal OHV content
Z	Unable to access, page would not load, URL has changed, page loads but is empty of content
ZDup	Content duplicate of previous entry

Table 2.4. Focus areas and definitions of Internet websites found to contain information regarding off-highway vehicle effects and policies.

Focus area	Definition
Administration	Information sourced from the Federal level
Citizen input	Site contains or informs about public comments on OHV use
EIS	Site contains a draft or final environmental impact statement (EIS) or environmental assessment (EA)—attempt was to keep this category to sites related to actions up through completion of a management plan although there is some overlap with the next category due to title ambiguity
Impact	Site focus was measurement of impact
Legal	Response to appeals regarding travel management plans or EIS documents
Manual	Site containing a handbook or manual
Management plan	Sites with completed management plans or actions stemming from implementation of a management plan; includes monitoring, revision of plans, assessment for roadless areas, and road analyses
Monitoring	Usually annual reports of monitoring activity prescribed by a management plan
Press	Site with information in the form of a press release or media announcement
Road guide	Sites intended to provide public information about OHV and recreation site use; includes descriptions, trail maps, event calendars, safety, licensing and regulation, and closure information

Table 2.5. Number of relevant publications found, and publication dates included, in a literature search on effects of off-highway vehicle activity as they pertain to the U.S. Bureau of Land Management's land health standards.

Land health category	No. citations	Years included
Soils	314	1959-2006
Air quality	104	1970-2006
Water quality	218	1959-2005
Vegetation	326	1962-2006
Wildlife	387	1970-2006
Land users	211	1967-2006

2.2.2 Internet Resources

The Internet search yielded nearly 30,000 State and Federal government websites, of which 8,693 were unique (a single HTML page or .pdf file); 23 percent (*n* = 1,998) were BLM websites, 55 percent (*n* = 4,817) were FS websites, and 7 percent (*n* = 568) were NPS websites (table 2.6). FS regions 2 (Colorado only), 3, and 4 together represented 12 percent of the websites. Of the 8,693 unique sites, 2,495 were visited and reviewed (29 percent) using the methods described above. All search term combinations returned at least 100 results, and some returned as many as 3,700 (table 2.6).

Of the unique websites identified, only 13% (*n* = 335) were classified as highly relevant; 15 percent were relevant, 22 percent were slightly relevant, and 50 percent were unrelated/not available (figure 2.1). Forty-seven percent of those classified as highly relevant were BLM sites, and 53 percent were FS sites. The majority of highly relevant sites (68 percent) were dedicated to environmental impact statements (EIS) or management plans (figure 2.2). Only two percent of the highly relevant sites focused on measuring or monitoring the effects of OHV activities. When all sites were considered, a slightly higher percentage of sites (approximately 5 percent) included monitoring or impact assessment (figure 2.3).

Key similarities between highly relevant BLM and FS websites (figures 2.4 and 2.5) included equal emphasis on road guides (about 12 percent of sites) and little emphasis on monitoring and impacts (0 and 1 percent for BLM, respectively; 1 and 1 percent for FS, respectively). The BLM sites had slightly greater emphasis on EISs and management plans (73 percent) than did FS sites (65 percent). Interestingly, 10 percent of the FS sites concerned legal issues—primarily appeals to decisions—whereas no material of this type was found on BLM websites. This may be due to an agency decision of what type of material is posted to the Internet or it may be a result of a difference in EIS/management plan implementation status between the two agencies. Table 2.8 presents examples of five highly ranked web sites (if available) from BLM and/or FS Region 2 for each of the focus areas.

Table 2.6. Search results, by topic, for all Internet websites pertaining to off-highway vehicle effects and policies (n = 22,990).

Search topic	Results
Air quality	3,040
Benefits	3,081
Domestic livestock grazing	373
Erosion	3,228
Human dimension(s)	241
Noise	2,650
Soil impact(s)	461
Transportation management	453
Travel	3,726
Vegetation impact(s)	155
Visual impact(s)	1,392

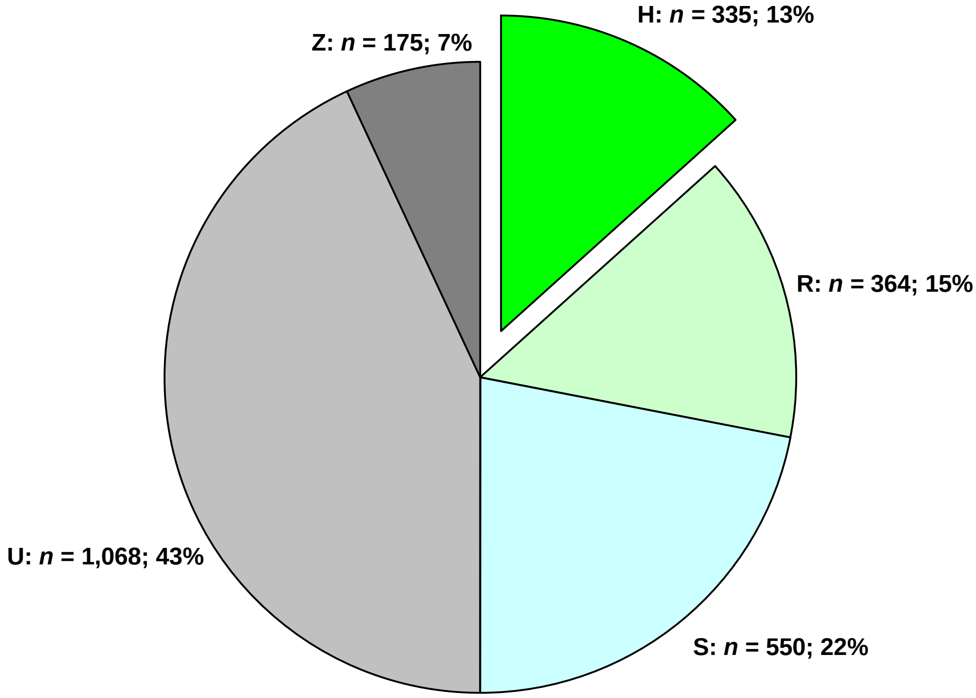

Figure 2.1. Breakdown of unique Internet websites (n = 2,495) classified as "highly relevant" (H), "relevant" (R), "slightly relevant" (S), and "unrelated/unavailable" (U, Z) to off-highway vehicle effects and policies.

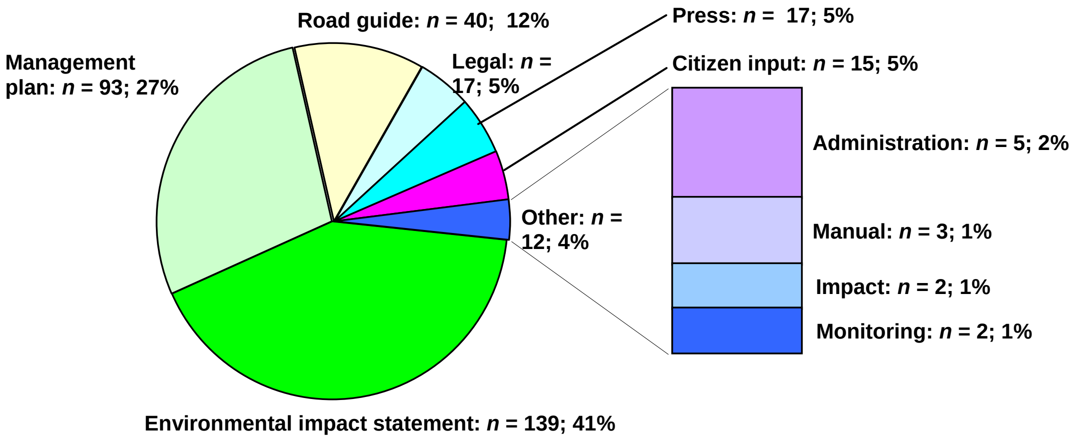

Figure 2.2. Focus areas of Internet websites (n = 333) classified as highly relevant to off-highway vehicle effects and policies.

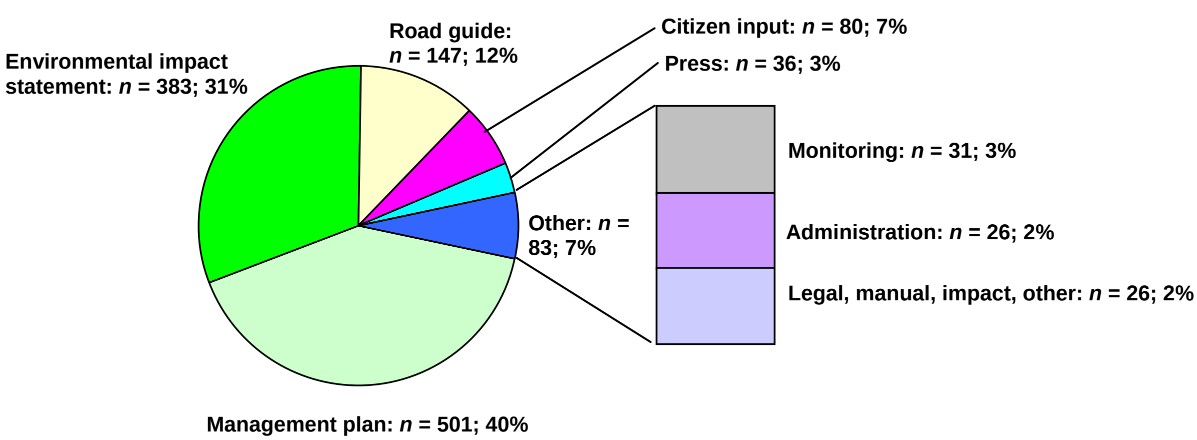

Figure 2.3. Focus areas for all Internet websites (n = 1,230) classified as highly relevant, relevant, and somewhat relevant to off-highway vehicle effects and policies.

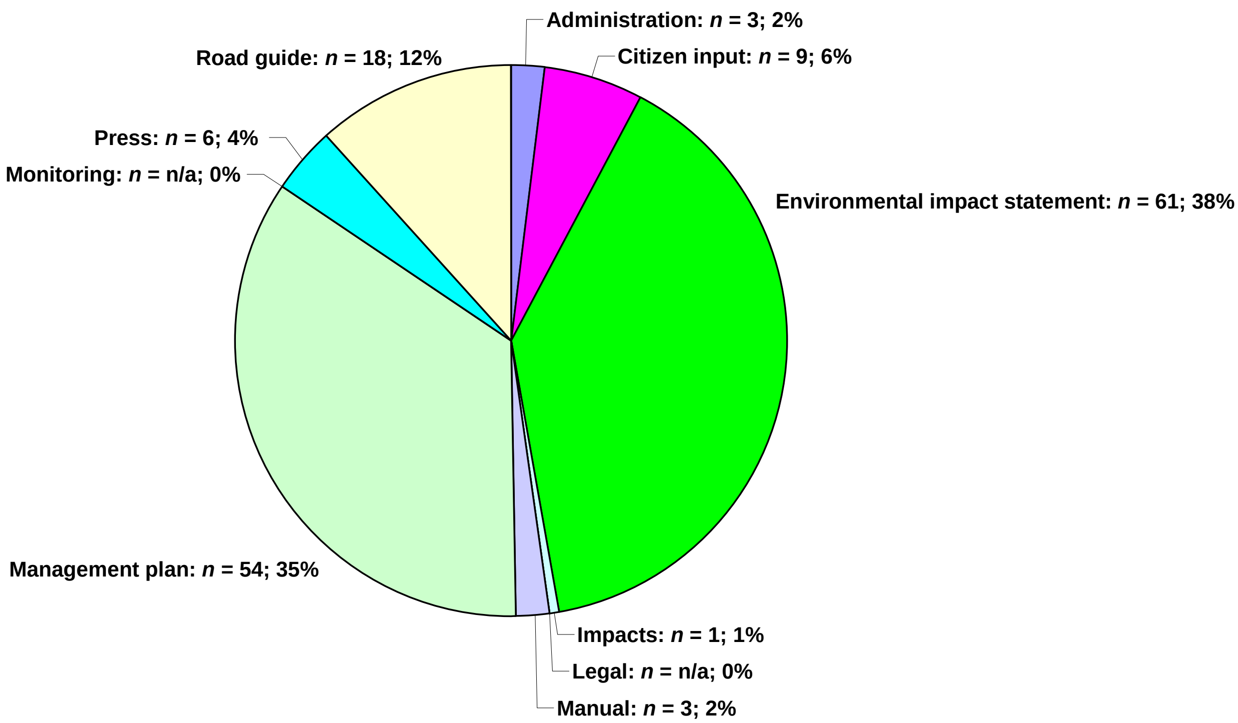

Figure 2.4. Focus areas of U.S. Bureau of Land Management Internet websites (n = 155) classified as highly relevant, relevant to off-highway vehicle effects and policies.

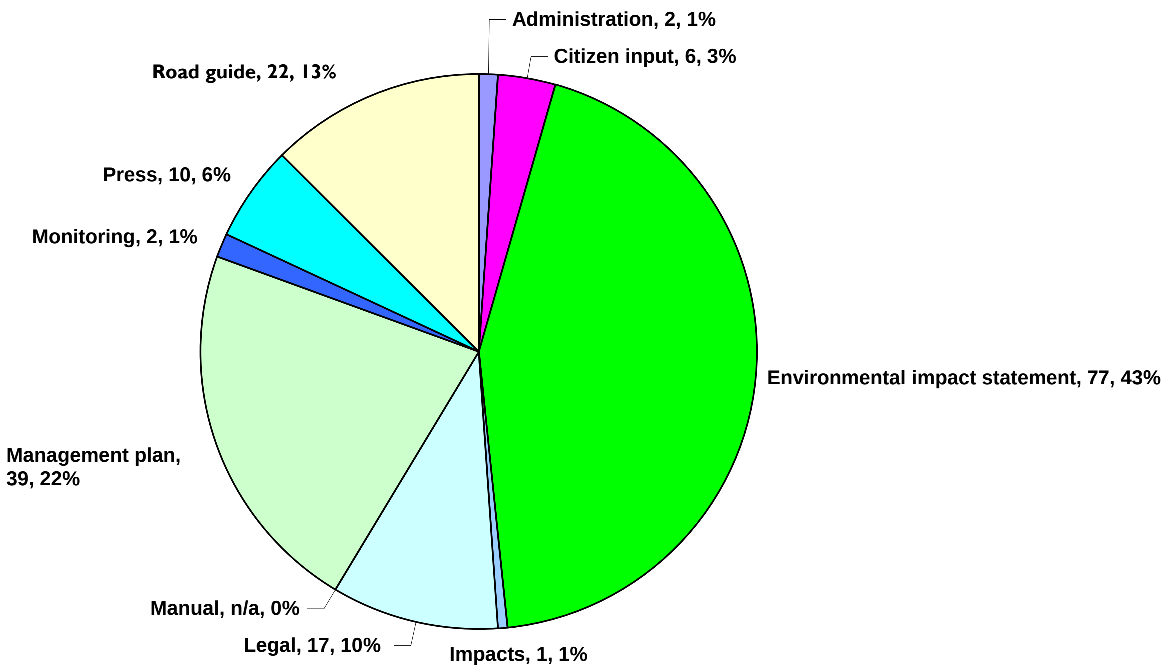

Figure 2.5. Focus areas of U.S. Forest Service (regions 2 [Colorado only], 3, and 4 only) Internet websites (n = 176) classified as highly relevant to off-highway vehicle effects and policies.

The large percentage of highly relevant websites assigned to the EIS and management plan categories indicate that OHV impact is an important topic to both the BLM and FS. The relatively small number of agency sites that address assessment or monitoring of OHV effects suggests that assessment or monitoring of OHV effects may be important topics for future website development. EISs and management plans do not focus on providing quantitative measures for determining the suitability of OHV trails (aside from the problem of trail redundancy for a given area); a trail-management plan of "fewer but better" seemed to be the approach in most plans, although a few plans specified that areas with a high degree of slope are unsuitable for trails. Almost no plans addressed OHV-related dust or noise problems, although some addressed areas of significant erosion by moving trails.

Overall, there appear to be more FS than BLM websites dealing with OHV management and related issues, including monitoring and the legal response to approved plans. Although road management plans are being developed for many land units by both the BLM and FS, only a few of these plans include OHV travel as an aspect of road management. The BLM, however, has produced a nation-wide OHV strategy, and Montana, Wyoming, and Idaho also have prepared state OHV management strategies. In general, policy appears to be taking the form of management plan implementation (such as closing areas due to muddy conditions or fire risk rather than making specific policy statements).

Table 2.7. Internet websites classified as highly relevant, by focus area and source, pertaining to off-highway vehicle effects and policies.

Focus/ source[1]	Title and URL
Administration	
BLM	BLM National Management Strategy for Motorized Off-Highway Vehicle Use *http://www.fs.fed.us/recreation/programs/ohv/blm_strategies.pdf*
FS-R2	Recreation, Wilderness, and Related Resource Management WO Amendment 2300-94-3 *http://www.fs.fed.us/cdt/admin.htm*
Environmental impact statement	
BLM	Environmental Assessment EA No.: AZ-020-04-0115 for the Arizona Association of Four-Wheel Drive Clubs 2004 4x4 Jamboree *http://www.blm.gov/ca/publish/etc/medialib/blm/az/pdfs/nepa/library/4wd.Par.7790.File.dat/ASA4WDC-Jamboree-EA.pdf*
BLM	Environmental Assessment**:** DARPA Grand Challenge *http://www.blm.gov/ca/pdfs/barstow_pdfs/darpa/chapter_3_affected_environment.pdf*
BLM	Final environmental impact statement for the Imperial Sand Dunes Recreation Area Management Plan and Proposed Amendment to the California Desert Conservation Plan 1980 *http://www.blm.gov/ca/pdfs/elcentro_pdfs/FinalEISandRAMP/FinalEIS.pdf*
BLM	Red Rock 4-Wheelers Jeep Safari and Fall Campout 5-Year Permit Renewal *http://www.blm.gov/ut/st/en/info/newsroom/2006/01/blm_renews_jeep_safari.html*
FS-R2	Decision Notice and Finding of No Significant Impact for the Clear/Crazy Designated Motorized Trail System, Powder River Ranger District, Bighorn National Forest *http://www.fs.fed.us/r2/bighorn/projects/projectfiles/clearcrazy/clear_crazy_fonsi_032005.pdf*
FS-R2	Scoping Document for the Hunt Mountain Travel Management Plan, Medicine Wheel/Paintrock Ranger District**,** Bighorn National Forest *http://www.fs.fed.us/r2/bighorn/projects/projectfiles/hunt_mtn_carea/scoping_carea.pdf*
FS-R3	Munds Park Roads and Trails Project: Environmental Assessment *http://www.fs.fed.us/r3/coconino/nepa/2004/munds-drft-ea-final-9_24_04.pdf*

Focus/ source[1]	Title and URL
Impact	
BLM	Air Quality Baseline and Analysis Report, Price Field Office, Price, Utah *http://www.blm.gov/utah/price/pricermp/documents/Baseline_and_Analysis_Report.pdf*
FS-R2	Anthropogenic Influences Used in Conducting Multiple Scale Aquatic, Riparian, and Wetland Ecological Assessments for the USDA Forest Service – Rocky Mountain Region, Report 2 *http://www.fs.fed.us/r2/projects/scp/arw/protocols/anthropogenicinfluencesusedinconductingmultiplescale.pdf*
Wildlands	Resource website that provides publications on PHV effects, restoration, enforcement, policy, and other related issues and materials *http://www.wildlandscpr.org/resources*
Legal	
FS-R2	Recommendation Memorandum for Uncompagre Travel Management Plan, July 13, 2000 *http://www.fs.fed.us/r2/projects/nepa/appeal-decisions/2000/gmug/uncompahgre_travel_39.htm*
FS-R2	Recommendation Memorandum for Gunnison Interim Travel Restrictions, July 13, 2001 *http://www.fs.fed.us/r2/projects/nepa/appeal-decisions/2001/gmug/gunntrvl_16.htm*
FS-R2	Recommendation Memorandum for Uncompahgre Travel Management Plan, June 18, 2002 *http://www.fs.fed.us/r2/projects/nepa/appeal-decisions/2002/gmug/tvl_mgmt_23.htm*
FS-R2	Recommendation Memorandum for Uncompahgre Travel Management Plan, June 19, 2002 *http://www.fs.fed.us/r2/projects/nepa/appeal-decisions/2002/gmug/tvl_mgmt_24.htm*
FS-R2	Recommendation Memorandum for Radial Mountain Travel Management Environmental Assessment, Sept. 13, 2001 *http://www.fs.fed.us/r2/projects/nepa/appeal-decisions/2001/mbr/radial_mtn_25.htm*
Manual	
BLM	Western Oregon Plan Revisions: Proposed Planning Criteria and State Director Guidance *http://www.blm.gov/or/plans/wopr/files/PlanningCriteriaDocument.pdf*
BLM	Interpreting Indicators of Rangeland Health *http://www.blm.gov/nstc/library/pdf/1734-6rev05.pdf*
BLM	Biological Soil Crusts: Ecology and Management *http://www.blm.gov/nstc/library/pdf/CrustManual.pdf*
Management plan	
FS-R2	U.S. Forest Service, Travel Management: New Rule *http://www.fs.fed.us/r2/recreation/travel_mgmt/*

Focus/ source[1]	Title and URL
FS-R2	Decision Notice & Finding of No Significant Impact, Grand Mesa Travel Management, December 1, 2003, Delta And Mesa Counties, Colorado *http://www.fs.fed.us/r2/gmug/policy/gm_travel/fonsi_dec2003.pdf*
FS-R2	Roads Analysis Report: Medicine Bow National Forest *http://www.fs.fed.us/r2/mbr/projects/roads/adobepdf/medbow/mbnf_rds_analysis_final.pdf*
FS-R2	Roads Analysis Report: Routt National Forest *http://www.fs.fed.us/r2/mbr/projects/roads/adobepdf/routt/routt_rap_final.pdf*
FS-R2	Travel Management Rule, Implementation Safety, Rocky Mountain Region *http://www.fs.fed.us/r2/recreation/travel_mgmt/references/Final_TravelMgmtStrategy.pdf*
Monitoring	
FS-R2	Forest Plan Monitoring and Evaluation Reports, Arapaho and Roosevelt National Forests and Pawnee National Grassland *http://www.fs.fed.us/r2/arnf/projects/forest-planning/monitoring/*
OHV road guide	
BLM	Killpecker Sand Dunes Open Play Area, BLM Wyoming Rock Springs Field Office, Wyoming *http://www.wy.blm.gov/rsfo/rec/dunes.htm*
BLM	Dunes OHV Area, Farmington Field Office, New Mexico *http://www.nm.blm.gov/recreation/farmington/dunes_ohv_area.htm*
BLM	Lark Canyon, El Centro Field Office, California *http://www.ca.blm.gov/elcentro/larkcany.html*
FS-R2	OHV, Pike & San Isabel National Forests, Cimarron & Comanche National Grasslands, South Park Ranger District, Colorado *http://www.fs.fed.us/r2/psicc/sopa/roads.shtml*
FS-R2	Rampart Range Motorized Recreation Area, Pike & San Isabel National Forests, Cimarron & Comanche National Grasslands, South Platte Ranger District, Colorado *http://www.fs.fed.us/r2/psicc/spl/spl_ohv.shtml*
FS-R2	OHV, Arapaho & Roosevelt National Forests, Sulphur Ranger District, Colorado *http://www.fs.fed.us/r2/arnf/recreation/ohv/srd/stillwaterpass-grandlake.shtml*

[1] BLM = U.S. Bureau of Land Management; FS = U.S. Forest Service; R = region (Region 2 limited to Colorado)

Equivalent Roaded Area as a Measure of Cumulative Effect of Logging

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ABSTRACT / A watershed disturbance index developed by the USDA Forest Service called equivalent roaded area (ERA) was used to assess the cumulative effect from forest management in California's Sierra Nevada and Klamath mountain ranges. The basins' ERA index increased as logging and road-building occurred and then decreased over time as management ceased and vegetation recovered. A refinement of the standard index emphasized disturbances in sensitive, near-channel areas, and evaluated recovery periods of 20, 30, and 50 years. Shorter recovery periods yielded better correlations between recovering forest systems and aquatic response than the longer recovery period, as represented by ERA and diversity or dominance, respectively. The refined ERA index correlated more closely with macroinvertebrate dominance and diversity information that was available for part of the study period. A minimum ERA threshold of 5% was detected, below which no effect to the macroinvertebrate community was observed. Above this threshold, elevated ERA values were associated with a decline in macroinvertebrate diversity and an increase in dominance of the top five taxa. Use of an ERA technique that emphasizes near-channel areas and biological thresholds would contribute to the Forest Service's implementation of ecosystem management.

In forested watersheds, excessive management may lead to disturbance adjacent to the stream channel and may raise water temperature, decrease slope stability, and increase the potential for erosion. As a result, prediction of land-use impacts on fisheries is important because fisheries are a major beneficial use of many stream systems in the western United States (Geppert and others 1984).

The need to evaluate and predict the effects of forest management activities has increased as competition for use of the limited resource base has increased. Most cumulative effects techniques evaluate disturbed area, potential for sediment yield, water quality, or changes in probable peak flow (Harr 1982, Dickert and Olshansky 1986, Haskins 1987, Weaver and others 1987, Johnston and others 1988). Although these methods are excellent approaches for assessing the physical and chemical impacts of cumulative land-use activities in watersheds, less work has been done to implement biotic assessments of management practices. Salo and Cederholm (1981) examined the cumulative effects of two different anthropogenic stresses superimposed on the natural mortality of a hypothetical coho population that might result in extinction. Furthermore, Geppert and others (1984) identified several aquatic habitat modifications caused by forest practices that could change rates of fish growth, survival, abundance, and species composition.

Aquatic insects are important indicators of aquatic ecosystem health (Rosenberg and others 1986). Because invertebrates are also a primary food source for fish, insect abundance is generally linked to productive fisheries (Plafkin and others 1989). Although fish and invertebrate populations vary dramatically due to climate, fires, and other factors unrelated to management, benthic macroinvertebrate communities better reflect management impacts on the aquatic system than most other biological measures considered for routine monitoring (Plafkin and others 1989). Their sedentary life-styles and relatively fast rate of reproduction make them useful indicators of local environmental changes. The effect of disturbances on macroinvertebrates is dependent on the magnitude and scale of the disturbance, and disturbance scales range from areas less than a few hectares (logging, mass wasting) up to whole basins (wildfire) (Richards and Minshall 1992). Macroinvertebrate indices such as

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abundance, dominance, and diversity vary from minor, short-term changes in response to small-basin logging, to decade-long shifts and extinctions because of wildfire or chemical spills. Diversity has been shown to decline after logging that caused disturbance in the near-stream area, and then approach predisturbance levels over times ranging from 7 to 11 years (Newbold and others 1980; Erman and Mahoney 1983; Roby and Azuma 1995).

Biotic assessments are useful indicators of current conditions, but unlike land-disturbance methods, they cannot be done in retrospect. Disturbance measures can be based on historical land-use changes, but the significance and veracity of the results are difficult to assess in most cases.

This paper reports on a study that uses the results of a long-term biotic assessment in California to evaluate a land disturbance methodology and suggest improvements to that methodology.

Equivalent Roaded Area: A Cumulative Effects Methodology

The USDA Forest Service (FS) has played an active role in developing and implementing a cumulative effects assessment methodology for public forest lands in California. Public concern, the National Environmental Policy Act of 1969, and the National Forest Management Act of 1976 demand that FS land managers assess current conditions and predict the effects of management alternatives on the terrestrial and aquatic ecosystems. Additionally, implementation of Section 208 of the Clean Water Act (as amended) caused the FS in California, in conjunction with other agencies, to develop best management practices for water quality management planning (USDA Forest Service 1988). Numerous lawsuits have alleged the FS's failure to consider various aspects of cumulative watershed effects (CWE), underscoring the need to fully incorporate an analysis of CWE in all resource planning.

Although controversy continues regarding the nature and definition of CWE, most authors agree that a basin's harvest history should be documented, and that effects are cumulative if they occur within a certain time period (Klock 1985). In addition to logging, forest fires, grazing, mining, recreation, and other activities affect watersheds and associated streams. The combined effects of the full range of land-use activities should be considered when any individual activity's effects are evaluated.

The Pacific Southwest Region of the FS developed a generalized framework to examine CWE in a small- or moderate-sized basin by: (1) identifying the beneficial uses of the stream and the water, (2) examining the factors influencing the beneficial uses, and (3) assessing the effects of multiple management actions on beneficial uses (Seidelman 1981; USDA Forest Service 1988). This disturbance index method was designed to satisfy the National Environmental Policy Act of 1969 through its consideration of past, present, and anticipated management activities. A land-use history is developed that details the date, area, type of logging and yarding, and miles of various types of roads for both public and private lands in the basin. The method standardizes past and planned disturbance activities (clear-cuts, selective cuts, prescribed burns, wildfires, etc.) in terms of equivalent roaded area or equivalent road acres (ERA) through the use of disturbance coefficients (Haskins 1987). For example, the coefficient for a light selective cut with tractor yarding counts half as much per acre as a clear-cut with tractor yarding. Clear-cuts that have had several decades to recover are discounted compared to recent cuts, and the disturbance areas in the basin are summed and divided by the total basin area to yield the basin ERA. By converting all typical activities to an ERA index, disturbances throughout the basin are incorporated. Although the ERA methodology was developed in California, the technique could be applied to other regions or disturbance regimes by modifying the coefficients to reflect local practices.

Basin disturbance indices such as the ERA method can be used to achieve dispersion of practices over space and time. Based on indices, logging would be spread across the forest rather than concentrated within a basin, or a clear-cut might be delayed until recently cut areas recover. Thresholds can be selected that correspond with the onset of detectable change, with the onset of undesirable change, or with the onset of damage levels thought to be irreversible. In watershed management, for example, 15–20% of the timber in a watershed must be cut within a decade before a measurable change in runoff volume will occur (Bosch and Hewlett 1982). Conversely, extreme disruption from logging, fire, or other disturbances may reduce the species diversity and carrying capacity of the aquatic ecosystem (Geppert and others 1984). To avoid undesirable change, a threshold of concern is selected based on an historical analysis of similar watersheds. If the ERA exceeds the threshold, the district ranger may call for on-site investigations and may then either delay the planned harvest or specify extra mitigation to avoid creating off-site effects.

The current ERA methodology is far from perfect. To reflect watershed disturbance accurately, disturbance coefficients need to be determined for land management activities such as grazing, mining, and recreational developments. Distinct thresholds of concern also are needed for diverse geographic regions and beneficial uses. Validation is difficult because of the multiresource, long-term data that are required. Climatic variability, particularly in the arid West, may mask the linkage between upland activities and stream ecosystem response. For example, periods of drought (such as occurred in California between 1987 and 1992) delay the transport of sediment through the system. Conversely, infrequent high flows from heavy rains on the snowpack can cause major changes in the fluvial and biotic systems that may or may not be affected by past management. The ERA methodology has been implemented by the FS in California, but further evaluation is needed.

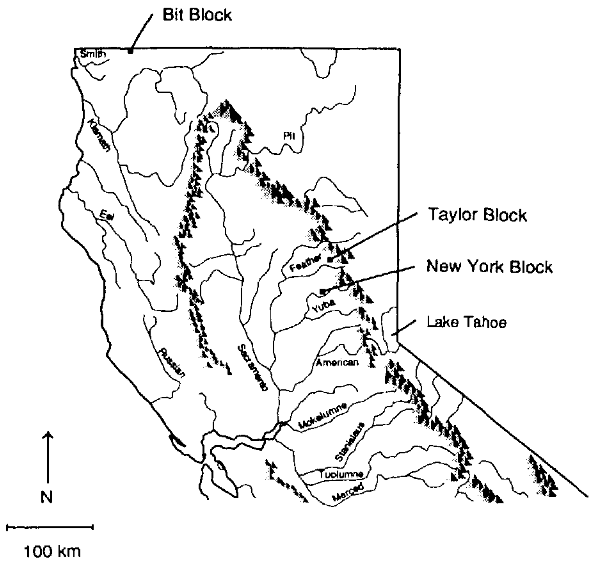

Figure 1. Study site locations in northern California, USA.

Research History and Site Description

The macroinvertebrate and recovery results in this study are derived from long-term work directed by Erman (Erman and others 1977, Erman and Mahoney 1983, Fong 1991). He and students from the University of California at Berkeley (UCB) showed that macroinvertebrate communities in several geographically distinct, northern California streams responded to localized logging-related disturbances (Erman and others 1977; Newbold and others 1980). Later studies directed by Erman followed the recovery of these communities after the initial disturbance (Erman and Mahoney 1983; O'Connor 1986; Fong 1991). These studies partitioned watersheds into blocks, and a subset of three blocks from the UCB studies was used in our analysis of the ERA index (Figure 1). The New York, Taylor, and Bit blocks have treatment watersheds with different amounts of logging-related disturbances and control drainages with minimal logging activities and roads. The treatment watersheds experienced timber removal within the streamside riparian zone during the early 1970s. Most controls had no logging activities next to the stream channel. If some harvest did occur, it was not close to the macroinvertebrate sampling sites.

Located primarily on national forest lands, the UCB study sites were first- and second-order streams draining midelevation watersheds ranging from 61 to 828 ha (Table 1). The New York block drains into the North Yuba River, the Taylor block drains into the North Fork Feather River, and the Bit block drains into the Klamath River. The predominant vegetation in the watersheds included Douglas fir (*Pseudotsuga menziesii*), true fir (*Abies* spp.), and mixed conifers (*Calocedrus decurrens*, *Pinus* spp., and *Abies* spp.). Hardwood trees and deciduous shrubs (*Acer* spp., *Alnus* spp., *Quercus* spp., and *Salix* spp.) occur frequently within the riparian corridors.

Methods

Equivalent Roaded Area

The methodology for calculating ERA has been described in detail by Seidelman (1981), Haskins (1987), and the USDA Forest Service (1988). Aerial photography has been the primary source of historical disturbance and condition information in cumulative impact studies (Dickert and Tuttle 1985; Grant 1988; Johnston and others 1988). Since the 1940s, most national forests have aerial photography of their lands at 5- to 15-year intervals. This photographic record remains the best (and often only) source of information over time concerning land use, soil disturbance, mass wasting, and vegetation cover (Dickert and Tuttle 1985).

Photographic coverage of the Bit and Taylor blocks was acquired for the period from the mid-1940s to the mid-1980s. Coverage of the New York block was unavailable before the mid-1960s. Photo scale ranged from 1:12,000 to 1:24,000. The earliest photos were used to approximate the ages of roads and timber harvests. The last 20 years of the photo record were combined with available timber sales records from FS district offices to develop a sequential land-use history. A zoom transfer scope was used to

Table 1. Range of characteristics for stream reaches in Taylor, New York, and Bit blocks of northern California[a]

Blocks	Subwatershed area (ha)	Reach elevation (m)	Mean annual precip. (cm)[b]	Geologic type[c]
Taylor	113-828	1700–1930	100	gr
New York	92-531	1030–1330	159	lp, gr
Bit	61-218	1145–1380	141	m, g

[a]Sources: Erman and others (1977), Erman and Mahoney (1983), and Fong (1991).
[b]NOAA (1989).
[c]lp = Paleozoic marine, m = pre-Cenozoic metamorphic, g = granitics, gr = Mesozoic granitics.

correct for geographic displacement and tilt as the photo information was transferred onto Mylar at a common scale. The information was entered into a geographic information system to analyze the temporal and spatial trends. A map of the Taylor block is included as an example (Figure 2).

Land-use information was transformed into percent ERA values following a procedure developed by Haskins (1987). The terms used in the procedure are defined as:

- Harvest unit ERA (area) = recovery factor × disturbance factor × area
- Road ERA (area) = road length × average width
- Total ERA (area) = sum of harvest unit and road ERAs
- Percent ERA = (total ERA/watershed area) × 100

Harvest units. Each harvest unit was classified according to harvest and timber removal methods. Harvest methods included clear-cut, light selection, and heavy selection. Removal methods included tractor, cable, and skyline techniques. The severity of selective harvests was classified by using an estimate of percent of the area covered by skid trails (5%–40%) and an estimate of percentage of trees removed. The area of each harvest unit was determined by the geographic information system or a planimeter.

Recovery factor. Barring additional disturbances, bare lands revegetate with grasses, shrubs, herbs, and trees. Rooting by herbaceous and woody plants stabilizes streambanks, retards erosion, and shapes bank morphology (Swanson and others 1982). As revegetation progresses, flow peaks and volumes have been shown to return to preharvest conditions in about 30 years (Swanson and Hillman 1977). Except for roads, ERAs for disturbed areas are therefore discounted based on an assumed 30-year linear recovery rate (USDA Forest Service 1988). In addition, 20- and 50-

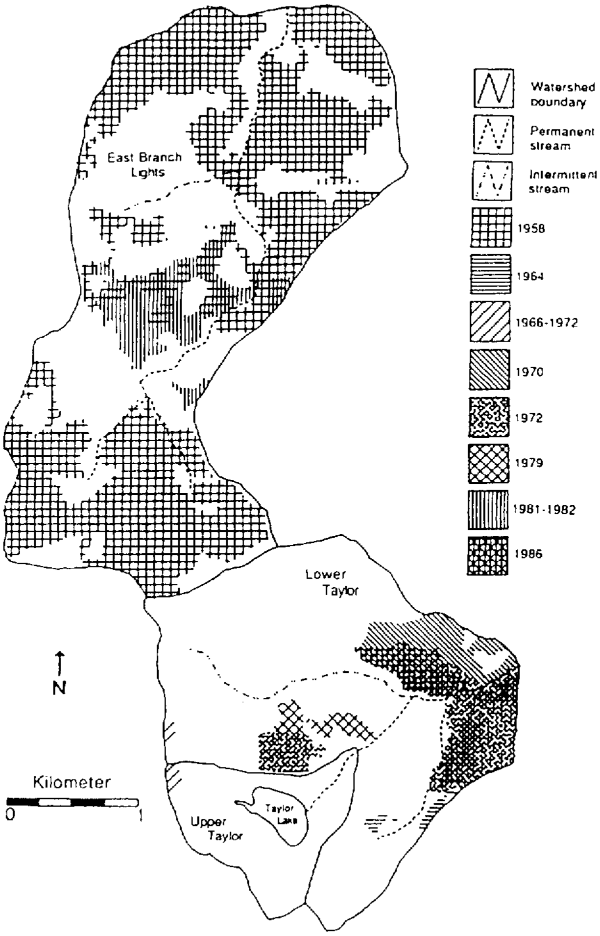

Figure 2. The extent and date of historic logging activities since the 1940s in watersheds of the Taylor block.

year recovery periods were tested. The recovery factor for any time after a harvest is:

Recovery factor = (analysis date − harvest date)/recovery period
or = 1 if harvest is over 30 years old

Table 2. Disturbance coefficients used to relate timber harvest practices to effects of road building[a]

Management activity	Disturbance coefficient
Road, cut, and fill	1.00
Tractor clearcut	0.2–0.3
Tractor light selective	0.1–0.15
Tractor heavy selective	0.2–0.3
Cable clearcut	0.15–0.2
Cable light selective	0.1

[a]Source: Haskins (1987), and USDA Forest Service (1988).

For example, a clear-cut that occurred this year has a recovery factor of zero, and the entire area is added to the basin's disturbance tally. If the clear-cut occurred 20 years ago and the recovery period is 30 years, then the cut is 67% recovered and only 33% of the original area is added to the tally. Although this procedure may be debatable in terms of movement of soil particles or nutrient releases, it is reasonable in terms of hydrologic factors such as peak flow and annual runoff volume.

In the case of the pre-1960 harvests, exact harvest dates were often unknown, so harvest dates were estimated as the midpoint year between two photographic dates. If the earliest photo showed a harvest, that photographic date became the year of harvest regardless of when it may have occurred.

We adopted a conservative approach by assuming roads would not recover. It is recognized that although roads produce sediment both immediately after they are constructed (McCashion and Rice 1983) and again after five to seven years when tree roots rot (Ziemer 1981), unused roads may not be major contributors of sediment a decade after construction. However, compacted road surfaces increase runoff and increase the risk of mass wasting. Culvert washouts also may occur at any time if they are undersized, unmaintained, or if large rain-on-snow storms occur.

Disturbance factor. The FS has established a range of disturbance factors or coefficients for various harvesting methods (USDA Forest Service 1988). The magnitude of these coefficients reflects the severity of land disturbance resulting from different practices (Table 2). Hence, tractor clear-cutting is assigned the highest coefficient after roads.

Roads. A road was defined as any constructed passage that can be used by a logging truck or passenger vehicle. This definition excluded skid trails and paths, but various sizes of roads were encountered. A width of 12 m was the dividing point in the classification of roads into narrow and wide classes. Cut and fill slopes visible on aerial photographs and in on-site inspections were included in the measurement of road width.

ERA modifications. The standard ERA method used by the FS in northern California does not include the location of roads and harvest units relative to streams and other sensitive landforms. Although the method recognizes the inherent sensitivity of active and dormant landslides, valley inner gorges, riparian areas, land slopes greater than 80%, and very highly erodible soils, these factors are not included in the determination of disturbance coefficients (USDA Forest Service 1988). In recognition of the role that sensitive lands play in CWE (Geppert and others 1984), we modified the standard FS method to focus on a 100-m streamside impact zone (SIZ) on each side of the channel. The SIZ typically includes many of the sensitive features listed above. The stream network was based on information from US Geologic Survey 7.5° topographic maps, supplemented by field observations. In essence, this modification makes the SIZ area the watershed area. Roads and harvests within each SIZ were estimated as per the FS methodology to produce an ERA for each SIZ. As with the standard ERA analysis, we computed ERA values for recovery periods of 20, 30, and 50 years.

Macroinvertebrate Analysis

Using Spearman rank correlation, diversity and dominance data for the three or four sample periods and all treated watersheds in Erman's 20-year UCB studies were compared to the time-series ERA indices for matching years and watersheds. The intent of the correlation was to determine if the pattern of the ERAs corresponded with the pattern shown by the macroinvertebrate communities over time and across all treated catchments. Erman's studies noted that Shannon diversity and taxa dominance were reflective of habitat changes that were related to management effects (Erman and others 1977; Erman and Mahoney 1983). Shannon diversity is defined as:

$$H = -\Sigma\,[p_i \times (\ln p_i)]$$

where p_i is the proportional abundance of the *i*th taxa (n_i/N), n_i is the individual abundance, and N is the total abundance (Magurran 1988). Relative dominance was estimated by two techniques: (1) the most abundant taxa; and (2) the five most abundant taxa per stream reach.

Table 3. Roaded area for watershed or streamside impact zone (SIZ), photograph date, watershed areas (entire basin and SIZ), control (C), and treatment (T) for Taylor, New York, and Bit blocks

Photo date	Roaded area in basin	SIZ area	Roaded area in basin	SIZ area	Roaded area in basin	SIZ area	Roaded area in basin	SIZ area	Roaded area in basin	SIZ
	Lower Taylor (T) (601 ha/116 ha)		Upper Taylor (C) (113 ha/11 ha)		E. Branch Lights (C) (828 ha/132 ha)					
1941	3.0	0.5	1.2	0.3	0.8	0.0				
1953	3.0	0.5	1.2	0.3	2.0	0.2				
1966	3.0	0.5	1.2	0.3	25.4	5.6				
1972	10.4	3.2	1.2	0.3	25.6	5.3				
1977	10.4	3.2	1.2	0.3	25.1	5.3				
1982	10.4	3.2	1.2	0.3	24.5	6.3				
1987	10.4	3.2	1.2	0.3	23.6	6.3				
	Mid. New York (T) (466 ha/124 ha)		New York Trib. (T) (92 ha/24 ha)		Upper New York (C) (347 ha/89 ha)		Empire (C) (532 ha/127 ha)			
1972	5.6	0.3	0.2	0.0	5.4	0.3	3.0	0.6		
1977	7.9	0.7	0.5	0.2	6.7	0.1	3.0	0.6		
1982	8.0	0.7	0.8	0.8	6.5	0.2	3.0	0.6		
1987	11.1	1.5	2.5	0.5	7.9	0.3	3.0	0.6		
	Lower Four Bit (T) (101 ha/21 ha)		Lower Two Bit (T) (120 ha/38 ha)		Upper Four Bit (C) (86 ha/15 ha)		Upper Two Bit (C) (61 ha/18 ha)		E. Fork Indian (C) (87 ha/15 ha)	
1944	0.8	0.0	1.7	0.5	0.8	0.0	1.2	0.0	1.1	0.0
1955	0.5	0.0	1.7	0.5	0.5	0.0	1.2	0.0	1.1	0.0
1964	3.7	1.0	3.5	1.3	2.9	0.3	1.2	0.0	1.7	0.0
1971	5.8	1.0	4.0	1.3	3.3	0.3	1.5	0.0	2.0	0.0
1975	5.8	1.0	4.0	1.3	3.3	0.3	1.5	0.0	2.0	0.0
1980	6.3	1.0	4.0	1.3	3.9	0.3	1.5	0.0	4.9	0.0
1986	6.3	1.0	4.0	1.3	3.9	0.3	1.5	0.0	4.9	0.0

Results and Discussion

Equivalent Road Area Trends

All study watersheds had either roads or logging activities throughout the length of the photographic period (Table 3). Some watersheds, such as Upper Taylor, had constant, nonzero ERA values that persisted throughout the photo record, which indicated the presence of roads but not harvests (e.g., Upper Taylor in Figure 3). In the mid-1950s, ERA values rose from near zero on the Taylor and Bit basins, reflecting significant logging activity in these areas (Figures 3 and 4). Major increases in harvesting and road building occurred in the 1960s in some control watersheds (Tables 3–6). In the mid-1950s and 1960s, ERA values for the treatment watersheds reached 10% on the Taylor, Bit, and New York blocks, and percentages of disturbed area reached 20–30%. Percentages of disturbed area in the New York block were very high and reached 86% (Table 5). However, these percentages included areas that were harvested more than once. Some control watersheds experienced heavy disturbances prior to the 1960s, but we concluded they were generally distant enough from

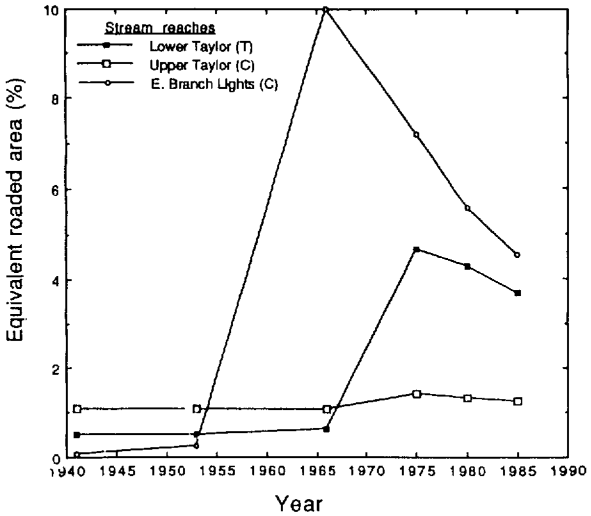

Figure 3. Trend in equivalent roaded area index for the Taylor block using a 30-year recovery period.

the channel to not invalidate the classification as a control. Figures are not shown for the New York block because there were no photographs prior to

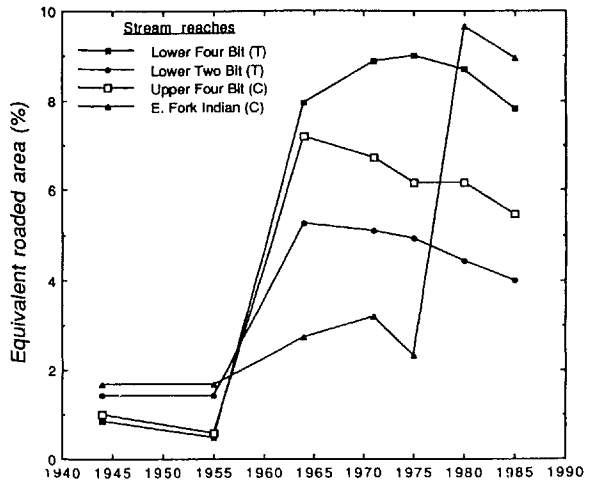

Figure 4. Trend in equivalent roaded area index for the Bit block using a 30-year recovery period.

1966. However, the New York watersheds had extensive cutting and mining prior to 1966. A similar pattern of disturbance occurred in the SIZs as occurred in the entire watersheds (Figures 5 and 6). Maximum ERA levels exceeded 8% for the Taylor and Bit blocks, and 10% for the New York block (no figure).

For the plots of ERA values, only the 30-year recovery curves are shown because the general pattern of declining ERA after harvest was similar when the 20- and 50-year recovery rates were used. ERA values dropped more quickly when the 20-year recovery rate was used, unlike the slow declines that resulted from using the 50-year rate. With the 50-year recovery rates, ERA values peaked slightly higher in a few cases when new harvests occurred before past harvest areas had recovered completely.

Near-Stream Activities

The modified ERA method emphasized activities within 100 m of the stream. For our sample of watersheds and using a Spearman correlation test, this refinement yielded significant correlation between some of the ERA values and community parameter indices derived from the UCB studies (Table 7). The nonparametric Spearman correlation was used to avoid making distributional assumptions about the data. The sample correlations between the modified ERA values and Shannon diversity were higher than they were with the standard ERA method. Comparison of ERA values and community dominance yielded similar results. Proximity of the disturbance to the channel increases the chance that the aquatic ecosystem will be affected. By limiting the effective watershed to a 200-m strip along the length of the channel, our refined method increased the proportional effect of any disturbances in that area. Alternately, an ERA method could be formulated that included the entire watershed but had different, much higher disturbance factors for areas within the SIZ. This revision would incorporate the management in the upland areas of the basin but reflect the sensitive nature of the near-channel zone.

Recovery Period

Following the cessation of logging activities, the physical and biological environment recovers and ERA values decline based on the assumed recovery schedule. Using a Spearman rank correlation test, the 30-year recovery period used by the FS produced significant correlations between ERA and diversity or dominance from the UCB studies (Table 7). Conversely, the 50-yr recovery period yielded lower, nonsignificant correlations (Steel and Torrie 1960). In one case, a significant correlation was determined for a 20-year recovery period but not for a 30-year period. These results suggest that shorter recovery periods better describe the association between recovering forest systems and aquatic response, as represented by the ERA and diversity or dominance, respectively.

Either the 20- or 30-year period allows for substantial vegetative and hydrologic recovery on the study watersheds. Revegetation typically reduces the risk of erosion and mass failure, and hydrologic recovery is furthered by litter accumulation, decreases in soil compaction, and recovery from soil surface armoring. Based on sampling of plants taller than 1.4 m in quadrats at 5 and 10 m from the channel along transects spaced at 40-m intervals, densities in the cut areas equaled or exceeded control areas in less than 15–17 years for the New York, Bit, and Taylor blocks (Fong 1991). Similarly, ten years after logging, harvested sites along Carnation Creek, British Columbia, contained vegetative cover equal to prelogging levels (Hartman and Scrivener 1990).

Thresholds for Detectable Change

We compared standard and modified ERA values with the UCB dominance and Shannon diversity data to determine if ERA and invertebrate community condition could be correlated. If ERA is linked to channel and aquatic ecosystem health, then such a correlation should exist. We hypothesized that as ERA increased, dominance of certain taxa should increase and diversity should decrease. We anticipated a curve with a moderate slope for low and moderate

Table 4. Dates and sizes of tractor-harvested areas in watersheds and streamside impact zones (SIZ) on control (C) and treatment (T) basins of the Taylor block

Watershed	Photo date	Year of harvest	Watershed cut area (ha)	SIZ cut area (ha)	Harvest method
Lower Taylor (T)	1941	None			
(601 ha/116 ha)	1953	None			
	1966	1964	4	1	Clear-cut
	1972	1966–1972	13	0	Clear-cut
	1972	1968–1969	3	0	Clear-cut
	1972	1970	36	2	Heavy selective
	1972	1972	25	5	Light selective
	1972	1972	30	13	Heavy selective
	1977	None			
	1982	1979	12	1	Light selective
	1987	1986	35	7	Cear-cut
Sum/% of area			158/26%	29/25%	
Upper Taylor (C)	1941	None			
(113 ha/11 ha)	1953	None			
	1966	None			
	1966	1968–1969	2	0	Clear-cut
	1972	None			
	1977	None			
	1982	None			
	1987	None			
Sum/% of area			2/2%	0/0%	
E. Branch Lights (C)	1941	None			
(828 ha/132 ha)	1953	None			
	1966	1958	335	32	Heavy selective
	1966	1958	97	17	Light selective
	1966	1958	6	0	Clear-cut
	1972	None			
	1977	None			
	1982	1981	3	0	Light selective
	1982	1981	3	0	Clear-cut
	1982	1982	53	4	Light selective
	1987	None			
Sum/% of area			497/60%	53/40%	

ERA values and then a steeper slope for ERA values past a threshold value of 15% or 20%. The comparison does show a decline in diversity with increasing disturbance levels (Figure 7). The comparison did not reveal an upper-limit threshold, possibly because the disturbance was not sufficiently severe, e.g., the ERA values did not exceed 10.5%.

The results do suggest, however, the presence of a minimum ERA threshold. Assuming a normal distribution of errors and using a two-phase linear regression model, successive fittings were made to find the join point that minimized the sums of squares (Hinkley 1971). For our sample, the join point between the two phases corresponded to an ERA value of 5.1%. The 95% confidence interval for this minimum ERA threshold was from 3.7 to 6.5%. Although the slope of line A is positive in Figure 7, the slope is *not* significantly different than zero and we *cannot* conclude that minor disturbances increase aquatic species diversity. The negative slope of line B is significant, however, at the 5% level, supporting the hypothesis of an inverse relationship between diversity and land disturbance for our sample. The minimum threshold value suggests that only low levels of disturbance in the streamside zone can be tolerated without at least a temporary effect on the aquatic community.

This finding suggests that land-management decisions based on hydrologic thresholds of concern are likely to have an effect on stream biota. For example, Haskins (1987) described a case study where the FS assessed the suitability of timber harvesting in selected watersheds of the Shasta-Trinity National Forest. Those watersheds with ERAs that exceeded a predetermined threshold of concern of 18% were not con-

Table 5. Dates and sizes of tractor-harvested areas in watersheds and streamside impact zones (SIZ) on control (C) and treatment (T) basins of the New York block

Watershed	Photo date	Year of harvest	Watershed cut area (ha)	SIZ cut area (ha)	Harvest method
Mid. New York (T)	1966	Pre-1966	245	40	Heavy selective
(466 ha/124 ha)	1972	1966–1972	7	4	Light selective
	1977	1974	28	13	Heavy selective
	1977	1974	6	0	Light selective
	1977	1974	4	3	Clear-cut
	1982	None			
	1987	1982–1987	30	0	Clear-cut
	1987	1982–1987	11	2	Heavy selective
	1987	1982–1987	27	5	Light selective
Sum/% of area			358/77%	67/54%	
New York Trib. (T)	1966	Pre-1966	44	3	Heavy selective
(92 ha/24 ha)	1972	None			
	1977	1974	19	6	Heavy selective
	1977	1974	4	3	Clear-cut
	1982	None			
	1987	1982–1987	9	0	Clear-cut
	1987	1982–1987	3	2	Heavy selective
Sum/% of area			79/86%	14/58%	
Upper New York (C)	1966	Pre-1966	200	36	Heavy selective
(347 ha/89 ha)	1972	1966–1972	7	4	Light selective
	1977	1974	3	1	Heavy selective
	1977	1974	0	0	Light selective
	1982	None			
	1987	1982–1987	22	0	Clear-cut
	1987	1982–1987	16	5	Light selective
	1987	1982–1987	11	0	Light selective
	1987	1982–1987	8	0	Heavy selective
Sum/% of area			267/77%	46/52%	
Empire (C)	1966–1987	None			
(531 ha/127 ha)					
Sum/% of area			0/0%	0/0%	

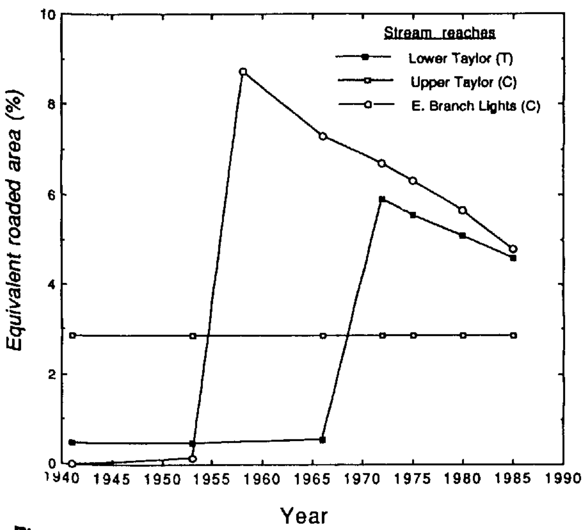

Figure 5. Trend in equivalent roaded area of the streamside impact zone for the Taylor block using a 30-year recovery period.

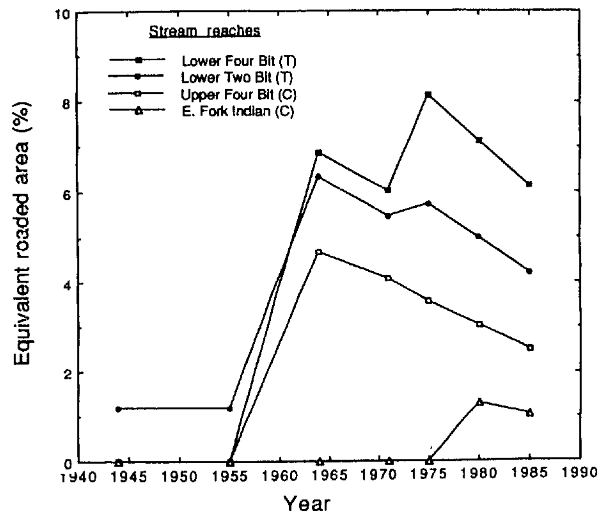

Figure 6. Trend in equivalent roaded area in the streamside impact zone for the Bit block using a 30-year recovery period.

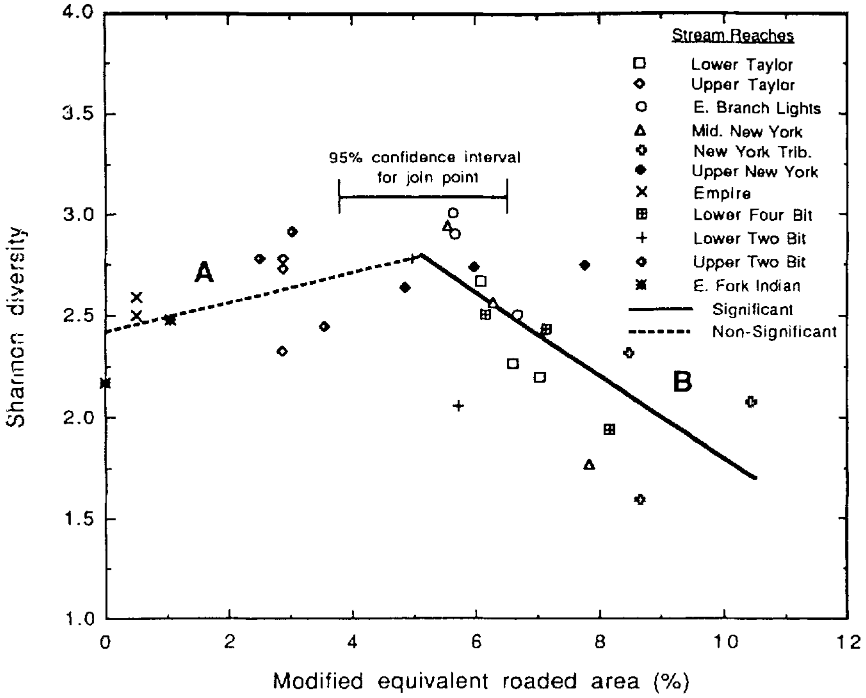

Figure 7. Beyond a minimum effect threshold, increases in roaded area result in a decreased diversity index as shown by line B (diversity data from Erman and others 1977; Erman and Mahoney 1980; Fong 1991).

sidered for further harvest activity. If a stream biota threshold of 5% had been used, fewer acres for timber harvesting would have been available, but effects on aquatic insects, and possibly the fisheries, would have been lessened. Based on the recent controversy in the Pacific Northwest over the spotted owl and the president's economic and environmental analyses, protection of near-stream zones is now much more stringent than it was in the past. With strict implementation of the new streamside protection zones, disturbances in these areas may not even reach the 5% threshold that we have correlated with declines in the diversity index.

Our results are from California, which is a Mediterranean hydrologic regime, and macroinvertebrate communities elsewhere may react differently to the same levels of disturbance. We hypothesize that land disturbance in other regions of the United States or in other countries would result in similar responses in the macroinvertebrate communities and that a method like the modified ERA would be appropriate outside of California. Analysis of the relationship between ERA and macroinvertebrate communities in other ecoregions, however, is certainly warranted. Because our study did not have ERA values in excess of 10.5%, additional research is needed to determine the influence of higher ERA values on the extent and duration of biological response.

Summary

Both the standard and modified forms of the ERA method yielded increases in the ERA index associated with harvests in the 1960s and 1970s, followed by decreasing ERA values after the harvest ceased. Long-term declines in ERA values should reflect revegetation and recovery of the aquatic ecosystem and increasing channel stability. Modifications to the standard ERA method that emphasized near-channel activities and contained 20- or 30-year recovery periods consistently yielded the largest correlation with diversity and dominance of macroinvertebrate communities. Changes in the ERA index were significantly correlated with changes in diversity and dominance. ERA values less than 5% were not associated with changes in diversity of aquatic insects, indicating that a minimum disturbance threshold may exist. As ERA values increased beyond 5.1%, a significant decrease in diversity was found.

Recommendations

The ERA methodology was designed primarily as a planning tool to aid Forest Service resource specialists in assessing the CWE of various management options at the forest or project level. Our experience indicates that disturbance indices for lotic systems should focus more explicitly on management activities within the

Table 6. Dates, sizes, and removal methods for clear-cut harvest areas in watersheds and streamside impact zones (SIZ) on control (C) and treatment (T) basins of the Bit block

Watershed	Photo date	Year of harvest	Watershed cut area (ha)	SIZ cut area (ha)	Removal method
Lower Four Bit (T)	1944	None			
(101 ha/21 ha)	1955	None			
	1964	1960–1964	11	3	Cable
	1964	1960–1964	6	1	Tractor
	1964	1960–1964	7	0	Cable
	1971	None			
	1975	1972	4	4	Cable
	1980	None			
	1986	None			
Sum/% of area			28/28%	8/33%	
Lower Two Bit (T)	1944	None			
(120 ha/38 ha)	1955	None			
	1964	1960–1964	10	5	Tractor
	1964	1960–1964	3	2	Cable
	1971	None			
	1975	1972	2	2	Cable
	1980	None			
	1986	None			
Sum/% of area			15/12%	9/24%	
Upper Four Bit (C)	1944	None			
(86 ha/15 ha)	1955	None			
	1964	1960–1964	18	3	Cable
	1964	1960–1964	2	0	Tractor
	1971	None			
	1975	None			
	1980	None			
	1986	None			
Sum/% of area			20/23%	3/20%	
Upper Two Bit (C)	1944–1986	None			
(61 ha/18 ha)					
Sum/% of area			0/0%	0/0%	
East Fork Indian (C)	1944–1975	None			
(87 ha/15 ha)	1980	1975–1980	12	1	Skyline[a]
	1980	1975–1980	8	0	Tractor/skyline
	1980	1975–1980	3	0	Tractor
	1980	1975–1980	1	1	Skyline
	1986	None			
Sum/% of area			24/28%	2/13%	

[a]Light selective harvest method coefficient used.

near-stream environment. This CWE analysis was based on knowledge of the long-term management history of some basins, but a comprehensive CWE program should also include long-term field monitoring of the biological characteristics of the aquatic systems. Resource managers need to have monitoring results to demonstrate that management is not adversely affecting a principal beneficial use on forested lands. As the FS shifts its focus toward integrated ecosystem management, it should expand the use of biological thresholds of concern in decisionmaking.

Acknowledgments

Don C. Erman (University of California at Berkeley) and his students originated the long-term biotic information, ecosystem recovery concept, and block design used in this study. Funding for a portion of Dr. Erman's work was provided by the Pacific Southwest Research Station, USDA Forest Service. Ken B. Roby (Plumas National Forest, Greenville, California) provided manuscript review and insight into equivalent roaded area estimation and helped us obtain land-use information for the study sites.

Table 7. Spearman correlation coefficients and significance of ERA values versus macroinvertebrate community parameters

Community parameters	Standard ERA			Modified ERA		
	20 yr	30 yr	50 yr	20 yr	30 yr	50 yr
Shannon diversity	−0.30	−0.19	−0.10	−0.42[a]	−0.43[a]	−0.24
Dominance (top taxa)	0.21	0.13	0.07	0.27	0.28	0.14
Dominance (top 5 taxa)	0.42[a]	0.31	0.25	0.47[a]	0.51[a]	0.35

[a]Significant at $P < 0.05$.

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Evaluating the Impacts of Logging Activities on Erosion and Suspended Sediment Transport in the Caspar Creek Watersheds[1]

Jack Lewis[2]

Abstract: *Suspended sediment has been sampled at both the North and South Fork weirs of Caspar Creek in northwestern California since 1963, and at 13 tributary locations in the North Fork since 1986. The North Fork gaging station (NFC) was used as a control to evaluate the effects of logging in the South Fork, in the 1970's, on annual sediment loads. In the most conservative treatment of the data, suspended loads increased by 212 percent over the total predicted for a 6-yr period commencing with the onset of logging. When the roles of the watersheds were reversed and the same analysis repeated to evaluate harvesting in the North Fork under California Forest Practice Rules in the 1990's, no significant increase was found at NFC in either annual suspended or bed load.*

With the advent of automatic pumping samplers, we were able to sample sediment concentration much more frequently in the 1980's. This allowed storm event loads from control watersheds in the North Fork to be used in a new regression analysis for NFC. According to this more sensitive analysis, for the 7-yr period commencing with the onset of logging, the sum of the suspended storm loads at NFC was 89 percent higher than that predicted for the undisturbed condition. The much greater increase after logging in the South Fork is too great to be explained by differences in sampling methods and in water years, and appears to be the result of differences in road alignment, yarding methods, and stream protection zones.

Similar analyses of storm event loads for each of the treated subwatersheds in the North Fork suggested increased suspended loads in all but one of the tributaries, but effects were relatively small or absent at the main stem locations. Of watersheds with less than 50 percent cut, only one showed a highly significant increase. The greater increase in sediment at NFC, compared to other main-stem stations, is largely explained by a 3,600-m^3 landslide that occurred in 1995 in a subwatershed that drains into the main stem just above NFC. Differences among tributary responses can be explained in terms of channel conditions.

Analysis of an aggregated model simultaneously fit to all of the data shows that sediment load increases are correlated with flow increases after logging. Field evidence suggests that the increased flows, accompanied by soil disruption and intense burning, accelerated erosion of unbuffered stream banks and channel headward expansion. Windthrow along buffered streams also appears to be important as a source of both woody debris and sediment. All roads in the North Fork are located on upper slopes and do not appear to be a significant source of sediment reaching the channels.

The aggregated model permitted evaluation of certain types of cumulative effects. Effects of multiple disturbances on suspended loads were approximately additive and, with one exception, downstream changes were no greater than would have been expected from the proportion of area disturbed. A tendency for main-stem channels to yield higher unit-area suspended loads was also detected, but after logging this was no longer the case in the North Fork of Caspar Creek.

[1] An abbreviated version of this paper was presented at the Conference on Coastal Watersheds: The Caspar Creek Story, May 6, 1998, Ukiah, California.

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Soil erosion and mass movement play major roles in shaping the landscapes that surround us. These processes complement those that build mountains and soils, resulting in landforms such as valleys, ridges, stream channels, and flood plains. Human activities that change the balances between these processes can have consequences that are detrimental to humans and the ecosystems we depend on. Human activities often lead to an acceleration of soil movement, net soil losses from hillslopes, and increases in sediment transport and deposition in stream channels. When soil erosion and mass movement directly damage roads, bridges, and buildings, the costs are immediate and obvious. Direct effects on ecosystem function and site productivity are also serious issues in many areas. Indirect impacts on downstream water quality and stream channel morphology, however, are often of greater concern.

Sediment-laden water supplies reduce the capacity of storage reservoirs and may require additional treatment to render the water drinkable. Sediment in irrigation water shortens the life of pumps and reduces soil infiltration capacity. Water quality is also an important issue for recreational water users and tourism.

Impacts of water quality on fish and aquatic organisms have motivated much of the research being presented at this conference. High sediment concentrations can damage the gills of salmonids and macroinvertebrates (Bozek and Young 1994, Newcombe and MacDonald 1991). High turbidity can impair the ability of fish to locate food (Gregory and Northcote 1993) and can reduce the depth at which photosynthesis can take place. However, suspended sediment is not always detrimental to fish, and indexes based on duration and concentration are unrealistically simplistic (Gregory and others 1993). Turbidity, can, for example, provide cover from predators (Gregory 1993).

If stream channels cannot transport all the sediment delivered from hillslopes, they will aggrade, resulting in increased risks for overbank flooding and bank erosion. It was this sort of risk, threatening a redwood grove containing the world's tallest tree, that motivated the expansion of Redwood National Park in 1978 (U.S. Department of Interior 1981). Accelerated delivery of sediment to streams can result in the filling of pools (Lisle and Hilton 1992), and channel widening and shallowing. Hence, fish rearing habitat may be lost, and stream temperatures often increase. Excessive filling in spawning areas can block the emergence of fry and bury substrates that support prey organisms. Settling and infiltration of fine sediments into spawning gravels reduces the transport of oxygen to incubating eggs (Lisle 1989) and inhibits the removal of waste products that accumulate as embryos develop (Meehan 1974). If aggradation is sufficient to locally eliminate surface flows during the dry season, fish can lose access to good upstream habitat or become trapped in inhospitable environments.

How Do Harvest Practices Affect Sediment Movement?

Figure 1 displays some of the mechanisms linking harvest activities with instream sediment transport. It is impossible to show all the potential interactions in only two dimensions, but the figure does hint at the complexity of controls on sediment movement. Timber harvest activities can accelerate erosion primarily through felling, yarding, skidding, building and using roads and landings, and burning.

Felling

Removing trees reduces evapotranspiration and rainfall interception, thus resulting in wetter soils (Keppeler and others 1994, Ziemer 1968). Loss of root strength and wetter soils can decrease slope stability (O'Loughlin and Ziemer 1982, Ziemer 1981). Trees near clearcut edges face increased wind exposure and become more susceptible to blowdown (Reid and Hilton, these proceedings), disrupting soils if trees become uprooted. Addition of woody debris to channels can cause scouring of the banks and channel, but also can reduce sediment transport by increasing channel roughness and trapping sediment (Lisle and Napolitano, these proceedings). The effects of felling upon erosion can be altered by controlling the quantity and the spatial and temporal patterns of cutting.

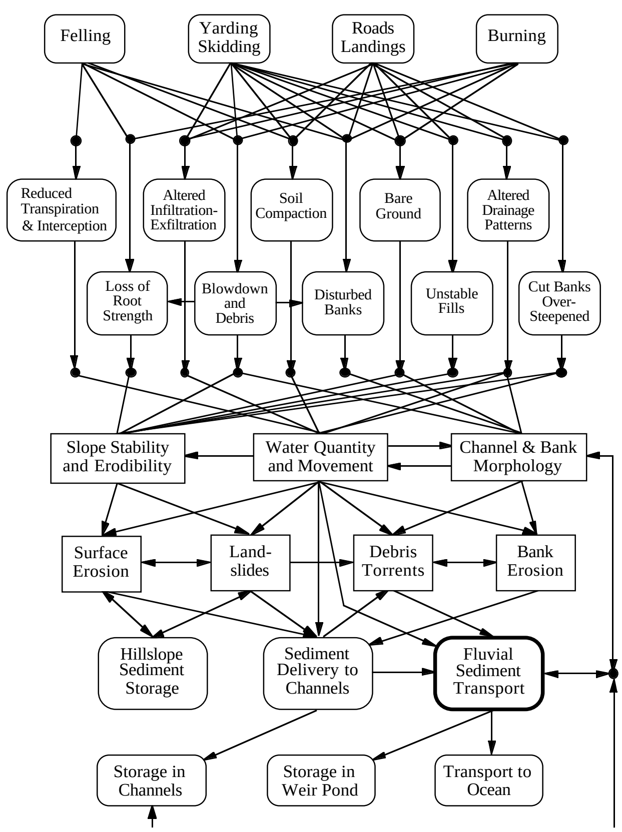

Figure 1—Conceptual diagram showing the major pathways through which logging activities influence fluvial sediment transport.

Yarding and Skidding

Heavy equipment compacts soils, decreasing infiltration and percolation rates and increasing surface water. If vegetation and duff are removed, the underlying soils become vulnerable to surface erosion. The pattern of yarding and skidding can alter drainage paths and redirect water onto areas that may be more likely to erode than naturally evolved channels. Damage from yarding and skidding is controlled primarily by the type of equipment, the care exercised by the equipment operator, timing of operations, landing location, and yarding direction.

Roads and Landings

Roads and landings have similar, but usually more pronounced, impacts as yarding and skidding, and their presence can greatly increase landslide risk. Compaction of the road bed can impede subsurface drainage from upslope areas, resulting in increased pore water pressures (Keppeler and Brown, these proceedings). Road cuts and fills are vulnerable to accelerated runoff and surface erosion, and are particularly vulnerable to slumping, especially on steep slopes or if the fill or sidecast material has not been properly compacted. Although roads and landings may be only a small part of the total forest area, they are responsible for a disproportionate amount of the total erosion (McCashion and Rice 1983, Swanson and Dyrness 1975), often more than half. The erosional impact of roads and landings can be managed through road alignment, design and construction, drainage systems, type and timing of traffic, and maintenance.

Burning

Burning can increase erodibility by creating bare ground, and hot burns can delay revegetation by killing sprouting vegetation. In some cases, burning can accelerate revegetation by releasing or scarifying seeds and preparing a seed bed. Burning in areas with sandy soils can create water-repellent soils and increase surface runoff (DeBano 1979). The effect of burning on erosion depends primarily on the temperature of the burn, soil cover, and soil and vegetation types. Soil moisture, wind, air temperature, humidity, slope steepness, and fuel abundance and distribution are the major factors affecting burn temperatures.

Site Factors

Some sites are particularly vulnerable to mass wasting, and these sites, while occupying a small part of the landscape, have been found to be responsible for a large proportion of the total erosion in northwestern California (Dodge and others 1976, Rice and Datzman 1981). In the Critical Sites Erosion Study, an evaluation of 157 mass failure sites (>153 m^3) and 326 randomly selected control sites from logged areas in northwestern California, Durgin and others (1989) concluded that management and site factors played an equal role in road failures. In contrast, management factors were secondary to site factors on hillslopes. The primary site factors associated with mass failures were steep slopes, noncohesive soils and fill materials, and incompetent underlying regolith. Most failures were associated with the concentration of subsurface water, as evidenced by perennial seeps, poorly drained soils, phreatic vegetation, and locations in swales, inner gorges, and lower slope positions. Previous slope failures were also evident at many of the sites. The primary management factors associated with mass failures were steep or overloaded fill slopes, steep cut banks, and inadequate maintenance of roads and drainage systems. A field procedure for estimating the probability of mass failure was also developed (Lewis and Rice 1990, Rice and Lewis 1991) from the Critical Sites Erosion Study.

Connecting Forest Practices with Water Quality

It is often difficult to identify the causes of erosion. Factors such as increased soil water or reduced root strength are not directly observable. Landslides are normal, stochastic, geomorphic events in many undisturbed areas. Therefore, it may be impossible to show that a landslide in a logged area would not have occurred had the area been treated differently.

There is usually a great deal of uncertainty in determining when and how much sediment from an erosion feature was delivered to a stream channel. And it is even more difficult to determine the origin of suspended sediment that has been measured at a gaging station.

Hence, many studies are correlative and rely on statistics to identify relations between disturbance and water quality. In environmental research, it is difficult to execute an experimental design that permits wide inference. The best designs require randomly assigning the treatments of interest to a large number of similar experimental units. The random assignment reduces the likelihood of associations between treatments and characteristics that might affect the response of some subset of experimental units. When studying a highly variable response such as sediment transport, large sample sizes are needed to detect changes even when the changes are substantial.

When the experimental unit is a watershed, it is usually impractical to randomly assign treatments or monitor a large number of watersheds. Instead, we use watersheds with similar physical characteristics and subject to similar environmental influences, and we repeat measurements before and after treatments are applied, maintaining at least one watershed as an untreated control throughout the study. If the relationship between measurements in the treated and control watersheds changes after treatment, then we can reason that the change is probably due to the treatment, unless some chance occurrence (unrelated to the treatments) affected only one of the watersheds. In reality, we have little control over such chance occurrences. For example, there is no guarantee that rainfall intensities will be uniform over the entire study area.

Such a paired-watershed design can provide a basis for concluding whether a change occurred (Chow 1960, Wilson 1978) and can be used to estimate the magnitude of changes. If chance occurrences can be eliminated, effects can be attributed to the *overall* treatment. If multiple watersheds are included in the design, it may be useful to relate the magnitude of response to disturbances such as proportion of area logged, burned, compacted by tractors, etc. But, without additional evidence, nothing can be concluded about specific causative mechanisms. Conclusions should be *consistent* with the statistical evidence, but cause and effect must be inferred non-statistically, by relating the results to concurrent studies of other responses and physical processes, field observations, and similar observations made elsewhere by others.

Study Area

The Caspar Creek Experimental Watersheds are located about 7 km from the Pacific Ocean in the Jackson Demonstration State Forest, Mendocino County, California (Preface, fig. 1, these proceedings). Until the 1970's, both the 424-ha South Fork and 473-ha North Fork watersheds were covered by second-growth redwood forests, originally logged between 1860 and 1904. Both watersheds are underlain by sandstones and shales of the Franciscan assemblage. Rainfall averages about 1,200 mm yr^{-1}, 90 percent of which falls during October through April, and snow is rare. The location, topography, soils, climate, vegetation, and land use history are described in detail by Henry (these proceedings). The geology and geomorphology are described by Cafferata and Spittler (these proceedings).

Methods

South Fork Treatment

The South Fork of Caspar Creek was roaded in the summer of 1967 and selectively logged in 1971-1973, before Forest Practice Rules were mandated in California by the Z'Berg Nejedly Forest Practice Act of 1973. About 65 percent of the stand volume was removed. In contrast with later logging in the North Fork, 75 percent of the roads in the South Fork were located within 60 m of a stream, all yarding was done by tractor, ground disturbance amounted to 15 percent of the area, and there were no equipment exclusion zones. Details are provided by Henry (these proceedings) and by Rice and others (1979). The North Fork was used as a control watershed to evaluate the effects of logging in the South Fork until the North Fork phase of the study was begun in 1985.

North Fork Treatments

The subwatershed containing units Y and Z (Preface, fig. 2, these proceedings) of the North Fork was logged between December 1985 and April 1986. At the time, this area was thought to have different soils than the remainder of the North Fork, so it was omitted from the study plan that specified logging would begin in 1989. The remainder of the North Fork logging took place between May 1989 and January 1992. Three subwatersheds (HEN, IVE, and MUN) were left uncut throughout the study for use as controls. Henry (these proceedings) summarizes the logging sequence. Briefly, 48 percent of the North Fork (including units Y and Z) was clearcut, 80 percent of this by cable yarding. Tractor yarding was restricted to upper slopes, as were haul roads, spur roads, and landings. Ground disturbance from new roads, landings, skid trails, and firelines in the North Fork amounted to 3.2 percent of the total area. Streams bearing fish or aquatic habitat were buffered by selectively logged zones 23-60 m in slope width, and heavy equipment was excluded from these areas.

Suspended Sediment and Turbidity Measurements

Accurate suspended sediment load estimation in small rain-dominated watersheds like Caspar Creek depends upon frequent sampling when sediment transport is high. Sediment concentrations are highly variable and inconsistently or poorly correlated with water discharge (Colby 1956, Rieger and Olive 1984). Since the 1960's, manual sampling methods have been standardized by the U.S. Geological Survey. However, adequate records are rare because it is inconvenient to sample at all hours of the night and weekends. Errors of 50-100 percent are probably typical when sampling is based on convenience (Thomas 1988, Walling and Webb 1988).

In the South Fork phase of the study from 1963 to 1975, sediment sampling was semi-automated by rigging bottles in the weir ponds at different heights. These *single-stage samplers* (Inter-Agency Committee on Water Resources 1961) filled at known stages during the rising limb of the hydrograph, but the much lengthier falling limb was sampled using DH-48 depth-integrating hand samplers (Federal Inter-Agency River Basin Committee 1952) and, in most cases, was not well-represented. In 1974 and 1975, the number of DH-48 samples was increased greatly and, in 1976, the single-stage samplers were replaced by pumping samplers. The average number of samples collected was 58 per station per year in 1963-1973 and 196 per station per year in 1974-1985.

During the North Fork phase of the study, in water years 1986-1995, the North Fork weir (NFC), the South Fork weir (SFC) and 13 other locations in the North Fork were gaged for suspended sediment and flow (Preface, fig. 2, these proceedings). Pumping samplers were controlled using programmable calculators and circuit boards that based sampling decisions on real-time stage information (Eads and Boolootian 1985). Sampling times were randomly selected using an algorithm that increased the average sampling rate at higher discharges (Thomas 1985, Thomas 1989). Probability sampling permitted us to estimate sediment loads and the variance of those estimates without bias. We also sent crews out to the watershed 24 hours a day during storm events to replace bottles, check equipment, and take occasional, simultaneous, manual and pumped samples. The average number of samples collected in 1986-1995 was 139 per station per year.

In water year 1996, we began using battery-operated turbidity sensors and programmable data loggers to control the pumping samplers at eight gaging stations, and monitoring was discontinued at the remaining seven stations. Although turbidity is sensitive to particle size, composition, and suspended organics, it is much better correlated with suspended sediment concentration than is water discharge. A continuous record of turbidity provides temporal detail about sediment transport that is currently impractical to obtain by any other means, while reducing the number of pumped samples needed to reliably estimate sediment loads (Lewis 1996). However, because these turbidity sensors remain in the stream during measurement periods, they are prone to fouling with debris, aquatic organisms, and sediment, so it was still necessary to frequently check the data and clean the optics. The average number of samples collected in 1996 was 49 per station per year.

Suspended Sediment Load Estimation

The basic data unit for analysis was the suspended sediment load measured at a gaging station during a storm event or hydrologic year. Annual loads were estimated only for NFC and SFC and, to facilitate comparisons with the South Fork study, these were computed by Dr. Raymond Rice using the same methods as in an earlier analysis (Rice and others 1979). This involved fitting sediment rating curves by eye, multiplying the volume of flow in each of 19 discharge classes by the fitted suspended sediment concentration at the midpoint of each class, and summing. As technology has improved over the years, our methods of sample selection have improved. Thus, although the computational scheme for estimating annual loads was repeated in both studies, the sampling bias has changed, and caution must be used when comparing the sediment loads from the two studies.

For estimating storm loads in 1986-1995, the concentrations between samples were computed using interpolations relating concentration to either time or stage. Concentration was first adjusted to obtain cross-sectional mean concentrations using regressions based on the paired manual and pumped samples. For those events in which probability sampling was employed, loads and variances were also estimated using appropriate sampling formulae (Thomas 1985, Thomas 1989). However, Monte Carlo simulations (Lewis and others 1998), showed that the interpolation methods were more accurate (lower mean square error). Based on the variance estimates and simulations, the median error of our estimates for storm events was less than 10 percent.

For estimating storm loads in 1996, concentration was predicted using linear regressions, fit to each storm, of concentration on turbidity. This method produced load estimates with the same or better accuracy than before, while substantially reducing the number of samples collected (Lewis 1996). Time or stage interpolation was employed for periods when turbidity information was unavailable.

Total Sediment Load Estimation

The bedload and roughly 40 percent of the suspended load settle in the weir ponds, and thus are not measured at NFC and SFC. The weir ponds are surveyed annually to estimate total sediment load (suspended plus bedload) by summing the pond accumulations and sediment loads measured at the weirs. Pond volumes are converted to mass based on a density of 1,185 kg m^{-3}. In some of the drier years of record (1972, 1976, 1987, 1991, 1992, and 1994), negative pond accumulations have been recorded. These values may result from settling or measurement errors, but some of the values were too large in magnitude to have resulted from settling alone, so negative values were converted to zero before adding pond accumulations to suspended loads. In the results below, only those that explicitly refer to *total* sediment load include any sediment that settled in the weir ponds.

Erosion Measurements

Starting in 1986, a database of failures exceeding 7.6 m^3 (10 yd^3) was maintained in the North Fork. This inventory was updated from channel surveys at least once a year. Road and hillslope failures were recorded when they were observed, but an exhaustive search was not conducted. Volume estimates were made using tape measurements of void spaces left by the failures, except in a few cases where more accurate survey methods were used. For each failure, crews recorded void volume, volume remaining at the site (starting in 1993), location, distance to nearest channel, and any association with windthrow, roads, or logging disturbance.

Discrete failures such as those included in the failure database are relatively easy to find and measure. In contrast, surface erosion is difficult to find and sample because it is often dispersed or inconspicuous. To obtain an estimate of dispersed erosion sources, erosion plots were randomly selected and measured in each subwatershed. Rills, gullies, sheet erosion, and mass movements were measured on independent samples of road plots and 0.08-ha circular hillslope plots. Road plots consisted of 1.5-m wide bands oriented perpendicular to the right-of-way, plus any erosion at the nearest downslope diversion structure (water bar, rolling dip, or culvert). A total of 175 hillslope plots and 129 road plots were measured. These data were collected for a sediment delivery study and are summarized in a separate report by Rice (1996).

Analyses and Results

Annual Sediment Loads after Logging the South Fork

Linear regressions between the logarithms of the annual suspended sediment loads at the two weirs were used to characterize (1) the relationship of SFC to NFC before the 1971-1973 logging in the South Fork and (2) the relationship of NFC to SFC before the 1989-1992 logging in the North Fork.

The calibration water years used in the South Fork analysis were 1963-1967, before road construction. The sediment load in 1968, after road construction, did not conform to the pretreatment regression (*fig. 2a*), but the data from the years 1969-1971 were not significantly different from the 1963-1967 data (Chow test, $p = 0.10$). In 1968, the increase in suspended load was 1,475 kg ha^{-1}, an increase of 335 percent over that predicted for an undisturbed condition. The years 1972-1978 (during and after logging) again differed from the pretreatment regression. Water year 1977 was missing owing to instrument malfunction. By 1979, the suspended sediment load at SFC had returned to pretreatment levels. The increased suspended load after logging amounted to 2,510 kg $ha^{-1}yr^{-1}$, or an increase of 212 percent over that predicted for the 6-yr period by the regression. (Predictions were corrected for bias when backtransforming from logarithms to original units.) The greatest absolute increases occurred in the years 1973 and 1974, followed by 1975 (*fig. 2b*).

A pair of large landslides (one in each watershed) occurred during hydrologic year 1974, complicating the analysis by Rice and others (1979), where the North Fork's sediment load was adjusted downward because most of the North Fork slide reached the stream, while most of the South Fork slide did not. However, that year did not appear anomalous in my analysis, and I did not make any adjustments. But the unadjusted prediction requires extrapolation of the regression line well beyond the range of the pretreatment

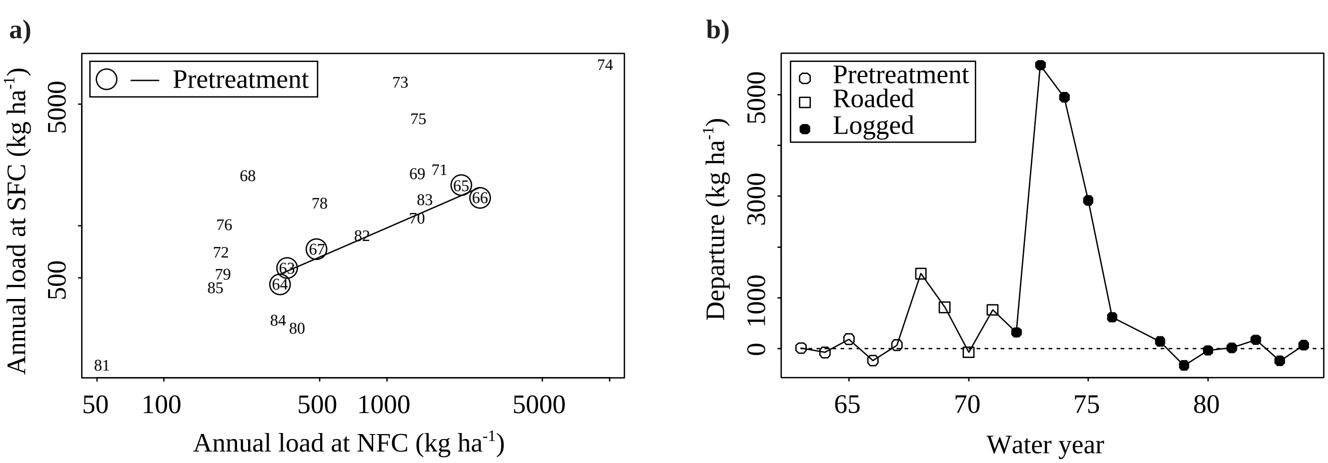

Figure 2—(a) Relation between estimated annual suspended sediment loads at South Fork Caspar Creek (SFC) and North Fork Caspar Creek (NFC) from 1963 to 1985. Pretreatment regression line is fit to the water years before roading and logging activity in the South Fork. (b) Time series of departures from the regression line.

data, so it is still suspect. If the adjustment of Rice and others (1979) is applied in my analysis, the revised increase in suspended sediment load is 2,835 kg $ha^{-1}yr^{-1}$, or an increase of 331 percent over that predicted for the 6-yr period. The adjusted figure reported for the 5-yr period (1972-1976) by Rice and others was 3,245 kg $ha^{-1}yr^{-1}$, an increase of 354 percent over that predicted.

Although no statistically significant logging effect on pond accumulation was detected, regression analysis using total sediment load (including data from 1974) revealed a similar pattern of impacts as that of the suspended load. The increased total sediment load after logging of the South Fork amounted to 2,763 kg $ha^{-1}yr^{-1}$, or an increase of 184 percent over that predicted for the 6-yr period by the regression.

Annual Sediment Loads after Logging the North Fork

The calibration period used in the North Fork analysis includes 1979-1985, the years after the South Fork's apparent recovery, as well as 1963-1967. The years 1986-1989 were not included in the calibration period because the Y and Z units were logged in 1985 and 1986. Applying the Chow test, neither 1986-1989 ($p = 0.43$) nor 1990-1995 ($p = 0.53$) was found to differ significantly from the suspended sediment calibration regression (*fig. 3a*). The (nonsignificant) departures from the regression predictions averaged 118 kg $ha^{-1}yr^{-1}$, amounting to just 28 percent above that predicted for the 6-yr period by the regression (*fig. 3b*). No effect was detected for pond accumulation by itself or total sediment load. For total sediment load, the (nonsignificant) departures from the regression predictions averaged -80 kg $ha^{-1}yr^{-1}$, or 8 percent below that predicted for the 6-yr period by the regression.

The absolute numbers reported in the above and earlier analyses of the South Fork logging (Rice and others 1979) must be viewed with reservation. The suspended load estimates were based on hand-drawn sediment rating curves describing the relation between the concentration of samples collected in a given year to the discharge levels at which they were collected. In several years, samples were not available from all discharge classes, so it was necessary to extrapolate the relation between concentration and discharge to higher or lower unrepresented classes. Also, a majority of the samples from the years 1963-1975 were collected using single-stage samplers that are filled only during the rising limb of hydrographs. In most storm events we have measured at Caspar Creek, the concentrations are markedly higher on the rising limb of the hydrograph than for equivalent discharges on the falling limb (e.g., *fig. 4*). Therefore, the fitted concentrations were likely too high. A plot of estimated sediment loads at NFC against annual water yield for the pre-logging years (*fig. 5*) suggests that there may be a positive bias during the single-stage years. The error associated with this method certainly varies from year to year, depending on the numbers of single-stage and manual samples and their distribution relative to the hydrographs. However, the plot indicates that loads were overestimated by a factor of between 2 and 3 in the range where most of the data occur. A comparison of the annual loads for the years 1986-1995 with annual sums of storm loads (the most accurate) shows very little bias, indicating that bias in the early years resulted mainly from sampling protocols rather than the computational method, which was the same for all years in this analysis.

North Fork Analysis Using Unlogged Subwatersheds as Controls

Because of improved and more intensive sampling methods, the suspended sediment loads for storm events beginning in 1986 are known far more accurately than the annual loads used in the NFC/SFC contrasts presented above. Four unlogged control watersheds were available (HEN, IVE, MUN, and SFC) for the analysis of storm loads. Unfortunately, only one large storm was available before logging. That storm was missed at SFC because of pumping sampler problems. Because of various technical difficulties, not all storms

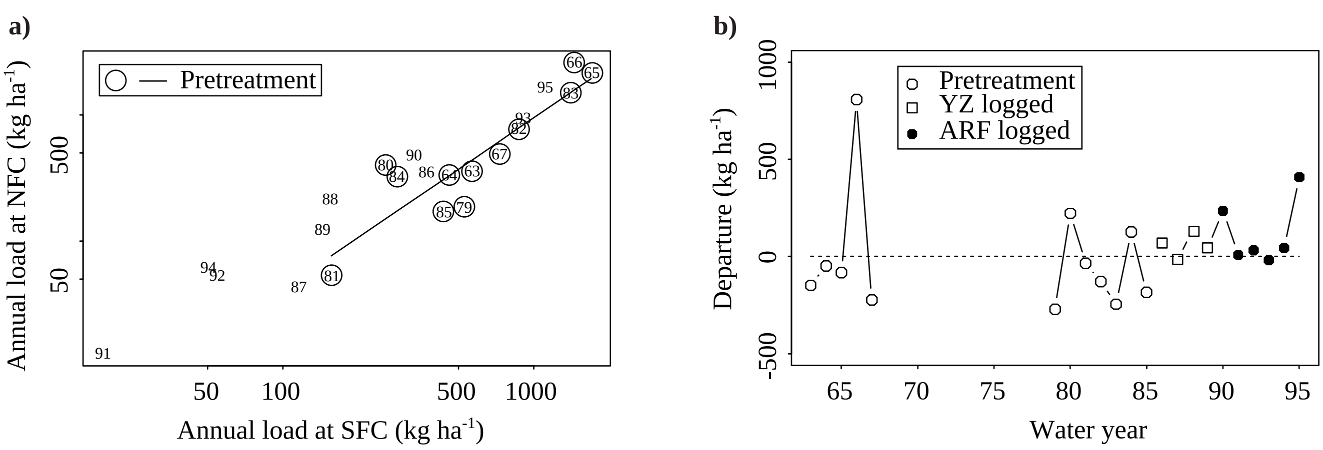

Figure 3—(a) Relation between estimated annual suspended sediment loads at North Fork Caspar Creek (NFC) and South Fork Caspar Creek (SFC) from 1963 to 1967 and 1979 to 1995, excluding years when sediment was elevated following logging in the South Fork. Pretreatment regression line is fit to the water years before roading and logging activity in the North Fork. (b) Time series of departures from the regression line.

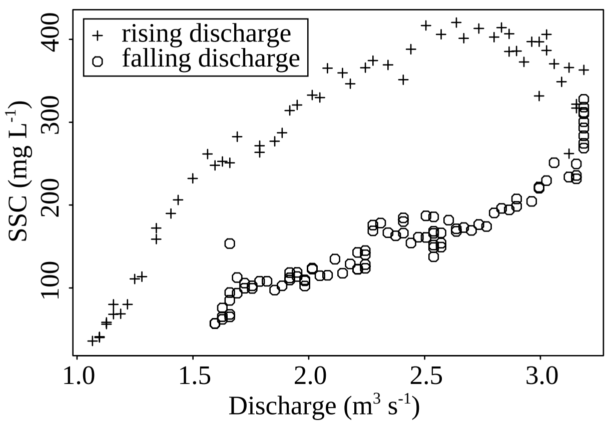

Figure 4—Storm event at lower main-stem station ARF, January 13-14, 1995, with water discharge and laboratory sediment concentrations (SSC) at 10-minute intervals.

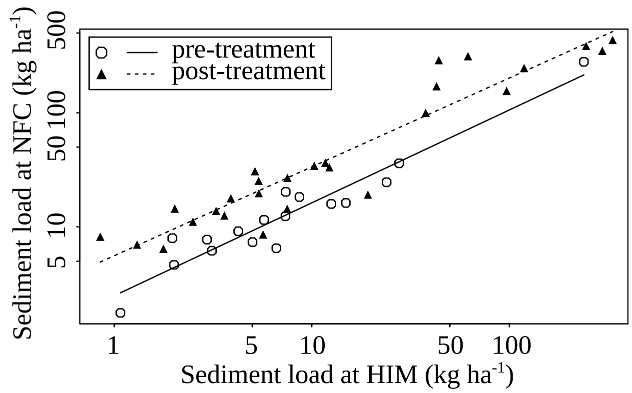

Figure 6—Relation between storm suspended sediment loads at North Fork Caspar Creek (NFC) and HIM control (mean suspended load of unlogged tributaries HEN, IVE, and MUN) from 1986 to 1995. Pretreatment regression line is based on storms in water years 1986-1989, before the major logging activity began.

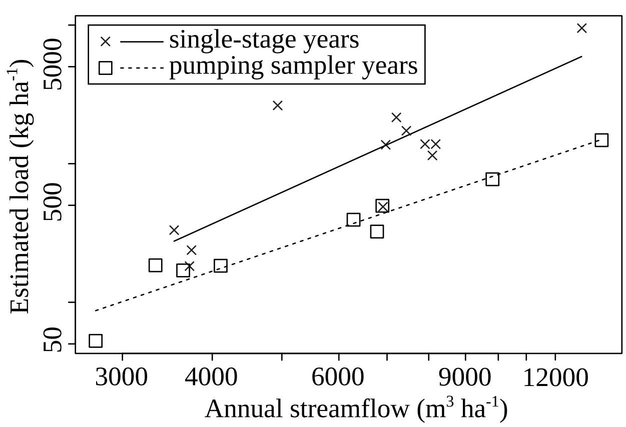

Figure 5—Relations between estimated annual suspended sediment loads and annual streamflow at North Fork Caspar Creek (NFC) prior to logging. Illustrates that load estimates based on sediment rating curves depend systematically on sampling protocols.

were adequately sampled at each station. However, the sample size for analyses was increased by using the mean of available data from the three tributary control watersheds, HEN, IVE, and MUN, in each storm. (SFC was eliminated because it had lower pretreatment correlations with the North Fork stations.) This mean (denoted HIM) provided a pretreatment sample size of 17 storms. The more accurate sediment loads, better controls, and larger sample size gave this analysis greater reliability and increased power to detect changes than the annual load analysis.

A weakness in analyses of logging effects at NFC was the need to use 1986-1989 as a calibration period even though 12 percent of the area had been clearcut. The clearcut area might be expected to somewhat diminish the size of the effect detected. The occurrence of only one large storm event before logging is mitigated by the fact that it was thoroughly sampled at both NFC and the three controls. An average of 59 sample bottles were collected at each of the four stations, and all the standard errors were less than 10 percent of the estimated loads, so there is little doubt about this point's validity.

Figure 6 shows regression lines fit to the suspended storm loads at NFC versus those at HIM before and after logging began in the spring of 1989. There was clearly an increase in suspended loads in small storms after logging began. In large storms there also seems to be an effect, although some post-treatment points are very close to the one large pretreatment point. The Chow test for a change after logging was significant with $p = 0.006$. The increases over predicted load, summed over all storms in the post-treatment period, average 188 kg ha^{-1}yr^{-1}, and amount to an 89 percent increase over background. The storms in this analysis represent 41 percent of the 1990-1996 streamflow at NFC, but carried approximately 90 percent of the suspended sediment that passed over the weir (based on figure 2 of Rice and others 1979).

A 3,600-m^3 landslide that occurred in the Z cut unit (Preface, fig. 2, these proceedings) increased sediment loads at the NFC gaging station starting in January 1995. NFC was the only gage downstream from this slide. The sum of suspended loads from storms preceding the landslide was 47 percent higher (64 kg ha^{-1}yr^{-1}) than predicted. The sum of suspended loads from storms after the landslide was 164 percent higher (150 kg ha^{-1}yr^{-1}) than predicted.

Individual Regressions for Subwatersheds

Similar analyses for each of the subwatersheds in the North Fork (*fig. 7* and *table 1*) indicate increased suspended sediment loads in all the clearcut tributaries except KJE. Sediment loads in the KJE watershed appear to have decreased after logging. The only partly clearcut watershed on a tributary (DOL) also showed highly significant increases in sediment loads. The upper main-stem stations (JOH and LAN) showed no effect after logging, and the lower main-stem stations (FLY and ARF) experienced increases only in smaller storms. Summing suspended sediment over *all* storms, the four main-stem stations all showed little or no change (*table 1*).

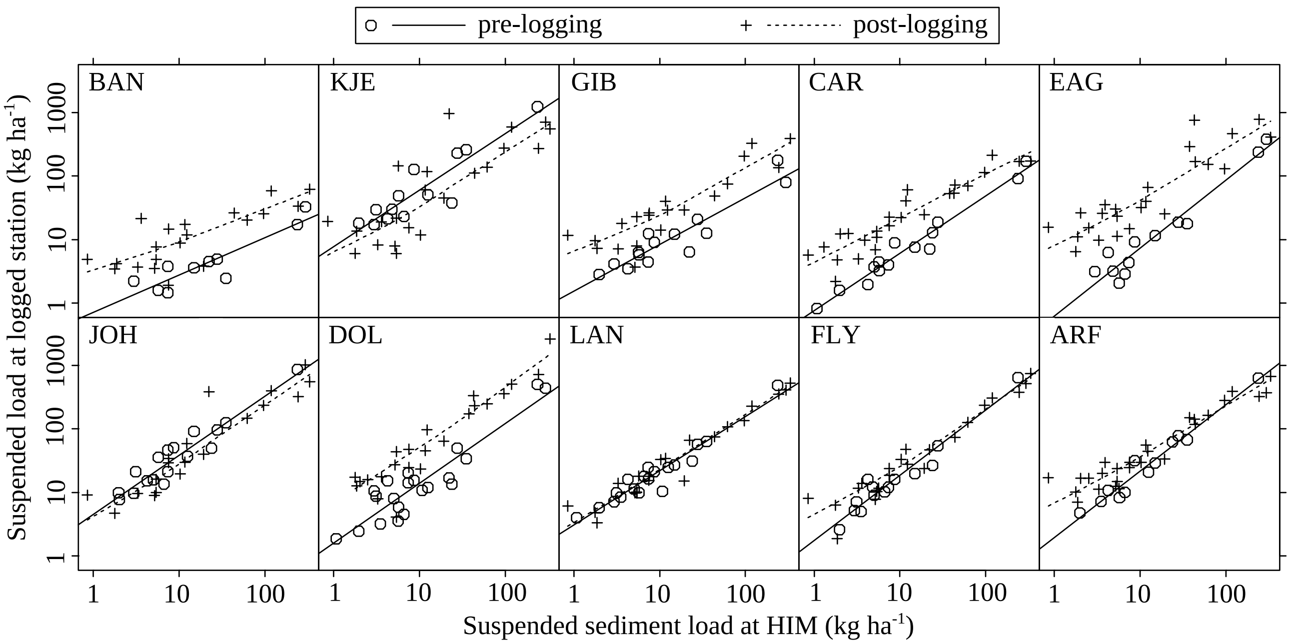

Figure 7—Relations between storm suspended sediment loads at logged subwatersheds in the North Fork and HIM control (mean suspended load of unlogged tributaries HEN, IVE, and MUN) from 1986 to 1995. Pre-logging regression lines are based on pretreatment years that are specific to each subwatershed. Post-logging relations are not assumed to be linear, hence were fitted by locally weighted regression (Cleveland 1993).

Table 1*—Summary of changes in suspended sediment load (summed over storms) after logging in North Fork subwatersheds. Predicted loads are computed from pre-treatment linear regressions between the logarithms of the storm sediment load in the treated watershed and the mean of the storm sediment loads at the control watersheds HEN, IVE, and MUN. Predictions were corrected for bias when back-transforming from logarithmic units. The number of years in the post-logging period varies from 4 to 6, depending upon when the watershed was logged and whether or not monitoring was discontinued in water year 1996.*

Treated watershed	Number of years	Observed ($kg\ ha^{-1}\ yr^{-1}$)	Predicted ($kg\ ha^{-1}\ yr^{-1}$)	Change ($kg\ ha^{-1}\ yr^{-1}$)	Change (%)
ARF	4	505	591	-86	-15
BAN	4	85	28	57	203
CAR	5	240	108	132	123
DOL	5	1130	306	824	269
EAG	5	710	210	500	238
FLY	5	536	555	-19	-3
GIB	4	358	119	239	200
JOH	5	667	865	-198	-23
KJE	5	821	1371	-551	-40
LAN	5	420	400	20	5
NFC	6	465	246	219	89

Aggregated Regression Model for Subwatersheds

To evaluate the relationships between suspended sediment load increases and possible explanatory variables, an aggregated regression model was fit simultaneously to all the subwatershed storms. The model utilized 367 estimated loads from 51 storms when HIM was used as the control or 333 estimated loads from 43 storms when HI (the mean of HEN and IVE) was used. Two regression coefficients were fitted for each watershed. A number of disturbance measures were considered (*table 2*), as well as an area term designed to describe cumulative effects, and a term explaining sediment increases in terms of flow increases. A great deal of effort went into developing a model that would permit valid tests of hypotheses concerning cumulative watershed effects. Therefore, the response model is coupled with a covariance model that describes variability in terms of watershed area and correlation among subwatershed responses as a function of distance between watersheds. These models were solved using the method of maximum likelihood and will be described in detail in a separate publication (Lewis and others 1998).

Departures from sediment loads predicted by the aggregated model for undisturbed watersheds were modest. The median increase in storm sediment load was 107 percent in clearcuts and 64 percent in partly clearcut watersheds. The median annual increase was 109 percent (58 kg $ha^{-1}yr^{-1}$) from clearcut watersheds and 73 percent (46 kg $ha^{-1}yr^{-1}$) from partly clearcut watersheds. The absolute flux values are underestimated somewhat because they include only sediment measured in storms, and no effort has been made to adjust for missing data. However, the major storms have been included, and virtually all of the sediment is transported during storms. Uncertainty due to year-to-year variability is certainly a much greater source of error.

The most important explanatory variable identified by the model was increased volume of streamflow during storms. Storm flow predictions (Ziemer, these proceedings) were based on an aggregated model analogous to that used for predicting sediment loads. The ratio of storm sediment produced to that predicted for an unlogged condition was positively correlated to the ratio of storm flow produced to that predicted for an unlogged condition (*fig. 8*). This result is not unexpected because, after logging, increased storm flows in the treated watersheds provide additional energy to deliver and transport available sediment and perhaps to generate additional sediment through channel and bank erosion.

Whereas individual watersheds show trends indicating increasing or decreasing sediment loads, there is no overall pattern of recovery apparent in a trend analysis of the residuals from the model (*fig. 9a*). This is in contrast with the parallel model for storm flow volume (*fig. 9b*), and suggests that some of the sediment increases are unrelated to flow increases.

Other variables found to be significant were road cut and fill area, and, in models using the HI control, the length of unbuffered stream channel, particularly in burned areas. Under California Forest Practice Rules in effect during the North Fork logging, buffers were not required for stream channels that do not include aquatic life and are not used by fish within 1,000 feet downstream except in confluent waters. As discussed earlier, one must be cautious about drawing conclusions about cause and effect when treatments are not randomly assigned to experimental units and replication is limited. Increases in sediment load in one or two watersheds can create associations with any variable that happens to have higher values in those watersheds, whether or not those variables are physically related to the increases. In this study, the contrast in response is primarily between watershed KJE, where sediment loads decreased, versus watersheds BAN, CAR, DOL, EAG, and GIB. Watershed KJE was unburned and also had the smallest amount of unbuffered stream of all the cut units. Watersheds EAG and GIB were burned and had the greatest amount of unbuffered stream in burned areas. Watershed EAG experienced the largest sediment increases and also had the greatest proportion of road cut and fill area. Because EAG was not unusually high in road surface area, the large road cut and fill area indicates that the roads in EAG are on steeper hillslopes.

There is little field evidence of sediment delivery from roads in

Table 2*—Explanatory variables considered in modeling storm sediment loads in North Fork subwatersheds.*

Mean unit area suspended load from control watersheds
Excess storm flow volume relative to that of control watersheds
Time since logging completed
Timber removed per unit watershed area
Areas of various disturbances as proportion of watershed:
Cable, tractor yarding
Stream protection zones, thinned areas
Burning (low intensity, high intensity)
Road cuts, fills, running surfaces
Skid trail cuts, fills, operating surfaces
Landing cuts, fills, operating surfaces
Areas of above disturbances within 46 m (150 ft) of a stream channel
Length of impacted stream in above disturbances per unit watershed area
Length of cabled corridors per unit watershed area
Watershed area

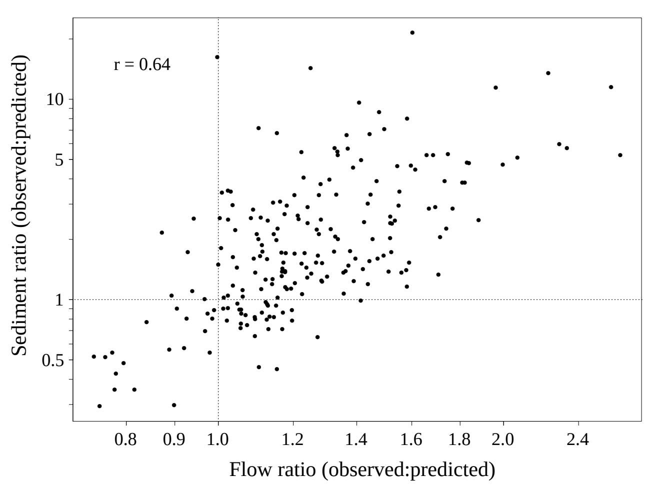

Figure 8—Relation between post-logging ratios of observed to predicted storm flow and suspended sediment load for all North Fork subwatersheds. Predictions are for undisturbed watersheds based on aggregated regression models using HI control (mean response of unlogged tributaries HEN and IVE).

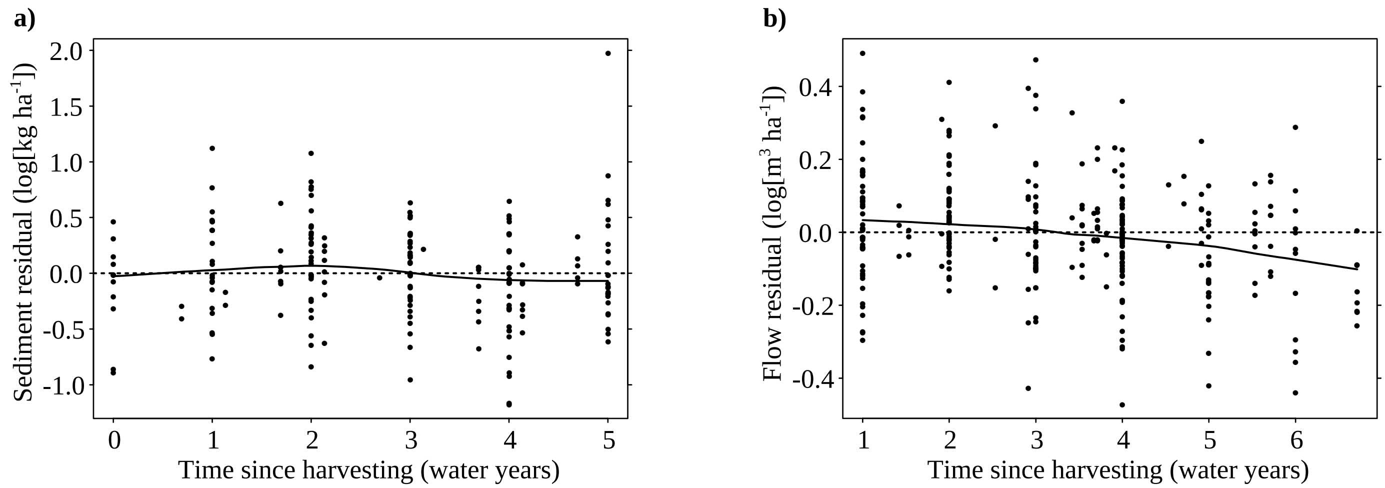

Figure 9—Relation between post-logging residuals from aggregated models and time (difference in water years) since harvesting. (a) model for storm suspended sediment loads, and (b) model for storm flow volumes. Curves were fitted by locally weighted regression (Cleveland 1993).

the North Fork watershed. In the inventory of failures greater than 7.6 m^3, only 8 of 96 failures, and 1,686 of 7,343 m^3 of erosion were related to roads. Nearly all of this road-related erosion was recorded as remaining on-site, and none of the road-related failures occurred in the EAG watershed. Based on the 129 random erosion plots (Rice 1996), the road erosion in EAG was 9.3 m^{3}ha^{-1}, compared to 34.5 m^{3}ha^{-1} for KJE and 16.6 m^{3}ha^{-1} for all roads in the North Fork. Thus it seems that the appearance of road cuts and fills in the model resulted from a spurious correlation.

On the other hand, channel reaches subjected to intense broadcast burns did show increased erosion from the loss of woody debris that stores sediment and enhances channel roughness (Keppeler, electronic communication). And increased flows, accompanied by soil disruption and burning in headwater swales, may have accelerated channel headward expansion, and soil pipe enlargements and collapses observed in watershed KJE (Ziemer 1992) and in EAG, DOL, and LAN.

Based on the 175 random erosion plots in harvest areas (Rice 1996), the average hillslope erosion rates in the burned watersheds EAG and GIB were 153 m^{3}ha^{-1} and 77 m^{3}ha^{-1}, respectively, the highest of all the watersheds. The average rate for the unburned clearcut watersheds BAN, CAR, and KJE was 37 m^{3}ha^{-1}. These figures include estimates of sheet erosion, which is difficult to measure and may be biased towards burned areas because it was easier to see the ground where the slash had been burned (Keppeler, verbal communication). About 72 percent of EAG and 82 percent of GIB were judged to be thoroughly or intensely burned, and the remainder was burned lightly or incompletely. It is unknown how much of this hillslope erosion was delivered to stream channels, but the proportion of watershed burned was not a useful explanatory variable for suspended sediment transport.

The failure inventory identified windthrow as another fairly important source of sediment. Of failures greater than 7.6 m^3, 68 percent were from windthrow. While these amounted to only 18 percent of the failure volume measured, 91 percent of them were within 15 m of a stream, and 49 percent were in or adjacent to a stream channel. Because of the proximity of windthrows to streams, sediment delivery from windthrow is expected to be disproportionate to the erosion volume. Windthrows are also important as contributors of woody debris to channels (Reid and Hilton, these proceedings), and play a key role in pool formation (Lisle and Napolitano, these proceedings). Because woody debris traps sediment in transport, it is unknown whether the net effect of windthrow on sediment transport was positive or negative.

Cumulative Effects

A full explanation of the rationale and methods of testing for cumulative watershed effects is beyond the scope of this paper, and final results on this topic will be reported by Lewis and others (1998). Preliminary results will simply be stated here.

I have considered three types of information that the aggregated model provides about the cumulative effects of logging activity on suspended sediment loads:

1. Were the effects of multiple disturbances additive in a given watershed?
2. Were downstream changes greater than would be expected from the proportion of area disturbed?
3. Were sediment loads in the lower watershed elevated to higher levels than in the tributaries?

The response being considered in all of these questions is the suspended sediment load per unit watershed area for a given storm event. Watershed area was used in the model to represent distance downstream.

The first question may be answered partly by looking at the forms of the storm flow and sediment models. Analyses of the residuals and covariance structures provide good evidence that the models are appropriate for the data, including the use of a logarithmic response variable. This implies a multiplicative effect for predictors that enter linearly and a power function for predictors that enter as logarithms. It turns out that the flow response to logged area is multiplicative, and the sediment response to flow increases is a power function. These effects, however, are *approximately* additive within the range of data observed for watersheds receiving flow from multiple cut units.

The second question was addressed by testing terms formed from the product of disturbance and watershed area. If the coefficient of this term were positive, it would imply that the effect of a given disturbance proportion increases with watershed size. A number of disturbance measures were considered, including road cut and fill area and length of unbuffered stream channels. None of the product terms were found to have coefficients significantly greater than zero, indicating that suspended load increases were not disproportionately large in larger watersheds. To the contrary, the sum of the observed sediment loads at the four main-stem stations were all within 25 percent of the sum of the loads predicted for undisturbed watersheds (*table 1*). Apparently, much of the sediment measured in the tributaries has been trapped behind woody debris or otherwise stored in the channels, so that much of it has not yet been measured downstream.

There is, however, one subwatershed where this second type of cumulative effect may be occurring. Watershed DOL, only 36 percent cut, includes the 100 percent cut watershed EAG, yet the sediment increases (269 percent at DOL versus 238 percent at EAG) have been similar. The increases in DOL seem to be related to channel conditions created in the historic logging (1900-1904) and, possibly, to increased flows from recent logging. At the turn of the century, the channel between the DOL and EAG gaging stations was used as a "corduroy road" for skidding logs by oxen. Greased logs were half-buried in the ground at intervals equal to the step length of the oxen (Napolitano 1996), and an abundance of sediment is stored behind them today (Keppeler, electronic communication). Energy available during high flows may be mobilizing sediment stored behind these logs. In the lower reach, the channel has a low width:depth ratio and is unable to dissipate energy by overflowing its banks. The high banks in this reach would be particularly vulnerable to increased peak flows, and have failed in a number of places in the years since EAG was logged.

The third question was addressed by testing watershed area as a linear term in the model. The coefficient of watershed area was positive ($p = 0.0023$), implying that the response, suspended sediment transport per unit watershed area, tends to increase downstream in the absence of disturbance. This tendency (with the exception of watershed KJE) is apparent in the pretreatment lines fit by least squares (*fig. 10a*), and could be reflecting the greater availability of fine sediment stored in these lower gradient channels. The relevance to cumulative effects is that downstream locations might reach water quality levels of concern with a smaller proportion of watershed disturbance than upstream locations.

To the extent that larger watersheds reflect average disturbance rates and therefore have smaller proportions of disturbance than the smallest disturbed watersheds upstream, one might expect sediment loads downstream to increase by less than those in the logged tributaries, reducing the overall variability among watersheds. In addition, as mentioned before, some of the sediment may be stored for several years before reaching the lower stations. That is what we observed in this study—the post-treatment regression lines (*fig. 10b*) were much more similar among watersheds than the pretreatment lines, and the main-stem stations no longer transported the highest sediment loads relative to watershed area.

Discussion

North Fork versus South Fork

My analysis of the South Fork logging data used a different model than was used by Rice and others (1979). However, the estimated increases in sediment loads were similar. For example, they reported suspended load increases of 1,403 kg $ha^{-1}yr^{-1}$ in the year after road construction and 3,254 kg $ha^{-1}yr^{-1}$ for the 5-yr period after logging. For the same periods, I estimated increases of 1,475 and 2,877 kg $ha^{-1}yr^{-1}$. Reversing the roles of the two watersheds for the later North Fork logging, the same analysis was unable to detect an effect. However, analysis of storm event loads from 1986 to 1996, using smaller subwatersheds within the North Fork as controls that had similar 19th-century logging histories as the whole North Fork, indicated that storm loads at NFC had increased by 188 kg $ha^{-1}yr^{-1}$. When comparing these figures, one should consider the differences between the water years 1972-1978 and 1990-1996, as well as differences in sampling methodologies that could have biased the estimated sediment loads. The mean annual unit area streamflow in the control (NFC) was 63 percent higher in 1972-1978 than that in the control (SFC) in 1990-1996. There is a surprisingly good relation between annual excess sediment load (departures from the pre-treatment regression) and water discharge in each of the studies (*fig. 11*). For equivalent flows, excess sediment loads in the South Fork analysis were six to seven times those in the North Fork analysis. It is probable that the sampling methods in the 1960's and 1970's resulted in overestimation of sediment loads in the South Fork analysis by a factor of 2 or 3. Therefore, comparisons between *relative* increases are more appropriate. Excess suspended load was 212 percent to 331 percent (depending on whether an adjustment is made for the 1974 North Fork landslide) after logging the South Fork, and 89 percent after logging the North Fork, suggesting that the effect of logging on suspended sediment load was 2.4 to 3.7 times greater in the South Fork than in the North Fork. These estimates approximately agree with estimates (Rice 1996) that both erosion and the sediment delivery ratio in the South Fork were about twice that in the North Fork.

Subwatersheds and KJE Anomaly

Analyses of the 10 treated subwatersheds in the North Fork drainage show suspended load increases at the gaging stations located

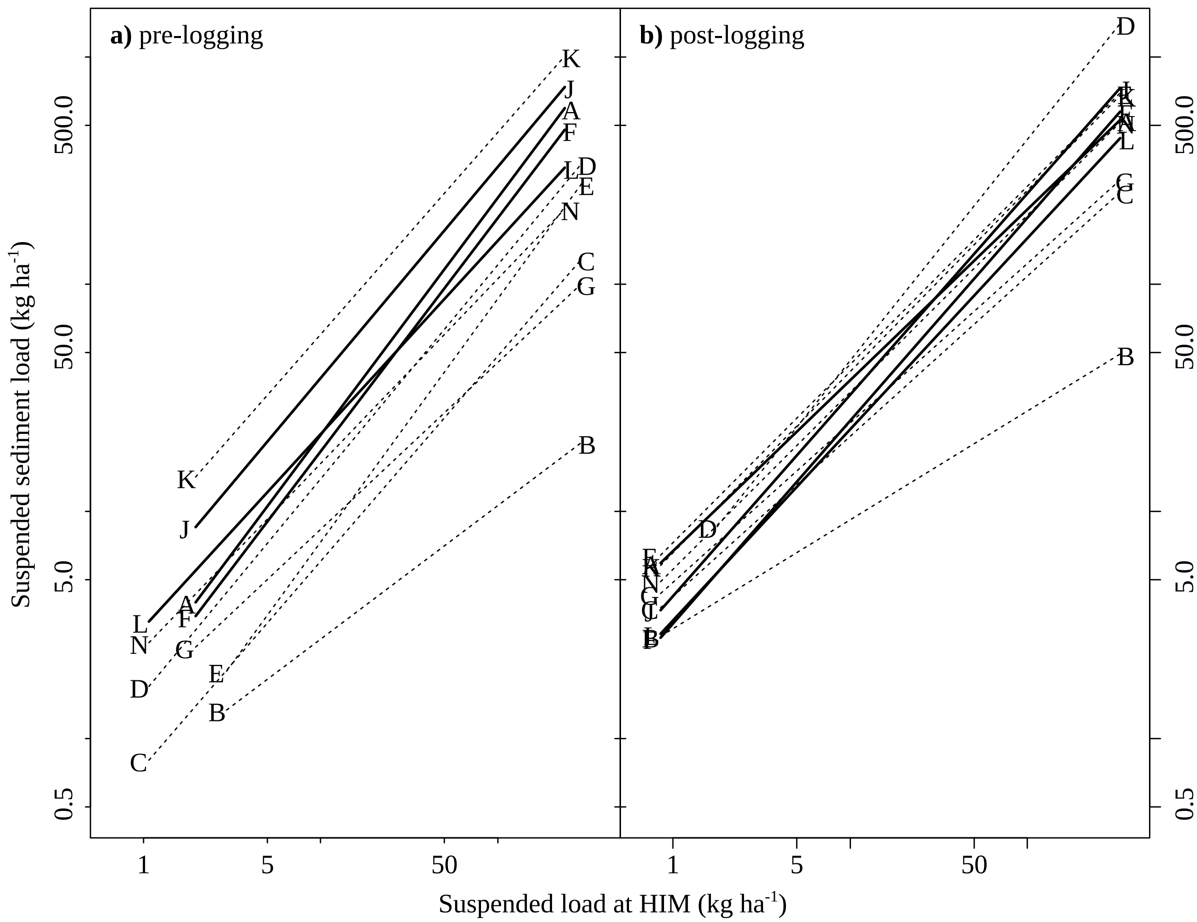

Figure 10—Regression lines for storm suspended sediment loads at treated watersheds in the North Fork, predicted from HIM control (mean suspended load of unlogged tributaries HEN, IVE, and MUN). (a) pre-logging, and (b) post-logging. Solid lines represent main-stem stations and dashed lines represent tributary stations.

immediately below clearcut units with one exception. At KJE, loads have decreased. A possible explanation for this anomaly lies in the tributary channel morphology. The stream channel in the KJE watershed is an extension of the main stem of the North Fork. It is (and, before recent logging, was) more deeply incised than the other tributaries, and it has the lowest gradient of tributaries other than the reach between the DOL and EAG gaging stations. The channel may have taken its gully-like form after the historic logging that took place between 1860 and 1904, when streams and streambeds were used as conduits for moving logs (Napolitano 1996). In any case, KJE had the highest pre-logging (1986-1989) unit area sediment loads of any of the tributaries (*fig. 10a*). Sediment in its channel is plentiful and the banks are actively eroding. It is likely, then, that the pre-logging sediment regime in KJE may have been energy-limited, which is more characteristic of disturbed watersheds. That is, sediment discharge was determined more by the ability of the stream to transport sediment than by the availability of sediment to be transported.

After logging, woody debris was added to the channel, and the number of organic steps in the buffered stream above KJE nearly doubled. Farther upstream, the channel was no longer shaded by the forest canopy and became choked with new redwood sprouts, horsetails, berry vines, and ferns, as well as slash that was introduced during logging. Although small storm flows did increase after logging, it is possible that channel roughness could have increased enough to reduce the energy available for sediment transport. An energy-limited stream would respond to increased sediment supply and reduced energy by reducing sediment transport. On the other hand, tributaries in a supply-limited sediment regime would have responded to a combination of increased sediment supply and reduced energy by increasing sediment transport. At some point, the increased supply probably converted these channels to an energy-limited regime, at which point stream power became the primary factor controlling variation in the increased transport levels. Rice and others (1979) concluded that is what happened after logging in the South Fork.

The aggregated regression for storm flow volumes (Lewis and others 1998; Ziemer, these proceedings) showed that flow increases

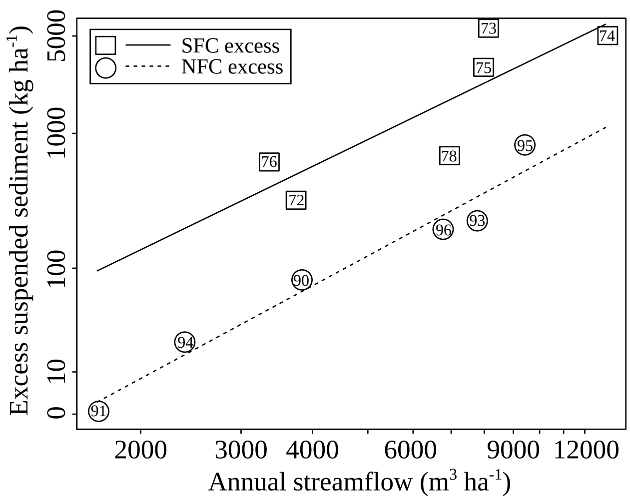

Figure 11—Relations between annual excess suspended sediment and annual streamflow for six years after logging in the South Fork and North Fork. South Fork excess loads are the departures from the pretreatment regression of figure 2. North Fork excess loads are the sums of storm departures from the pretreatment regression of figure 6.

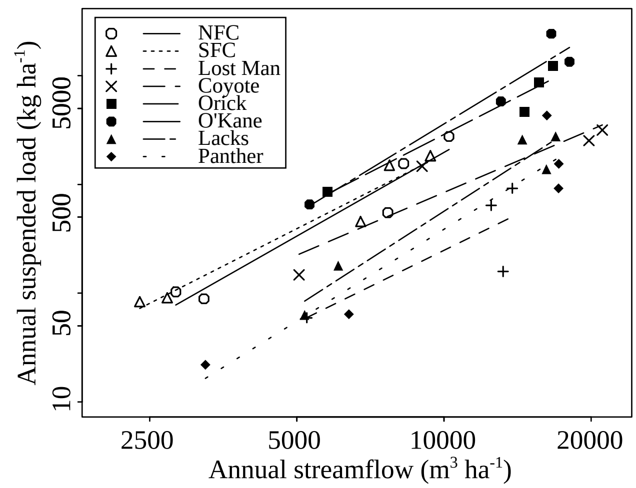

Figure 12—Relation between annual suspended sediment loads and annual streamflow for water years 1992-1996 at North Fork Caspar Creek (NFC), South Fork Caspar Creek (SFC), and 6 gaging stations in the vicinity of Redwood National Park. Caspar Creek sediment loads were divided by 0.6 to account for suspended sediment settling in the weir ponds.

could be largely explained by the proportion of a watershed logged, an antecedent wetness index, and time since logging. The aggregated regression for storm suspended sediment showed that much of the variability in suspended sediment load could, in turn, be explained by the flow increases. The implication is that, after logging, the channels were indeed in an energy-limited regime.

Flow increases accounted for only part of the variability in sediment production. Road systems would typically be expected to account for much of the sediment. However, in this case, roads were relatively unimportant as a sediment source because of their generally stable locations on upper hillslopes far from the stream channels. Field observations of increased bank erosion and gully expansion in clearcut headwater areas indicate that some of the suspended sediment increases were associated with the length of unbuffered stream channels in burned areas and, to a lesser degree, in unburned areas. Further indirect evidence that factors besides flow volume are elevating the suspended loads is that storm flows show a recovery trend, whereas storm suspended loads do not (*fig. 9*). This supports the hypothesis that the sediment regime has changed to one that will support elevated transport levels until the overall sediment supply is depleted. This can happen only after erosion and delivery rates to the channel decline and flows have been adequate to export excess sediment stored in the channels.

Cumulative Effects

Before logging, the larger main stem watersheds generally yielded the highest unit area sediment loads. But the increases after logging were greatest in the tributaries, resulting in a much narrower range of transport, for a given storm size, after logging (*fig. 10*). The North Fork of Caspar Creek is a small watershed (4.73 km^2). To see whether these results might be generalizable to larger watersheds, annual sediment loads for water years 1992-1996 were plotted against annual water yield (*fig. 12*) for NFC, SFC, and six gaging stations on streams in the vicinity of Redwood National Park (RNP). These watersheds were selected because of the high quality of their data and because, like Caspar Creek, they are underlain by the highly erodible Franciscan formation and historically supported mostly redwood forest with varying amounts of Douglas-fir. Caspar Creek receives less rainfall than the RNP watersheds, hence the lower annual flows.

In contrast to Caspar Creek, the RNP main-stem stations (Redwood Creek at Orick, 720 km^2, and at O'Kane, 175 km^2) continue to yield higher sediment loads than the RNP tributaries even after intensive management. Except for Little Lost Man Creek, these watersheds have been heavily logged at various times over the past 50 years, including the 1980's and 1990's. (Ground disturbance from logging in these watersheds was much more severe than that in Caspar Creek.)

The watershed with the lowest sediment loads is the unlogged Little Lost Man Creek (9.0 km^2), which is also the smallest of the RNP watersheds. Lacks Creek (44 km^2), Coyote Creek (20 km^2), and Panther Creek (16 km^2) are high-gradient (4-7 percent) channels in three different geologic subunits of the Franciscan formation (Harden and others 1982). Part of the explanation for the higher sediment loads at the main-stem stations may lie in the greater abundance of fine sediments available for transport in these low gradient (<1 percent) channels. Note that the Caspar Creek main stems are intermediate in both stream gradient (~1 percent) and sediment transport between the RNP tributaries and main stems. Regardless of the cause, if these lower reaches have the poorest water quality, then the incremental effect of an upstream disturbance may be cause for concern whether or not a water quality problem develops at the site of the disturbance. In other words, activities that have acceptable local consequences on water quality might have unacceptable consequences farther downstream when the preexisting water quality downstream is closer to harmful levels.

Cumulative effects considered in this paper were limited to a few hypotheses about water quality that could be statistically evaluated. But cumulative effects can occur in many ways. For example, resources at risk are often quite different in downstream areas, so an activity that has acceptable local impacts might have unacceptable offsite impacts if critical or sensitive habitat is found downstream. For a much broader treatment of cumulative effects see the discussion by Reid (these proceedings).

Conclusions

The main conclusions from these analyses are:

- Improved forest practices resulted in smaller increases in suspended load after logging the North Fork than after logging the South Fork. Increases were 2.4 to 3.7 times greater in the South Fork with roads located near the stream, all yarding by tractor, and streams not protected.
- Much of the increased sediment load in North Fork tributaries was related to increased storm flow volumes. With flow volumes recovering as the forest grows back, these increases are expected to be short-lived.
- Further sediment reductions in the North Fork probably could have been achieved by reducing or preventing disturbance to small drainage channels.
- Sediment loads are probably affected as much by channel conditions as by sediment delivery from hillslopes. The observed changes in sediment loads are consistent with conversion of those channels that were supply-limited before logging to an energy-limited regime after logging.
- The effects of multiple disturbances in a watershed were approximately additive.
- With one exception, downstream suspended load increases were no greater than would be expected from the proportion of area disturbed. To the contrary, most of the increased sediment produced in the tributaries was apparently stored in the main stem and has not yet been measured at the main-stem stations.
- Before logging, sediment loads on the main stem were higher than on most tributaries. This was no longer the case after logging. However, limited observations from larger watersheds suggest that downstream reaches in some watersheds are likely to approach water-quality levels of concern before upstream reaches.

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PROFILE

Livestock Grazing and Habitat for a Threatened Species: Land-Use Decisions Under Scientific Uncertainty in the White Mountains, California, USA

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ABSTRACT / The North Fork of Cottonwood Creek, in the White Mountains, Inyo National Forest, California, is a critically important refuge for the Paiute cutthroat trout (*Oncorhynchus clarki seleniris*), a federally listed threatened species. Habitat for these fish appears to be limited by excessive levels of fine sediment in the channel, and livestock grazing of riparian meadows has been implicated in delivery of sediment to the channel. However, the relationships between land use and sediment yield have not been conclusively determined, in large part because there are no historically ungrazed sites to serve as long-term controls. Accordingly, land-use decisions must be made under scientific uncertainty. To reduce erosion and sedimentation in the stream, the Forest Service spent approximately US$260,000 from 1981 to 1991 to repair watershed damage from livestock grazing, prevent livestock from traversing steep banks, and limit livestock access to the channel. Throughout this period, livestock grazing has continued on these lands, yielding less than $12,000 in grazing fees. In revising its Allotment Management Plan for the basin, the Forest Service rejected the "no-grazing" alternative because it was inconsistent with its Land and Resource Management Plan, which specifies there is to be no net reduction of grazing.

Livestock grazing occurs on 54 million ha administered by the US Forest Service, as well as 60 million ha administered by the Bureau of Land Management (BLM) and 82 million ha of private land, in the western United States. Livestock grazing is increasingly recognized as having a deleterious impact on riparian and aquatic habitat (Armour and others 1991). Grazing can increase runoff and erosion rates from uplands by reducing vegetative cover, compacting soil, and physically dislodging soil (Blackburn 1983). Probably more important than these changes in upland hydrology are the direct impacts of livestock on riparian zones, where livestock tend to concentrate because of availability of water, shade, and succulent foliage (Armour and others 1991, Platts and Wagstaff 1984). Vegetation may be removed from streambanks (and seedlings of woody species prevented from growing) by grazing, the weight of livestock may collapse overhanging banks, and hooves may chisel banks, directly introducing sediment into the channel. As summarized by Armour and others (1991, p. 7):

> Livestock grazing can affect the riparian environment by changing and reducing vegetation or by actual elimination of riparian areas by channel widening, channel aggradation, or lowering of the water table. The most apparent effects on fish habitat are the reduction of shade, cover, and terrestrial food supply, resultant increases in stream temperature, changes in water quality and stream morphology, and the addition of sediment through bank degradation and off-site soil erosion.

Grazing has been so widespread in western North America that it is difficult to locate ungrazed sites to serve as controls in studies of grazing impacts. This is a fundamental constraint on research in this field (Rinne 1988). Much of the existing evidence is derived from studies of livestock exclosures, fenced areas from which cattle are excluded. Exclosures have been used along many streams to encourage recovery of riparian vegetation and to enhance aquatic habitat (Platts and Wagstaff 1984). Studies comparing conditions within exclosures with those outside have documented increased riparian vegetation (Rickard and Cushing 1982, Kauffman and others 1983, Chaney and others 1990, Armour and others 1991), reduced area of eroding banks (Chaney and others 1990), and

KEY WORDS: Livestock grazing; Public lands; Cumulative watershed effects; Paiute cutthroat trout; Aquatic habitat

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improved trout production (Van Velson 1979). On the other hand, Keller and others (1979) found insignificant differences between trout populations within and outside exclosures. Despite these research results, site conditions are sufficiently variable and the processes of erosion and sediment delivery sufficiently complex, that to quantify the contribution of livestock grazing to a watershed's sediment yield and its impact on aquatic habitat conditions is difficult. Even with site-specific studies, some degree of uncertainty is likely to persist.

Livestock grazing is only one of the multiple uses permitted on US Forest Service lands. Other management objectives, such as maintenance of fish and wildlife habitat, water quality, and recreational opportunities, are to be balanced in making land-use decisions for specific areas. Grazing is managed in land units termed grazing allotments, and decisions about appropriate stocking levels are presented in Allotment Management Plans. Although all allotments are grazed, Allotment Management Plans had not been prepared for 27% of US Forest Service allotments, or for 66% of BLM allotments as of 1988 (GAO 1988). Thus, a great deal of grazing has been conducted without the most basic planning and evaluation by the administering federal agencies.

Many US Forest Service programs are subsidized in that their administrative costs exceed the revenue they produce. These include programs such as administration of wilderness areas and enhancement of fish and wildlife habitat, which most would agree yield a net long-term benefit to the quality of the environment. However, the magnitude of the subsidy for grazing on Forest Service lands, coupled with the adverse environmental impacts attributable to the practices, raise fundamental questions about the value system underlying land use decisions in the Forest Service. In fiscal year 1986, management of the Forest Service grazing program in the 16 western states cost approximately $24 million, while total receipts were $7.3 million (GAO 1988). These management costs do not include many expenditures to repair livestock damage to watersheds (such as the costs documented in this paper) and do not include continuing damage to natural resources, so the true subsidy is greater than these figures suggest.

Given the general inability to assess scientifically the contribution of livestock grazing to the degradation of aquatic habitat in any given watershed, the Forest Service is faced with making land-use management decisions under uncertainty. In this paper, we review the decision by the Inyo National Forest to permit continued grazing in the watershed of the North Fork Cottonwood Creek, habitat for a threatened species of trout, in light of available environmental information.

Status of the Paiute Cutthroat Trout

The Paiute cutthroat trout (*Oncorhynchus clarki seleniris,* formerly *Salmo clarkii seleniris*) constitutes a distinct subspecies of cutthroat trout, distinguished by its absence of body spots and coppery to purplish pink color (Vestal, 1947, Shapovalov and Dill 1950). The Paiute cutthroat trout was listed as endangered in 1970 (Federal Register 35, p. 16047), and reclassified to threatened in 1975 (Federal Register 40, pp. 29863–29864). The Paiute cutthroat is native to Silver King Creek, a tributary to the East Fork Carson River, in Alpine County, California. The subspecies evolved in isolation from other fish species since desiccation of Pleistocene Lake Lahontan, becoming distinct from the Lahontan cutthroat trout (*O. clarki henshawi*) only about 5000–8000 years before present (Behnke and Zarn 1976).

Paiute cutthroat populations are susceptible to displacement by introduced salmonids as a result of competition and introgressive hybridization (USFWS 1985). By the mid-1950s, the population of Paiute cutthroat in the Silver King Creek drainage had become introgressed with introduced Lahontan cutthroat trout and rainbow trout (*Oncorhynchus mykiss*). Before this occurred, pure strains of Paiute cutthroat had been transplanted to several other drainages in eastern California. Most of these transplants ended in failure, but the transplant into the formerly fishless North Fork Cottonwood Creek (North Fork) in the White Mountains east of Bishop in 1946 (Vestal 1947) proved to be a success (Figure 1). Two barriers to upstream fish migration protect the Paiute cutthroat population residing in the upper 5.6 km of the North Fork from downstream populations of rainbow trout and brook trout (*Salvelinus fontinalis*). However, in 1965, hybridized Paiute cutthroat were discovered above the most downstream barrier in the North Fork. A habitat management plan was prepared (Schneegas 1967), and the lower reach of the stream was chemically treated in 1970. Subsequent discovery of hybridization below the uppermost barrier necessitated repeat treatments in 1981 and 1982; a genetically pure population remained above the treatment area (Wong 1991).

Wong (1975) estimated the population in the North Fork at 500 in 1973, based on electroshocking surveys and prolonged visual observations. Following chemical treatment in 1981 and 1982, pure-strain

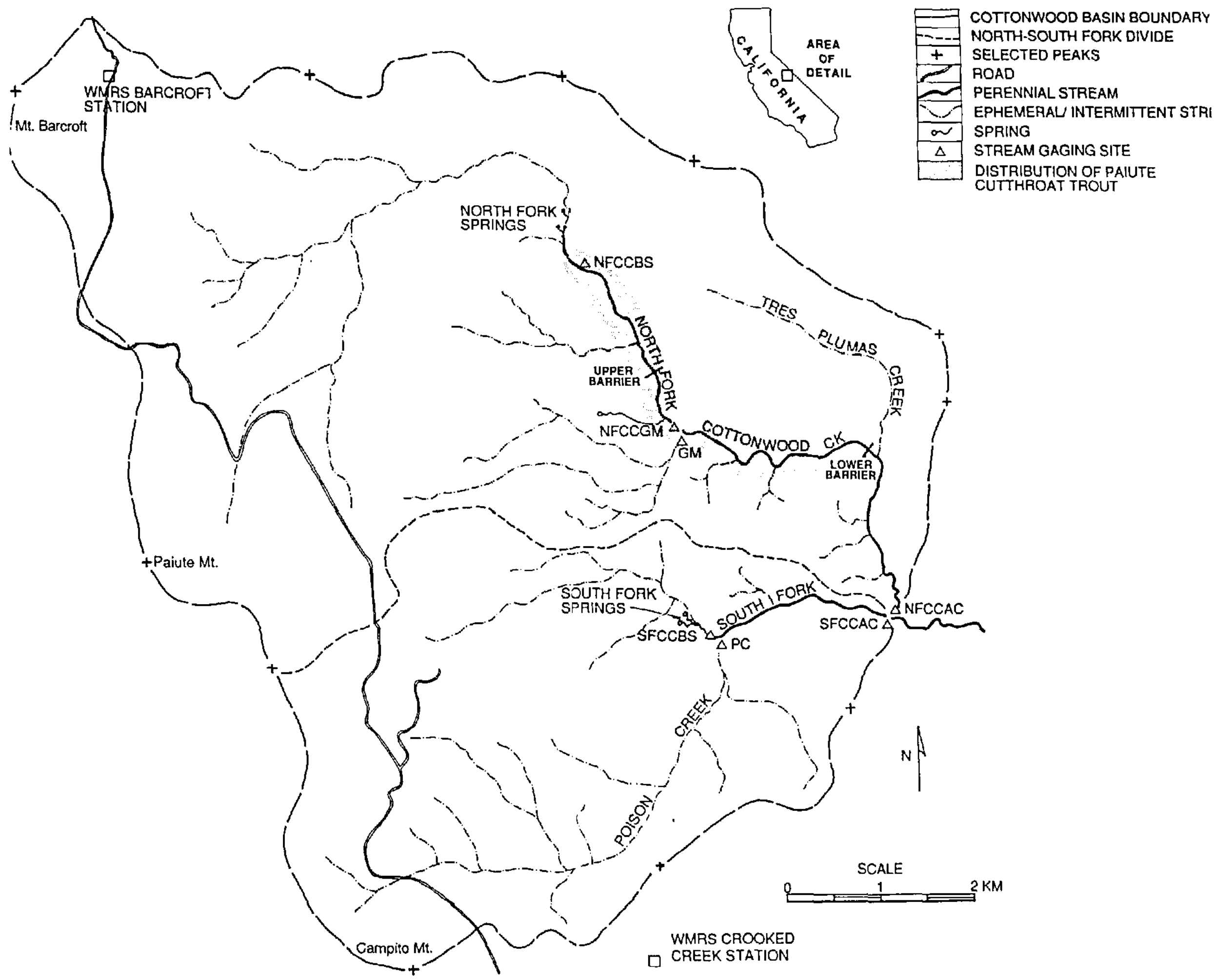

Figure 1. Map of Cottonwood Basin, showing basin divides, locations of White Mountain Research Station (WMRS) facilities, stream gauging stations, and the range of Paiute cutthroat trout.

Paiute cutthroat were transplanted from upstream reaches into treated downstream reaches from 1983 to 1987. Subsequent fish population surveys indicated the population increased annually to approximately 1260 adults and juveniles in 1988 and subsequently declined to about 670 in 1991, probably largely in response to reduced flows during a prolonged drought. Fish population estimates after 1973 are subject to uncertainty because they were based on visual estimates; electroshock surveys were avoided due to their potential impact on a small population (D. Wong, California Department of Fish and Game, Bishop, personal communication, 1991). Excessive levels of fine sediment and a paucity of suitable spawning gravels in the North Fork have been cited as principal factors limiting habitat for Paiute cutthroat by Wong (1975), Diana (1975), USFWS (1985), and USFS (1988a). USFWS (1985) cited livestock grazing as a factor contributing to instream sedimentation and argued for modification of land-use practices in the basin.

Study Area

Cottonwood Basin is a topographically distinct basin, defined here as the drainage basin of the North and South Forks of Cottonwood Creek (the North Fork and South Fork) above their confluence (Figure 1). Overall, Cottonwood Basin ranges in elevation from numerous peaks over 3800 m along the range crest to 2760 m at the confluence of the forks. The channel is flanked by stringer meadows extending from the confluence up to 3260 m along the North

Figure 2. Stringer meadow along North Fork Cottonwood Creek. View downstream towards South Fork confluence. (Photograph by author, September 1991.)

Fork and to 2880 m along the South Fork (Figure 2). The meadows are dominated by annual grasses, sedges (*Carex* sp.), thistle (*Cirsium drummondii*), sagebrush (*Artemisia ludoviciana*), paintbrush (*Castilleja miniata*), and several species of beardtongue (*Penstemon*), monkey flower (*Mimilus*), and evening primrose (*Oenothera*) (Wong 1975). Woody riparian vegetation is dominated by willow (*Salix* sp.) and aspen (*Populus tremuloides*).

Cottonwood Basin lies entirely within the White Mountain Ranger District of the Inyo National Forest and is a popular destination for hikers and fishermen because of its spectacular granitic outcrops, meadows, fishing for introduced rainbow and brook trout in the South Fork, and access by a four-wheel drive road. The meadows along perennial reaches of the streams are grazed by a permittee as part of the Cottonwood–Tres Plumas Allotment. Typically, about 200 cattle graze the meadows for a period of one to two months in the summer as they migrate through the entire allotment (Bob Suter, Inyo National Forest, personal communication, 1991).

Because of the importance of the aquatic resource in the basin, the Inyo National Forest implemented an extensive program of watershed restoration activities designed to reduce erosion and sediment delivery to the channel, funded in large part by funds allocated by the California Department of Fish and Game Wildlife Conservation Board. From 1981 to 1991, Forest Service work crews built about 500 erosion control structures, rerouted 500 m of livestock trails to avoid sensitive areas, and barricaded over 4 km of streambank from cattle access. The erosion control structures included emplacing rock to protect gully headcuts, installation of redwood and rock check dams, construction of supports for hillside trails, construction of waterbars along trails, and emplacing brush in gullies. Also over this period, 164 artificial gravel pockets were placed in the streambed to enhance spawning habitat (Figure 3). The gravel pockets were usually placed at sites where previous spawning attempts had been observed or where spawning was judged likely based on previous observations (D. Wong, California Department of Fish and Game, Bishop, personal communication, 1991). Construction of the artificial gravel pockets required the importation (by truck, mule, and back-pack) of 13.4 m^3 of suitable gravel (Mark Dawson and Jack Anderson, Inyo National Forest, personal communication, 1991).

Ongoing Field Studies in Cottonwood Basin

Methods

Because grazing has been nearly ubiquitous in meadows of the White Mountains for decades, no "control" meadows exist and thus it is impossible to design a controlled experiment to determine the long-term impacts of grazing on basic hydrology and sediment yield. However, geomorphic studies were initiated in the Cottonwood Basin in 1989 to quantify short-term sediment delivery to the channel to the extent possible. The 1987–1992 drought provided an opportunity to document the effects of drought on

Figure 3. Pocket of imported gravel in North Fork Cottonwood Creek. Cobbles are arrayed downstream to stabilize the gravel deposit. Flow from left to right. (Photograph by author, June 1989.)

baseflow (Kondolf 1992), but the low runoff during the study period limited opportunities to quantify sediment transport. To monitor sediment yield from tributary channels and gullies, the channels immediately above check dams (installed as part of erosion control efforts) were surveyed in 1990 to provide a baseline against which future accumulation of sediment can be measured. To monitor sediment mobility and bed conditions within the channels of the North and South Forks, cross sections were surveyed in 1990 and 1991. In three cross sections tracer gravels (made by painting gravels removed from the streambed) were used to document downstream particle movement. Fine sediment infiltration was measured using plastic mesh boxes filled with clean gravel emplaced in the bed within nine pockets of imported gravel and in one pocket of natural gravels. By the time this study was initiated, most of the 164 pockets of imported gravel had already been constructed, so it was not possible to systematically compare spawning gravel availability before and after the gravel enhancement projects of 1986–1990. However, to provide some indication of gravel quality, I sampled two pockets of imported gravel and one native gravel pocket using a cylindrical core sampler, and analyzed the samples for particle size.

From files of the Inyo National Forest White Mountain District and the Department of Fish and Game Wildlife Conservation Board in Sacramento, I compiled costs of watershed restoration and gravel enhancement projects in Cottonwood Basin, numbers of structures emplaced, and total amounts of gravel emplaced. For some years, total amounts budgeted and spent for the work were available, in other cases we computed approximate total costs by multiplying the number of various structures emplaced by their average unit cost. I computed total amounts received in grazing fees over the period 1981–1991 by multiplying head of cattle by the months in the entire allotment and current fee per animal unit month. An "animal unit" is a cow and calf; an animal unit month is one month of grazing by an animal unit. Actual usage was commonly less than permitted, and the permittee received credits for the difference, so the actual receipts were less than computed here (Bob Suter, Inyo National Forest, personal communication, 1991). I also reviewed existing land management policies in light of the data collected for this study and the objectives of the Paiute Cutthroat Trout Recovery Plan (USFWS 1985).

In-Channel Sediment Conditions

High flows in North Fork Cottonwood Creek are evidently rare. None occurred in 1973 (Wong 1975); a flow of 0.051 m^3/sec was observed in 1974 in response to a thunderstorm (Diana 1975); none occurred in 1990 or 1992, and a bankfull flow of 0.142 m^3/sec was generated by snowmelt in 1991. The 1991 flow produced partial scour of imported gravels and tracer gravels at all but one cross section. For example, at cross section XSNF-9, 44 tracer particles (22–32 mm in diameter) emplaced within the pocket of imported gravel were transported an average distance of 1.7m, and the bed degraded about 0.1 m at the redd site

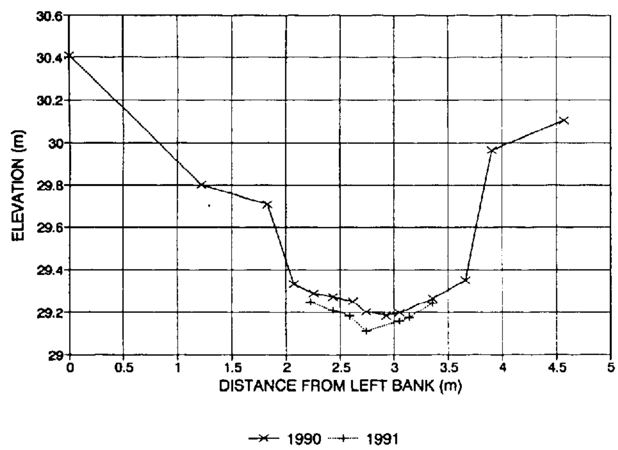

Figure 4. Cross section (XSNF-9) across imported gravel deposit on North Fork Cottonwood Creek about 0.1 km downstream of Granite Meadows, showing bed degradation over the 1990–1991 season.

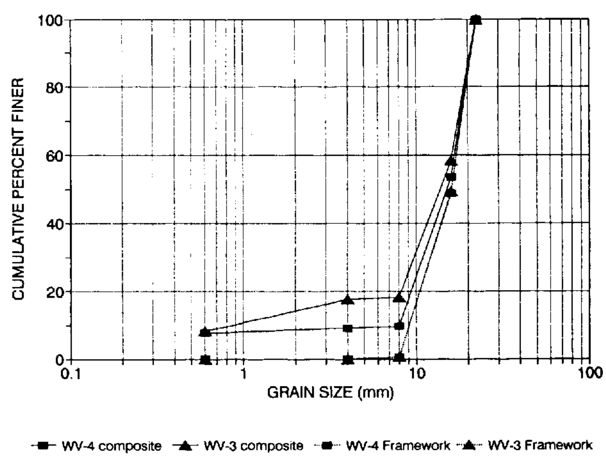

Figure 5. Cumulative size distribution curves for gravels in plastic mesh boxes prior to (dashed line) and after (solid line) infiltration.

(Figure 4). Of ten plastic mesh boxes of gravel placed in the bed of the North Fork, eight were scoured in place or transported downstream. The two boxes that remained stable displayed substantial intrusion of fine sediments from August 1990 to August 1991, as depicted in the cumulative size distribution curves for the experimental gravel shown in Figure 5. The percentage of sediment finer than 4 mm increased from 0% to 18% (WV-3) and 9% (WV-4) due to infiltration. However, even with the increased proportion of fine sediments, these gravels would still be suitable for spawning based on standards drawn from published studies (Kondolf 1988).

The two pockets of imported gravels sampled had median diameters of 12.5 mm and 24 mm, sizes well suited for spawning by small trout (Kondolf and Wolman 1993). The median diameter of the native gravel pocket sampled was 4.6 mm, somewhat smaller than typically encountered in spawning gravels. Its small median was largely influenced by its large proportion of sediment finer than 4 mm (45%), an excessively high fine sediment content. Percentage finer than 4 mm in the pockets of imported gravel (0.2 and 15%) was well within acceptable levels of fine sediment based on criteria drawn from published studies (Kondolf 1988).

Redd surveys indicate heavy usage of pockets of imported gravel. For example, in 1988, from the Lower Barrier to Granite Meadow (Figure 1), 28 redds were observed in artificial gravel sites and 25 redds in native deposits (unpublished data in files of California Department of Fish and Game, Bishop). Over the same reach in 1990, 17 redds were observed in pockets of imported gravel, seven in native deposits. The better quality of the imported gravels and their heavy usage by spawning fish suggest that the pockets of imported gravel constitute good spawning habitat. However, I could not quantify the improvement in spawning habitat resulting from the enhancement program because most native gravel pockets had already been augmented with imported gravels by 1989. Moreover, the extensive scour produced by the bankfull flow in 1991 indicates that the constructed pockets are transient features and that the stream will readily redistribute the gravels during its infrequent high flows.

Sources of Fine Sediment

Because no ungrazed meadows exist as controls and without major runoff in the basin during the study period, I could not directly measure the contribution of livestock grazing to sediment yield to the channel. However, areas of bare ground within the meadows in the study area were almost invariably associated with cattle trails or stream crossings. Within a 5-ha exclosure established in 1967, tall grasses and sedges grow along the streambanks, probably providing cover for fish and serving to filter sediment carried to the channel by overland flow. Outside the exclosure at the end of the growing season, banks have been closely cropped and trampled by livestock. However, a survey of channel cross sections within and outside the exclosure showed no significant differences in channel width, suggesting that changes in channel morphology lag behind vegetative recovery within the exclosure. Moreover, the channel within the exclosure is still subject to influence from hydrologic impacts of grazing upstream.

Available sediment-related data in Cottonwood Basin are inconclusive regarding effects of livestock grazing. While high rates of fine sediment infiltration within gravels suggest sediment delivery rates to the channel may be impacting aquatic habitat, it is not possible to state what percentage of this sediment results from livestock grazing.

Watershed Restoration Costs and Grazing Receipts

Watershed repair and gravel enhancement projects involved over 200 days by a six-person work crew, at a total cost of about $260,000 from 1981 to 1991. This value does not include costs of administering the grazing programs, costs which alone exceed receipts in many national forests. Total grazing receipts for the Cottonwood–Tres Plumas Allotment from 1981 to 1991 were $11,657, as computed from permitted number of head, grazing period, and the current USFS grazing fee. Because permittees have received refunds for shorter grazing periods or fewer head than permitted, actual USFS receipts were less.

Forest Service Land Management Policies

The principal document providing planning direction for the Inyo National Forest is its Land and Resource Management Plan (USFS 1988b), which sets policy for various land uses in the Forest, including forestwide levels of range production. In addition, management of the North Fork Cottonwood Creek and tributary lands is guided by the Paiute Cutthroat Trout Habitat Management Plan (USFS 1988a), prepared pursuant to Section 7 of the Endangered Species Act of 1973 as amended in 1983. The habitat management plan summarizes guidance from the Forest Service Manual that, "The conservation of endangered and threatened species will receive priority in management activities. Habitats of listed species will be protected from adverse modification or destruction." (USFS 1988a, p. 4). The habitat management plan calls for the Forest Service to monitor effects of grazing and other land-use activities on Paiute cutthroat trout habitat and to develop semiquantitative habitat suitability and monitoring criteria. However, it does not specifically call for cessation of land-use activities if monitoring indicates they have a detrimental effect upon the species' habitat. This is in contrast to the US Fish and Wildlife's Paiute Cutthroat Trout Recovery Plan, which calls for modification of "land use practices . . . in the . . . Cottonwood Creek drainage[ ] to reduce streambed sedimentation and promote growth of riparian vegetation to provide the cover and food producing areas necessary for the fish's survival" (USFWS 1985). The habitat management plan calls for construction of erosion control structures, importation of gravel to the channel, and eventual expansion of the species range into the South Fork and mainstem Cottonwood Creek, but does not indicate how the success of these efforts is to be evaluated (USFS 1988a).

The Inyo National Forest is currently preparing an allotment management plan for the Cottonwood–Tres Plumas grazing allotment, which includes the North Fork Cottonwood Creek. In the draft allotment plan, the "no-grazing" alternative was presented but not discussed as a viable alternative because it is in conflict with direction provided by the land and resource management plan, which emphasizes maintaining current forestwide levels of range production of 41,000 animal unit months. There is to be no net reduction in animal unit months resulting from a change in any given allotment (USFS 1988b). Thus, the land and resource management plan is interpreted to preclude any net reduction in grazing, even if monitoring indicated that grazing was negatively impacting habitat quality. Although the Paiute Cutthroat Trout Recovery Plan (USFWS 1985) calls for changes in land use, the habitat management plan (USFS 1988a) does not discuss reduction in grazing as a management option, implicitly assuming that watershed restoration projects can mitigate impacts of grazing, despite a lack of supporting data.

Land-Use Decisions under Scientific Uncertainty

The Forest Service has made land management decisions without data quantifying relations between land use and aquatic habitat quality applicable to the watersheds in question. Given the scientific uncertainty, these land-use decisions will inevitably reflect values and management bias. The present policies are biased towards grazing. This bias is evident in land-use decisions for riparian areas and tributary lands of the North Fork Cottonwood Creek, where grazing has been permitted to continue, despite a stated policy to give priority to habitat for Paiute cutthroat trout, and despite recognition by the US Fish and Wildlife Service that grazing is a factor in degradation of habitat in the North Fork Cottonwood Creek (USFWS

1985), and the implicit recognition of this link by the US Forest Service in the extensive erosion control works and livestock trail rerouting undertaken in Cottonwood Basin.

The Forest Service spent $260,000 on erosion control and watershed enhancement in Cottonwood Basin from 1981 to 1991. During this period, less than $12,000 was received in grazing fees. In effect, the private lessee is receiving a greater than 20:1 subsidy from the government to graze the area. Many government programs subsidize exploitative land uses for perceived social benefits such as employment, so the unfavorable economics of these grazing leases alone cannot condemn the practice. However, the risk posed to survival of a threatened species should make any environmental degradation less acceptable in this watershed. Moreover, despite this massive public expenditure, no convincing data are available to indicate that the effects of grazing upon habitat for the Paiute cutthroat trout have been fully mitigated by the watershed restoration works.

An alternative policy would be to orient land management towards preservation and enhancement of aquatic habitat for the Paiute cutthroat trout, consistent with the "priority in management activities" afforded conservation of a threatened species (USFS 1988a). Under such a policy, livestock grazing would be permitted only if it could be demonstrated that grazing would have no significant deleterious influence on aquatic habitat. Under this approach, the scientific uncertainty would remain, but the burden of proof would be shifted such that research findings would be required not to restrict grazing but to permit it. Given the unfavorable economics of these grazing leases, the most sensible course could well be to eliminate or severely reduce livestock grazing in the Cottonwood Basin.

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Rangeland Grazing as a Source of Steroid Hormones to Surface Waters

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Cattle and other livestock excrete endogenous steroid hormones, including estrogens, androgens, and progestins; therefore, allowing grazing livestock direct access to surface waters can result in the release of steroids in agricultural watersheds. Elevated concentrations of steroids are problematic because low concentrations of certain steroids can affect fish reproduction. To assess the occurrence and transport of steroids arising from grazing cattle, gas chromatography–tandem mass spectrometry (GC/MS/MS) was used to quantify a suite of estrogens, androgens, and progestins in small creeks impacted by rangeland grazing. Steroids were detected in 86% of samples from rangeland creeks where cattle had direct access to the water, with concentrations as high as 44 ng/L observed shortly after rain events at the beginning of the winter wet season. Estrogens were present at concentrations above the predicted no-effect concentrations for fish in 10–20% of the samples, and androstenedione was detected at concentrations higher than response thresholds for pheromonal communication in fish. The results suggest that, in certain cases, measures such as stream fencing in rangeland areas to limit direct discharge of animal wastes to surface waters or better manure management practices might be merited to protect ecosystem health.

Introduction

Research conducted over the last two decades has shown that the reproduction and development of fish are affected by low concentrations of dissolved steroid hormones. The attention of the scientific community first focused on this phenomenon after reports implicated estrogenic steroids in municipal wastewater effluent as the cause of feminization of fish (*1–3*). More recent studies have documented the masculinization of fish after exposure to androgens in pulp and paper mill effluent (*4*, *5*) and runoff from confined animal feeding operations (*6*). Although exposure of fish to progestins has not been shown to cause observable morphological changes; progestins, along with certain androgens, are important pheromones in many fish species (*7–9*). Because fish biochemistry and behavior are altered by pheromonal steroids at extremely low concentrations (*7–9*), pheromonal communication and reproductive behavior may be susceptible to disruption near anthropogenic sources of steroids (*10*).

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Relative to point sources, diffuse sources of steroids related to agriculture, such as rangeland grazing or runoff from manure-treated fields, have received much less attention despite the large areas of land used for rangeland grazing. For example, rangeland grazing is the predominant land use for over 32% of the land in the United States (*11*), and in many rangeland areas, cattle are allowed direct access to ponds, rivers, and creeks for drinking water. Although the potential impact of allowing livestock to access surface waters has been recognized as part of a strategy to control waterborne pathogens and stream erosion, the potential for this practice to contribute steroids to surface waters has received only limited attention (*12*, *13*). Given that cattle excrete 0.045–105 mg/day of steroids per animal (*14*, *15*), the daily steroid excretion from a single animal can potentially elevate steroid concentrations by 1 ng/L in as much as 105,000 m^3 of uncontaminated water.

Most studies of steroid occurrence in watersheds with animal agriculture have focused on estrogenic steroids derived from land application of animal manures or discharges from waste storage lagoons (*16–19*). However, livestock excrete androgens and progestins at rates comparable to or greater than rates for estrogens (*14*), and the available data indicate that these steroids also occur in watersheds with animal agriculture (*20–22*). Recent studies examining receiving waters in grazing rangelands suggest that estrogen contamination may be widespread, with direct discharges of animal wastes to receiving waters serving as a significant source of contamination (*12*, *13*). Additional research is needed to assess the temporal and spatial variability of steroid occurrence and the role of storms in mobilizing steroids in watersheds with grazing livestock.

To investigate the contribution of rangeland grazing areas to steroids in surface waters, samples were collected and analyzed from sites representative of cattle-grazing rangelands in the western United States. Concentrations of steroids detected during wet and dry seasons were compared with threshold values for biological effects to assess the potential impacts of grazing on sensitive fish species. To evaluate the effects of livestock density and rangeland management practices on water quality, the loading of steroids to surface waters was estimated from measured concentrations and estimates of stream discharge.

Materials and Methods

Materials. All reagents were purchased from Fisher Scientific (Pittburgh, PA) at the highest possible purity. Steroids, also at the highest possible purity, and heptafluorobutyric anhydride (purity >98%), were purchased from Sigma-Aldrich (St. Louis, MO). Deuterated steroids, which were used as surrogate standards, were purchased from CDN Isotopes (Quebec, Canada). Reverse osmosis water was produced using a Nanopure II system (Barnstead, Dubuque, IA).

Sample Sites. Between April 2005 and March 2006, 30 sites were sampled in Stanislaus, Marin, and Sonoma counties in central California (Figure 1). The sampling locations and frequencies were chosen to evaluate the effects of precipitation, streamflow, density of animals, and creek accessibility on concentrations of steroids. These locations are representative of the small headwater creeks (discharge 0.01–9.7 m^3/s, average width 1–12 m) that characterize many watersheds where cattle grazing is the predominant land use. This area of California has a Mediterranean climate, with a dry season typically lasting from April to October. Approximately 60% of the creeks sampled were ephemeral and only contained water between October and July. Samples

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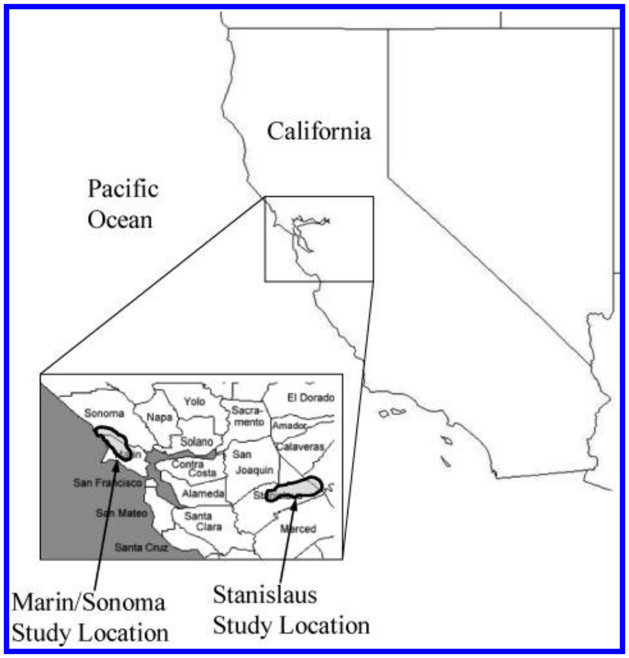

FIGURE 1. Grazing rangeland study locations in California.

were collected at all locations with flowing water during both dry and wet seasons.

The Stanislaus county sites were all located in the Dry Creek watershed in California's Central Valley. This watershed is approximately 540 km^2 and contains over 390 km of tributaries. The maximum elevation of the watershed is 384 m and the average annual rainfall is approximately 32 cm. The basin is primarily low−middle relief, characterized by grasslands, oak woodland, and oak chaparral vegetation types. Although the number of cattle in the Dry Creek watershed is unknown, 284,000 cattle were present in Stanislaus county (2004 data), and the Dry Creek watershed covers 14% of the land area in Stanislaus county. Cattle were present throughout the watershed with estimated densities between 0 and 20 per hectare, with higher densities observed sporadically. Samples were collected from the Stanislaus county sites on April 22, May 16, August 31, and December 29, 2005, and on March 9, 2006.

The Marin/Sonoma county sites were located in five contiguous coastal watersheds with a mixture of rangeland grazing, dairy farms, and crop agriculture land uses. The watersheds encompassing this study area are approximately 845 km^2 in area, with at least 310 km of tributaries upstream of the sampling locations. Relative to the Stanislaus county sites, the Marin/Sonoma county sites are wetter (average rainfall approximately 63 cm/yr), hillier (maximum elevation 601 m), and more densely vegetated with coastal grasslands, oak woodlands, and mixed deciduous−conifer forest types. There were 35,500 cattle in Marin county (2145 km^2) and 80,000 cattle in Sonoma county (4580 km^2) in 2004, and cattle grazing densities were similar to those observed in the Stanislaus sites. Samples were collected from the Marin/Sonoma county sites on June 23, September 8, November 8, and December 2, 2005.

Chemical Analysis. All samples were collected in 12-L fluorinated Nalgene (Rochester, NY) containers. After pressure filtration, the steroids were extracted using 90 mm C-18 solid-phase extraction discs followed by derivatization and gas chromatography−tandem mass spectrometry as described previously (Supporting Information, Table S1, *10*, *20*). The method was modified to include 17α-estradiol, which eluted 1.2 min prior to 17β-estradiol. The daughter ions used for quantification of the 17α-estradiol are identical to those used for 17β-estradiol. Also, d_5-17β-estradiol and d_4-estrone were used as surrogate standards in addition to mesterolone.

Quality assurance and quality control consisted of one reverse osmosis water blank, one duplicate sample, and one matrix recovery sample amended with a mixture of all steroid analytes at concentrations of 0.5−5.5 ng/L per sample event. No steroids were observed in the blank samples, and the coefficient of variation for the duplicate samples was less than 15%. Recovery of the matrix spikes typically ranged from 72 to 160%, with the exception of medroxyprogesterone, where one spike exhibited an unexpectedly high recovery of 261%. In general, the highest matrix spike recoveries were observed for steroids that did not have a deuterated surrogate. For the two analytes that had deuterated surrogates (i.e., 17β-estradiol and estrone), the spike recoveries ranged from 72 to 106%. Due to a problem with the solvent wash step used for cleanup of organic matter, the recovery for estriol, the most polar of the analytes, was unexpectedly low; therefore, estriol was not reported. Method detection limits ranged from 0.1 to 0.2 ng/L, and were analyte dependent (Table S1).

To evaluate the occurrence of steroids in cattle manure leachate, on August 31, November 8, and December 2, 2005, 500 g (wet weight) of fresh cattle manure was collected in the field and placed in a container with 10 L of Nanopure water at 4 °C. After 1 day, 1 L of this water was sampled, filtered, and analyzed for steroid hormones. A second sample from this container was collected and analyzed 1 week later.

Streamflow was measured on March 9 (Stanislaus county sites) and March 20, 2006 (Marin/Sonoma county sites) using a Flowmate model 2000 (Marsh-McBirney, Frederick, MD) electronic flowmeter. Stream discharge was estimated for all sample sites and dates by adjusting the measured stream discharges to archived data from the Dry Creek gauging station (DCM station; CA Dept. of Water Resources) for the Stanislaus county sites and the Lagunitas Creek station (station 11460400; U.S. Geological Survey) for the Marin/Sonoma county sites.

Results

All of the steroid analytes were detected in one or more of the 88 water samples (see Table S2 for complete data). Estrone was detected more frequently than the other steroids, with detectable concentrations in 78% of the samples at concentrations as high as 38 ng/L (Table 1). The estrogen 17α-estradiol was present in 31% of the samples at concentrations up to 25 ng/L, while 17β-estradiol was present in 18% of the samples at concentrations up to 1.7 ng/L. The androgens testosterone and androstenedione were detected less frequently than the estrogens, with detectable concentrations in 11% and 18% of the samples, respectively. Testosterone concentrations never exceeded 2.3 ng/L, whereas androstenedione was detected at concentrations up to 44 ng/L. Progesterone was present in 5% of samples; however, when detected, the concentrations of progesterone were generally higher than those of the other steroids. In three of the four samples in which progesterone was detected, the occurrence of progesterone coincided with the detection of androstenedione. Medroxyprogesterone was only detected in one sample near an urbanized area at a concentration below the limit of quantification.

The maximum concentrations detected for each of the steroids occurred during the wet season (November−March), with the exception of medroxyprogesterone, which was only detected in May. The highest concentrations of 17α-estradiol, estrone, and progesterone occurred in the Marin/Sonoma county watersheds immediately after the first major storm (i.e., >2 cm of precipitation) of the wet season, which occurred on November 6−7, 2005 (Figure 2). This storm was the first storm with enough rainfall for the ephemeral creeks to begin

TABLE 1. Occurrence and Maximum Concentration of Steroid Hormones in Surface Waters on Grazing Rangelands, near Dairy Farms, and in Municipal Wastewater Effluent

receiving water	17α-estradiol	17β-estradiol	estrone	testosterone	androstenedione	medroxy-progesterone	progesterone
Grazing Rangeland (Surface Water)							
percent occurrence [%]; $N = 89$	31	18	78	11	18	1	5
max. concentration [ng/L]	25	1.7	38	4.3	44	0.4	27
Dairy Farms (Surface Water)[a]							
percent occurrence [%]; $N = 32$	NA[c]	6	38	25	0	12	0
max. concentration [ng/L]	NA	0.7	17	1.9	< 0.3	1.0	< 0.4
Municipal WWTP (2°, 3° Effluent)[b]							
percent occurrence [%]; $N = 95-144$	NA	32	37	21	11	14	NA
max. concentration [ng/L]	NA	7.8	19.5	8.0	4.5	14.9	NA

[a] Data from ref *10*. [b] Data from ref *20* and unpublished data. [c] NA = not analyzed.

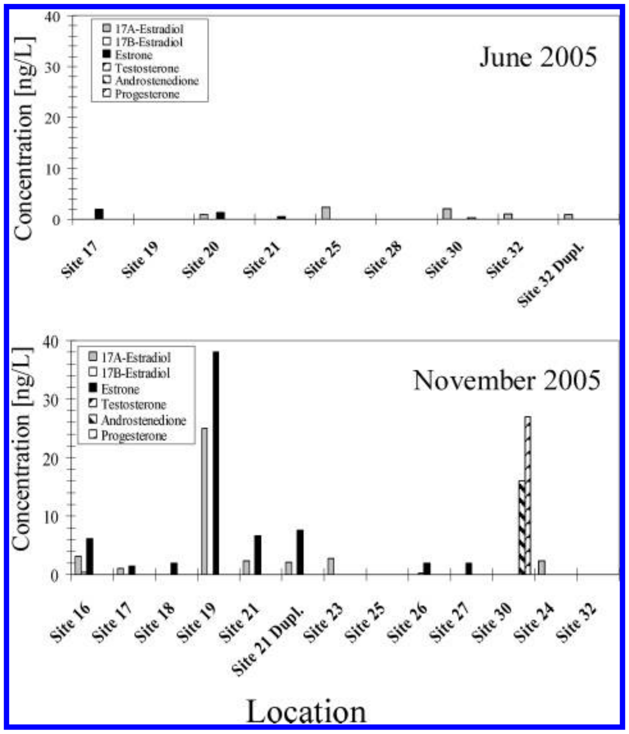

FIGURE 2. Steroid hormone concentrations in the Marin/Sonoma county sampling locations in June 2005 and after the first winter rainstorm in November. Locations are arranged in order of increasing streamflow, from left (lowest flow) to right (highest flow).

flowing, suggesting that a pulse of steroids occurred as winter rains leached steroids from cattle wastes deposited near the creeks. During the dry season, steroid concentrations were below 3 ng/L in all Marin/Sonoma county samples, with the exception of progesterone which was detected at 22 ng/L in one sample. After the first storm, 17α-estradiol, estrone, androstenedione, and progesterone were all detected at several sites in the Marin/Sonoma county watersheds at concentrations above 10 ng/L. Elevated concentrations of steroids also were detected in the Marin/Sonoma county sites after the second major winter storm in December 2005.

At the Stanislaus county sites (Figure 3), concentrations of steroids were similar before and after the first major winter storm on December 26−27, 2005. The differences between the two study areas may be due to the lower rainfall (15−20 mm) of this storm compared to the first storm at the Marin/Sonoma county sites (20−25 mm), to differing intensities of rainfall during the storms, to differing hill slopes, or to other geographical attributes of the study locations.

To investigate the contribution of cattle manure as a source of the steroids detected in the surface water samples, steroids in leachate from three cattle manure samples were quantified (Table S2). Although the data are limited by the small number of samples and a lack of information regarding the physiological status of the cattle that produced the manure, the steroids 17β-estradiol, estrone, testosterone, androstenedione, and progesterone were all detected in leachate samples. Estrone was present in all leachate samples, while the androgens detected in two of the three manure samples suggests that these particular samples were from male cattle.

In an attempt to correlate steroid concentrations to other water quality parameters, nitrate, total coliforms, and *E. coli* were measured in the samples (Table S2). Nitrate concentrations ranged from <0.1 to 70.8 mg/L (as $N-NO_3$). Total coliforms ranged from 480/100 mL to $>2.4 \times 10^6$/100 mL, while *E. coli* ranged from 0/100 mL to $>2.4 \times 10^6$/100 mL. Concentrations of *E. coli* were generally higher during the wet season, with the highest concentrations measured in wet season samples from the Marin/Sonoma county watersheds (Figure S1).

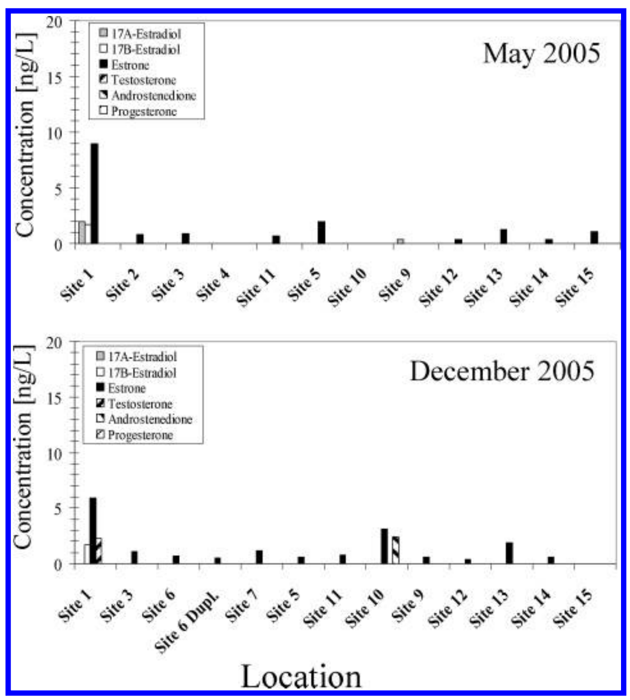

FIGURE 3. Steroid hormone concentrations in the Stanislaus county sampling locations in May 2005, and after the first winter rainstorm in December. Locations are arranged in order of increasing streamflow, from left (lowest flow) to right (highest flow). Locations 1−13 are headwater tributaries where grazing is the predominant land use, while locations 14−15 are downstream locations with little grazing.

Discussion

The concentrations of nitrate and fecal coliforms observed in the creeks in these grazing rangelands are similar to concentrations reported for other surface waters impacted by animal agriculture (*23*, *24*), indicating that animal wastes may be contaminating the creeks. Because animal wastes contain significant concentrations of steroids (*14*), it is not surprising that steroids also were present in the creeks. In these watersheds, cattle often have direct access to surface waters to drink; therefore, it is likely that animal wastes are introduced directly into these surface waters. The frequent detection of 17α-estradiol implicates cattle as the main source of the steroids, as this steroid is predominantly excreted by cattle and not by other livestock or humans (*15*). Additionally, the occurrence of steroids in the surface waters agrees with the data from the manure leachate, also implicating cattle as the main source of steroids. The exception was 17α-estradiol, which has been detected in cattle wastes (*25*) and was observed frequently in the watershed, but was not detected in the manure leachate samples. The most likely explanation for this observation is that 17α-estradiol is predominantly excreted by cattle in urine (*13*).

In the majority of the samples, the concentrations of estrogens were below levels that result in effects in fish. However, in some samples, concentrations of estrogens were high enough to potentially affect fish or other aquatic organisms. While concentrations of the most potent endogenous estrogen, 17β-estradiol, were generally low, with only two detections above 1 ng/L, 17α-estradiol and estrone were present at concentrations up to 25 and 38 ng/L respectively, and estrone was present in most samples. To assess the potential effects of these estrogens on fish, the estrogen concentrations were converted to 17β-estradiol equivalents and compared to predicted no-effect concentrations (PNECs; *26*, *27*). PNECs represent a concentration below which biological effects (i.e., vitellogenesis, morphological changes) have not been observed for sensitive species, while also accounting for the differing potencies of the estrogens (*26*, *27*). 17β-Estradiol equivalents were calculated using a potency factor of 1.0 for 17β-estradiol and 0.2 for estrone (*28*). The potency factor for 17α-estradiol is uncertain, with Khanal et al. (*28*) reporting a potency factor of 1−2 and Legler et al. (*29*) reporting a potency factor of 0.05. Assuming that 17α-estradiol is substantially less potent than 17β-estradiol (potency factor = 0.05), 9 of the 88 samples contained more than the 1 ng/L 17β-estradiol equivalent PNEC (*26*, *27*). By season, 6 samples exceeded PNECs during the dry season and 3 wet season samples exceeded PNECs. If a potency factor of 1.0 is used (*28*), the number of samples exceeding PNECs increases to 20; with 9 of these occurring during the dry season and 11 occurring during the wet season. The prevalence of samples exceeding PNECs in this study indicates that the estrogenic steroids originating from cattle wastes pose a potential risk to aquatic organisms in grazing rangeland surface waters.

In addition to the elevated concentrations of estrogens, relatively high concentrations of androstenedione and progesterone were observed in some samples (Table S2). In particular, these steroids were present at concentrations above 10 ng/L in samples from Site 30 during both the November and December sampling. Androstenedione is a known fish pheromone (*9*). The maximum concentrations of androstenedione detected in this study are above levels at which responsive fish detect this pheromone and are comparable to or higher than concentrations that would be expected to alter biochemistry and behavior in receptive fish (*9*, and personal communication, Peter W. Sorensen, University of Minnesota). The effects of androstenedione at these concentrations might include the inhibition of responses to other fish reproductive pheromones (*9*), potentially altering the fertility and reproductive success in fish species employing pheromonal communication.

The detection of progesterone in these rangeland surface waters was somewhat unexpected, as Schwarzenberger et al. (*30*) suggested that the rapid metabolic transformations of progesterone would decrease its concentrations very quickly. However, progesterone may be more stable than expected as evidenced by its previous detections in surface waters (*31*) and in municipal wastewater effluent (*32*). Additionally, progesterone, along with androstenedione, has been detected in pulp and paper mill effluent by Jenkins et al. (*33*), who hypothesized its formation from microbial transformations of phytosterols. Because androstenedione and progesterone were detected in leachate from cow manure, it is likely that the source of these steroids in the rangeland surface waters was animal wastes; however, phytosterol transformations cannot be ruled out as a source. Consistent with the findings of Jenkins et al. (*34*), it is possible that microbial transformations of phytosterols produce androstenedione and progesterone in the intestinal tract of livestock, raising the possibility that animal wastes could contain two separate pools of steroids: one due to excretion of endogenous steroids and one due to microbial transformations of phytosterols. Finally, because cattle excrete 5−30 times more progestins than either estrogens or androgens (*14*), it is likely that progestins other than progesterone (e.g., 17-hydroxyprogesterone, pregnanediol, pregnanetriol) also are present in animal wastes and contaminated surface waters.

The data indicate that certain locations within these watersheds more frequently exhibit elevated steroid hormone concentrations. For example, Site 1 in the Stanislaus county watershed consistently had some of the highest concentrations of steroids detected. Estrogen concentrations at Site 1 exceeded PNECs on three of the four sampling occasions (assuming 17α-estradiol potency factor = 0.05), while only one other sample from the Stanislaus county watershed exceeded PNECs. Although the land uses adjacent to Site 1 closely resembled the surrounding sampling locations, it also had one of the lowest streamflows of any location in this study, suggesting a relationship between flow and concentration. At Site 30 in the Marin/Sonoma county watersheds, androstenedione and progesterone concentrations exceeded 15 ng/L in both of the wet weather samples. Although more data are needed to determine the temporal variability in steroid concentrations, these results suggest that certain locations might have elevated concentrations of steroids that persist over time periods of a month or more. The persistence and timing of elevated steroid concentrations is a significant issue, as recent studies have demonstrated that exposing fish to low concentration mixtures of estrogenic chemicals, including steroids, resulted in reduced fecundity after periods of less than one week (*35*).

Although steroid concentrations were usually low, the maximum concentrations detected in receiving waters in grazing rangelands were comparable to, or in some cases, higher than, concentrations that were observed in surface waters near dairy operations (*20*) and in municipal wastewater effluent (*10*, *31*, *36*, *37*; Table 1). This suggests that fish in rangeland receiving waters might exhibit some of the symptoms associated with endocrine disruption as have been observed in fish downstream of municipal wastewater effluents. Observation of biological responses to steroids in agricultural watersheds is likely to be complicated by spatial and temporal variability in steroid concentrations. In rangeland receiving waters, steroid concentrations should vary with livestock density, behavior patterns, creek access, weather, streamflow, and geography. Additional research is needed to assess the relative importance of these variables and to identify biological effects.

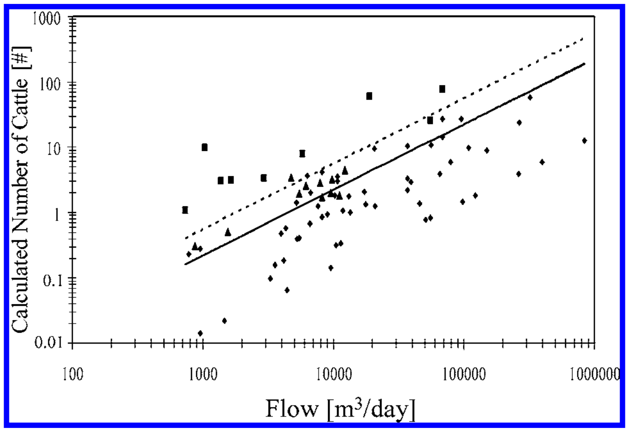

FIGURE 4. Minimum number of cows needed to account for all of the estrogens measured at each field site as a function of streamflow: (■) represent sites > PNEC if 17α-estradiol potency = 0.05 17β-estradiol potency; (▲) represent sites > PNEC if 17α-estradiol potency = 1.0 17β-estradiol potency; (◆) represent sites < PNEC. The dashed line plots the theoretical dilution capacity for each stream if 17α-estradiol potency = 0.05 17β-estradiol potency, while the solid line plots the theoretical dilution capacity if 17α-estradiol potency = 1.0 17β-estradiol potency.

Anecdotal field observations indicate that in some cases the detection of 17α-estradiol might be correlated with the presence of cattle near or in the creek upstream of the sampling location at the time sampling occurred. For example, some of the highest concentrations of 17α-estradiol measured in this study were at Site 1 (April 2005), Site 13 (August 2005), and Site 16 (November 2005). On these dates, cattle were observed near or in the creek 100−400 m upstream of the sampling location during sampling. Biotransformation reactions produce estrone from 17β-estradiol over time scales of 0.2−9 days in rivers (*28*), and 17α-estradiol is likely to react at similar time scales. Therefore, elevated concentrations of 17α-estradiol, and also 17β-estradiol, might be evidence that animal wastes were recently released to surface waters.

In the Stanislaus watershed, steroid concentrations were generally higher in headwater reaches where grazing is the predominant land use (sites 1−13) relative to downstream reaches where farming and residential land uses dominate (sites 14 and 15, see Figure 3 and Table S2). Headwater reaches contained steroids at concentrations up to 14 ng/L, while downstream reaches had no concentrations above 2 ng/L. Although the differences between headwaters and downstream locations are not statistically different due to small sample size, organisms in headwaters probably are exposed to higher steroid concentrations than those living downstream where agricultural steroid inputs are diluted with uncontaminated water from other sources.

The mass of steroids discharged from these watersheds was estimated using the streamflow measurements and data from nearby stream gauging stations. In these watersheds, the maximum mass discharges for estrogens were 800 mg/day, for androgens 3000 mg/day, and for progestins 1400 mg/day. The mass discharges can be used to estimate the minimum number of cows needed to account for the steroids measured in these samples (Figure 4 and Table S3) using previously published excretion data (*14*, *15*, *25*). For the estrogens, the data were used to determine the dilution capacity at each location, which is an estimate of the number of cattle that could discharge all of their wastes into the receiving waters before the total estrogen load exceeds the PNEC. The data in Figure 4 indicate that in small creeks (flow of ~1000 m^3/day, or ~0.012 m^3/s), one animal can elevate estrogen concentrations above PNEC values, while PNECs are exceeded in the largest creeks only by the wastes of 100 cattle or more. Therefore, if cattle are allowed direct access to receiving waters, each animal needs at least 1800−4500 m^3/day (range due to different values for 17α-estradiol potency) of streamflow to maintain estrogen concentrations below PNECs. For comparison, the estimated dilution capacity for nitrate is 250−300 m^3/day, assuming a PNEC of 1 mg NO_3−N/L for sensitive organisms (*23*, *38*, *39*). This estimate assumes that all nitrogen excreted by cattle is quickly oxidized to nitrate, and all excreted nitrogen is deposited in, or transported to, the receiving water. By this metric, 68% of the samples exceeded nitrate PNECs, although it is unclear whether the nitrate is entirely derived from animal wastes and not fertilizer or atmospheric deposition. These dilution estimates indicate that a larger volume of dilution water is needed for estrogens than for nitrate.

In cases where the density of cattle discharging to a watershed exceed the dilution capacity, rangeland management practices that limit the direct access of animals to the water could be used to reduce steroid concentrations below PNECs. Limiting the access of cattle to streams is likely to be effective because only a fraction of steroids produced by an animal will enter the water if they are deposited away from stream banks.

In the case of androstenedione and progesterone, our estimates suggest that some of the Marin/Sonoma county sites discharge steroid concentrations equivalent to hundreds or even thousands of cattle. The large number of cattle needed to explain these steroid concentrations suggests that large masses of dispersed steroids have been accumulating in the watershed during the dry season and are being transported during the first storm. Alternatively, it may indicate that steroids are being transported from concentrated sources of animal wastes, such as leaking animal waste lagoons or leachate from manure processing areas.

One of the objectives of this study was to correlate steroid occurrence and concentration to easily measured surrogate parameters that might be indicative of animal waste contamination. For this reason, nitrate and fecal coliforms were measured along with the steroids. The measured nitrate concentrations were comparable to those reported for receiving waters in agricultural areas contaminated with animal wastes (*23*, *39*), and the coliform concentrations in the rangeland creeks were similar to concentrations downstream of animal agriculture operations and pastures (*24*). Despite the fact that the highest concentrations of steroids and fecal coliforms occurred in the Marin/Sonoma county watersheds during the wet season, no direct correlation was observed between these parameters (data not shown). These data suggest that elevated fecal coliforms or nitrate might be useful in identifying impacted watersheds, but will not necessarily indicate the presence of elevated concentrations of steroids in a specific location.

The data from this study indicate that non-point sources of steroids such as grazing cattle can, in certain instances, elevate steroid concentrations in receiving waters to levels of concern for aquatic organisms. In the majority of the samples, steroid concentrations were not above threshold concentrations, suggesting that rangeland practices that allow cattle direct access to surface waters do not impact ecosystem health when there is adequate dilution of the receiving water. However, in approximately 10−20% of the rangeland samples, steroid concentrations exceeded PNECs for the feminization of fish, indicating that allowing cattle direct access to surface waters may impact the health of aquatic organisms in the receiving waters. Additionally, the detection of endogenous steroids in these rangeland watersheds likely indicates that natural and synthetic steroids administered to livestock for pharmaceutical purposes could be released through similar mechanisms. None of the common synthetic steroid hormones were analyzed during this study. One period where special concern might be warranted is immediately after heavy rains when wastes that have accumulated in the watershed are flushed from the system. These pulses of steroids may correspond to periods when certain species are vulnerable. For example, in some of the Marin/Sonoma county sites, endangered Coho salmon (*Onchorhynchus kisutch*) enter the watersheds to spawn soon after the first winter rains. Coincidentally, this is also the time period when the highest concentrations of steroids were detected. More research is needed to assess the temporal and spatial variation of steroid concentration during these periods, as well as the effect of anthropogenic sources of pheromonal steroids such as androstenedione on aquatic organisms.

Acknowledgments

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Supporting Information Available

Detailed description of the analytical methods, a table of the analytes and related instrument parameters, the complete data set, and a table of the mass discharges of the androgens and progesterone. This information is available free of charge via the Internet at http://pubs.acs.org.

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Livestock Grazing, Golden Trout, and Streams in the Golden Trout Wilderness, California: Impacts and Management Implications

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Abstract.—Impacts of livestock grazing on California golden trout *Oncorhynchus mykiss aguabonita* and their habitat were studied inside and outside of livestock exclosures in the Golden Trout Wilderness, California. In two consecutive years, the majority of stream physical characteristics showed large differences between grazed and ungrazed areas, and the directions of these differences were consistent with the recovery of exclosed streams and riparian areas from impacts caused by livestock grazing. Ungrazed areas consistently had greater canopy shading, stream depths, and bank-full heights and smaller stream widths than grazed areas. California golden trout were very abundant in the study sites; their densities and biomasses were among the highest ever recorded for stream-dwelling trout in the western United States. California golden trout density and biomass per unit area were significantly higher in ungrazed than in grazed areas in three of four comparisons. Differences between grazed and ungrazed areas were less consistent when density and biomass were calculated on the basis of stream length. Our results suggest that current levels of livestock grazing are degrading the stream and riparian components of the study meadows to the detriment of golden trout populations.

Grazing of domestic livestock is the most widespread land use in western North America (Wagner 1978). In the western United States, grazing occurs on the majority of federal lands, including national forests, national wildlife refuges, lands administered by the U.S. Department of Interior's Bureau of Land Management (BLM), and some national parks. In the western states, grazing affects 64 million hectares administered by the BLM and 53 million hectares administered by the U.S. Forest Service (Armour et al. 1994). Impacts of livestock grazing on stream and riparian ecosystems are widespread (for recent reviews, see Kauffman and Krueger 1984; Platts 1991; Fleischner 1994) and are particularly acute in the arid western states. In arid climates, lush vegetation is often found only near stream corridors and as a result, livestock tend to congregate in these areas (Roath and Krueger 1982; Gillen et al. 1984). Although western riparian zones are the most productive habitats in North America (Johnson et al. 1977), at least 50% of these ecosystems are degraded as a consequence of livestock grazing (Armour et al. 1994). A recent review of conditions on U.S. Forest Service lands also concluded that most riparian ecosystems are in need of restoration (USGAO 1988).

Livestock grazing directly affects three general components of stream and riparian ecosystems: streamside vegetation; stream channel morphology, including the shape of the water column and streambank structure; and water quality (Platts 1979; Kauffman and Krueger 1984). These impacts can alter the population structure of resident fish, particularly salmonids (Platts 1991). Although the spatial and temporal variability of stream salmonids may often obscure any population changes caused by land management practices (Hall and Knight 1981; Platts and Nelson 1988), a recent review reported that 15 of 19 studies showed that stream fish were diminished in the presence of livestock grazing (Platts 1991).

In 1993 and 1994, we conducted a study of grazing impacts to streams in the Golden Trout Wilderness, California, using a series of livestock exclosures. The 133,500-ha Golden Trout Wilderness (GTW) was created in 1977, in part to protect the habitat of the two subspecies of golden trout *Oncorhynchus mykiss* sspp. The Little Kern golden

[1] To whom reprint requests should be sent.

trout *O. m. whitei*[2] is native to the Little Kern River and is currently listed as a threatened species under the federal Endangered Species Act (Behnke 1992). The California golden trout *O. m. aguabonita* is native to the South Fork Kern River and Golden Trout Creek (Behnke 1992), and most of its native range lies within the GTW. The basic ecology of the California golden trout remains poorly understood, although recent research on stream populations in the GTW shows that individuals are long lived, slow growing, and exist at high densities (Knapp and Dudley 1990). The California golden trout has been the subject of much management interest because of its status as California's state fish, its limited natural distribution, and several perceived threats to its viability, including introduction of nonnative brown trout *Salmo trutta* and habitat degradation caused by livestock grazing. Because of these threats, the California golden trout is being considered for federal listing as a threatened species.

Although much attention has been focused on damage caused by past livestock grazing in the GTW (e.g., T. A. Felando, Inyo National Forest, unpublished report, 1982), very little is known about whether current levels of livestock grazing are causing additional degradation of stream and riparian ecosystems. Therefore, we quantified a series of riparian, stream, and fishery variables inside and outside three grazing exclosures to address the following questions: (1) Are stream and riparian habitat variables different between areas inside and outside of exclosures? and (2) Are the density and biomass of golden trout different between areas inside and outside of exclosures?

Study Area

The GTW is at the southern end of the Sierra Nevada, California (118°15'N, 36°22'W). This study was confined to the eastern portion of the GTW in the Inyo National Forest. This area was largely unaffected by Pleistocene glaciation (Odion et al. 1988) and is characterized by large subalpine meadows (up to approximately 7.5 km^2). These meadows are found primarily along the South Fork Kern River and a major tributary, Mulkey Creek. Meadows are dominated by sagebrush *Artemisia cana*, but streamside zones are typically dominated by sedge *Carex* spp. and willow *Salix* spp. (Odion et al. 1988; Sarr 1995). Over 90% of the annual precipitation falls as snow (Major 1977), and the remainder mostly occurs during summer thunderstorms.

Livestock have grazed the area now contained within the GTW since at least 1860, and there are reports of 200,000 sheep in the area during a year between 1860 and 1890 (Felando, unpublished report) and of 10,000 cattle in the late 1800s (Inyo National Forest 1982). Past overgrazing has resulted in widespread riparian degradation (Albert 1982; Felando, unpublished report), and large-scale restoration efforts have been implemented by the U.S. Forest Service (Inyo National Forest) during the past 70 years.

Mulkey and Ramshaw meadows are currently grazed by approximately 950 cow–calf pairs (in 1993, 235 in Mulkey and 700 in Ramshaw; in 1994, 235 in Mulkey and 730 in Ramshaw). Mulkey Meadow is typically grazed for several weeks in July and again in September. In Ramshaw Meadow, cattle are generally trailed through in late July with only light grazing and are gathered into the meadow for a week of high-intensity grazing in October.

The fish fauna of this watershed is composed of two native species, the California golden trout and Sacramento sucker *Catostomus occidentalis*. However, we encountered only California golden trout during our surveys. These populations are self-sustaining, are not subject to management activities (e.g., fish stocking), and experience very light angling pressure.

The Inyo National Forest constructed exclosures in several GTW meadows in 1983 and 1991 to protect stream segments from grazing impacts. Our study sites were inside and outside three grazing exclosures in Ramshaw Meadow (2,660 m; Figure 1) and Mulkey Meadow (2,850 m; Figure 2). Cattle rarely trespass inside these exclosures (D. Hubbs, Inyo National Forest, unpublished data), and we consider stream sections inside the exclosures to have been ungrazed since exclosures were constructed. All stream reaches used in our study were typical of those found in low-gradient meadows (types C-4 and E-4 of Rosgen 1994).

Study Design

An inherent problem with studies that use exclosures to investigate the impacts of livestock grazing is treatment (i.e., exclosure) replication. The most statistically robust study design would incorporate numerous randomly placed exclosures. Such a design is difficult to achieve because of the limited availability of grazed sites with sim-

[2] Although the systematics of western salmonids remains controversial, we use the most recent classification and common names as described by Behnke (1992).

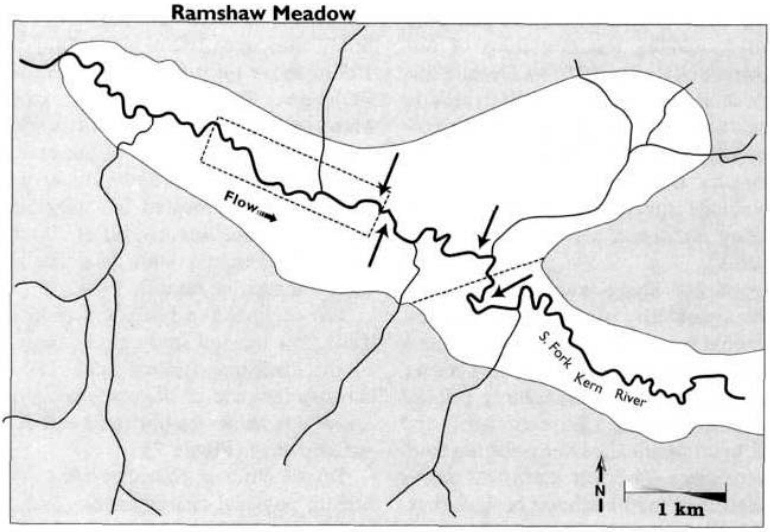

FIGURE 1.—Map of Ramshaw Meadow. The upper exclosure is depicted by the dotted rectangle and the lower drift fence exclosure is depicted by the dotted line across the stream and meadow. Arrows show study sites inside and outside exclosures. Shaded area is forest surrounding the meadow.

ilar site potential and the cost of constructing multiple exclosures (Platts and Wagstaff 1984); as a result, we are aware of very few studies that have used such a design (Buckhouse et al. 1981; Kauffman et al. 1983; Platts and Nelson 1985a). Much more common are grazing studies that take advantage of exclosures placed by land management agencies to protect particular stream sections from grazing impacts (Rinne 1988). These studies frequently are based on comparisons of stream characteristics inside and outside of a single exclosure (Keller and Burnham 1982; Platts and Nelson 1985b; Odion et al. 1988; Kondolf 1993). However, statistical analyses based on differences in-

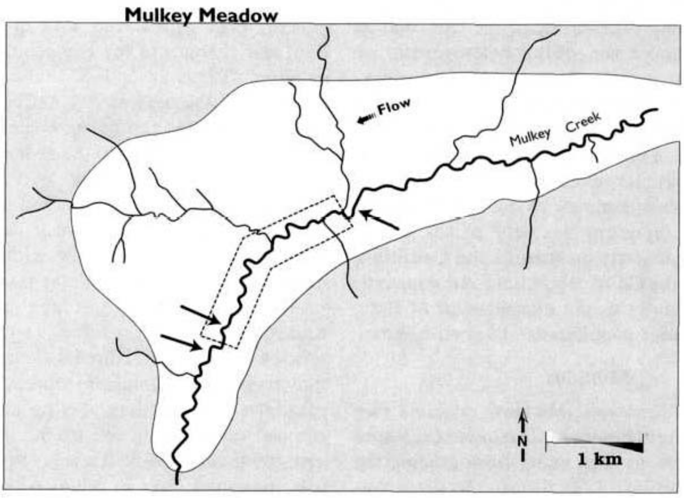

FIGURE 2.—Map of Mulkey Meadow. The exclosure is depicted by the dotted rectangle. Arrows show study sites below, inside, and above the exclosure. Shaded area is forest surrounding the meadow.

side and outside of single exclosures are generally pseudoreplicated, making interpretations of differences problematic (Hurlbert 1984). Despite this flaw and other shortcomings inherent to exclosure studies (Rinne 1988), unreplicated exclosures often provide the only avenue available for the study of grazing impacts. Indeed, such studies provide the bulk of available information on the effects of livestock grazing on stream and riparian ecosystems (Platts 1991).

The design of our study was similarly constrained by the availability of grazing exclosures. Although we could have used our three exclosures as replicate treatments, we chose not to because the exclosures were placed nonrandomly (violating assumptions underlying statistical tests) and because of the low statistical power resulting from a sample size of three. This low statistical power would have increased the likelihood of type II error, the finding of no significant difference when a true difference exists. As a result, we chose to make separate grazed versus ungrazed comparisons for each exclosure.

The location of exclosures also constrained our placement of grazed and ungrazed study sites. Although it may have been statistically more appropriate to place habitat transects along the entire stream length inside the exclosures and over similar distances outside exclosures, we were prevented from using this design because the exclosures contained several very different stream channel types. Because comparison of stream and riparian conditions across disparate channel types could obscure any differences in the sites due to grazing, we reduced the spatial heterogeneity of sites used in our grazed versus ungrazed comparisons by locating study sites as close together as possible (only Rosgen's C-4 and E-4 channel types were included). This design reduced the spatial scale of the study, however.

Despite the shortcomings of our study design, the exclosures represent the only means of assessing grazing impacts on streams and California golden trout in the GTW. We believe our data contribute meaningfully to the management of California golden trout populations and their habitat.

Methods

Study sites.—Ramshaw Meadow contains two grazing exclosures (Figure 1). The lower exclosure was built in 1991 to keep cattle from grazing the entire lower portion of Ramshaw Meadow. Approximately 700 cattle are trailed through the exclosure every year in early and late summer but affect only a very small portion of the exclosure. Study sites were 100 m below (ungrazed site) and 100 m above (grazed site) the upstream end of the exclosure. The exclosure in upper Ramshaw Meadow was built in 1983. The ungrazed site was just inside the lower end of the exclosure and the grazed site was 100 m downstream of the exclosure (Figure 1). We located the ungrazed site at the exclosure fence line instead of 100 m upstream to allow comparisons with data that had been collected at this location in 1984.

The exclosure in Mulkey Meadow was built in 1991. We located study sites 100 m downstream of the exclosure (grazed site), 100 m above the downstream end of the exclosure (ungrazed site), and 100 m above the upstream end of the exclosure (grazed site) (Figure 2).

Stream physical characteristics.—We quantified stream physical characteristics and surveyed fish populations within each study site during August 20–30, 1993, and August 16–24, 1994. All sites contained 125 m of stream, and at each site we measured characteristics along each of 25 transects spaced 5 m apart and arranged perpendicular to stream flow (Simonson et al. 1994). At each transect, we measured channel width, channel depth, stream width, stream depth, bank-full height, bank overhang, bank angle, and bank water depth. These variables are potentially sensitive to land use activities, such as livestock grazing, that influence channel stability. We defined channel width as the cross section containing the stream that is distinct from the surrounding area due to breaks in the general slope of the land, lack of upland vegetation, and changes in the composition of substrate materials (Platts et al. 1983). Channel width was measured to the nearest 10 cm. Channel depth, the distance from the top of the channel to the water surface, was measured to the nearest 5 cm. Stream width, the width of the present water surface not including islands, was measured to the nearest 5 cm. We measured unvegetated steam width, the total stream width minus the width of any bands of emergent vegetation along each bank, to the nearest 5 cm. This vegetation is typically the beaked sedge *Carex rostrata* (=*C. utriculata*), a species that rapidly colonizes stream margins in the absence of grazing in Mulkey and Ramshaw meadows and that may play an important role in channel stabilization and stream narrowing (Rosgen 1994; Sarr 1995). Unvegetated stream width was measured only in 1994. Although bank-full height is generally defined as the height reached by a stream on average very 1.5 years (Gordon et al. 1992), this measure is unlikely to be strongly influenced by land management practices. Instead, we defined bank-full height as the height above the current water level at which the banks lose their ability to contain the stream (Gordon et al. 1992); it was measured to the nearest 5 cm.

We quantified streambank morphology by measuring bank angle, bank overhang, and bank water depth of both banks along each transect. We measured bank angle to the nearest 5° using a clinometer on a 1.5-m rod placed against the bank slope (Platts et al. 1983). Overhanging banks have bank angles of 0–89°, and laid-back banks have bank angles of 90–180°. If banks were overhanging, the extent of overhang was measured to the nearest 5 cm from the deepest bank undercut to the furthest point of bank protrusion. We defined bank water depth as the water depth 15 cm from each bank and measured it to the nearest 5 cm.

To quantify instream characteristics, we measured vegetative canopy shading, substrate composition, and, at equally spaced points along each transect, water depth, water velocity, and height of any aquatic vegetation. Canopy shading was measured at each bank and in midstream by facing up and downstream with a densiometer (Platts et al. 1983). Water depth, water velocity, and height of submerged vegetation were measured at 10 equally spaced points along each transect in 1993, and at 5 equally spaced points in 1994. The reduced number of points in 1994 was necessitated by extremely low flows that would have caused points to be too close together (often <10 cm). Water depth and height of submerged vegetation were measured to the nearest 1 cm. Water velocity was measured with a Marsh-McBirney[3] model 2000 current meter, and each measurement represented a 10-s average. Velocities at each point were measured at 60% of the water depth with a top-setting wading rod.

In 1993, we quantified substrate particle size distributions (geometric mean diameter—D_g of Platts et al. 1979—and percent fines) by taking a core sample with a shovel at the deepest point along each transect. The shovel was inserted into the substrate to a depth of 15 cm and then lifted from the stream (Grost et al. 1991). Samples were placed into resealable plastic bags for transport to a laboratory, where they were dried for a minimum of 72 h to a constant weight and separated into 11 size-fractions with a mechanical shaker. In 1994, we measured substrate particle size distributions by measuring the substrate contacted by the upstream edge of the wading rod base at each of the five equally spaced points along each transect. Particles 1 mm or larger in diameter were measured to the nearest millimeter. Particles smaller than 1 mm were classified as fine sand (0.5 mm) or silt (0.1 mm). We used this technique in 1994 instead of the shovel sampler because of the difficulty of transporting heavy samples out of the remote study areas and the time-consuming nature of the sifting process.

To quantify the size-structure of riparian willows, we counted and measured heights of all willows within 2 m of each streambank at each site in 1994. Willow heights were measured to the nearest 1 cm.

Of the 14 measured variables, most were expected to respond relatively quickly and in a predictable direction to removal of livestock from the streamside zone (Platts 1991). For these variables, we predicted the following changes in ungrazed areas inside exclosures relative to grazed areas outside exclosures.

(1) Total stream width and unvegetated stream width will have decreased as vegetation invaded the channel and banks stabilized.
(2) Bank-full height will have increased as vegetation colonized point bars and captured sediment.
(3) Canopy shading will have increased as willow and sedge species recolonized streambanks.
(4) Bank angle will have decreased and bank overhang and bank water depth will have increased as banks stabilized and were transformed from laid-back to undercut configurations.
(5) Stream water depth will have increased as stream width decreased and constricted stream flow.

Several additional measured variables were expected to change very slowly in response to livestock exclosure or were not anticipated to change in a predictable direction. We included these in our study to allow post hoc determinations of whether sites inside and outside of excolsures were similar. Large differences in most or all of these variables would suggest that paired sites differed before exclosures were built. Channel width and channel depth were expected to respond to the removal of livestock grazing, but these changes were likely to be measurable only over a period of several decades and not the 2–11 year time scale used

[3] Trade names and commercial enterprises are mentioned solely for information. No endorsement by the U.S. Forest Service is implied.

in this study (Kondolf 1993). For two additional variables, substrate size and water velocity, we did not predict any direction of change. Although substrate size would be expected to increase after the removal of livestock as a consequence of a reduction in delivery of fine sediments to the stream, livestock grazing has continued upstream of the exclosures. Therefore, even if fine sediment inputs were reduced within the exclosures, we expected these changes to be masked by continued inputs from upstream (Rinne 1988).

Although we do not have preexclosure data for most sites, we did have a detailed map (accurate to 10 cm) drawn in 1984 of the stream section inside the upper Ramshaw exclosure (T. L. Dudley and R. A. Knapp, unpublished data). To determine whether stream width inside the upper Ramshaw exclosure had changed since the time of exclosure construction in 1983, we compared the mean stream width from the map with the mean stream width at the site in 1993 and 1994. To extract total stream widths from the 1984 map, we drew 25 transects on the map, each perpendicular to stream flow and spaced 5 m apart. At each transect, we measured the total stream width with a ruler and converted these measurements to their actual dimensions (1 cm = 1 m). If stream width had not changed since 1984 (i.e., no recovery had occurred), the mean stream width as measured from the 1984 map should have been similar to the stream width measured in 1993 and 1994.

Fish population characteristics.—We surveyed fish populations using standard electrofishing depletion techniques (Van Deventer and Platts 1983). To facilitate electrofishing, we divided each 125-m site into five 25-m sections with block nets. We conducted three passes through each section with a Smith-Root type XII electrofisher that produced 400 V and a pulse frequency of 90 Hz. The length of time that the electrofisher was running on each pass was similar within a section to ensure a similar electrofishing effort on each pass. Captured fish were measured for fork length to the nearest 1 mm and weighed on an electronic balance to the nearest 0.1 g. Fish were released into the section from which they were captured after the final pass within the section was completed.

The number of fish in each site was estimated from the rate of depletion by maximum-likelihood estimation techniques (Microfish, version 3.0 software; Van Deventer and Platts 1985). The depletion data used in the maximum-likelihood estimations were obtained by adding all fish from the first pass in sections 1–5 (= total number of fish in pass 1), the number of fish from the second pass in sections 1–5 (= total number of fish in pass 2), and the number of fish from the third pass of sections 1–5 (= total numbers of fish in pass 3) (Van Deventer and Platts 1985). Capture probabilities were similar among size-classes—in 1993 they were 0.50 for adults ($\geq$100 mm) and 0.36 for age-0 fish (55 mm) (paired t-test, $N = 7$, $P > 0.09$); in 1994 they were 0.59 and 0.52, respectively ($P > 0.2$)—and all size-classes were pooled for the population estimates. The number of fish per square meter was calculated for each 125-m site by dividing the estimated number of fish per site by the stream surface area (average stream width $\times$ 125 m). The estimated total weight of fish in each section was extrapolated from the mean weight of fish captured. We calculated fish weight per square meter by dividing the estimated total weight per site by the stream surface area.

On the basis of a recent review of livestock impacts on stream fish populations (Platts 1991), we predicted that California golden trout density (number/125 m, number/m^2) and biomass (g/125 m, g/m^2) should be greater in ungrazed than grazed areas. Although recent evidence shows that trout of some species move considerable distances (Young 1995a) and that movement may confound studies of habitat-related differences in trout population structure when sites are close to each other (Young 1995b), other research indicates that adult California golden trout rarely move more than a few meters (Matthews 1996). In addition, numerous grazing studies have documented differences in trout populations in adjacent study sites inside and outside exclosures (Platts 1991).

Statistical analysis.—To provide an overview of the differences between sites inside and outside the three exclosures, we tallied the number and direction of differences in physical stream characteristics. If livestock grazing outside exclosures had not influenced stream characteristics, an equal number of changes in variables should have agreed and disagreed with expectations. Differences from equality were evaluated with the binomial test. To determine the magnitude of differences between paired grazed and ungrazed sites, we also treated the upper Ramshaw, lower Ramshaw, and Mulkey exclosures as separate analyses. Most physical stream variables could not be normalized for particular sites, so we relied primarily on nonparametric Kruskal–Wallis one-way analyses of variance (ANOVA) to test for differences in values inside and outside of exclosures. One- or two-tailed tests were used according to the null hypothesis being tested. Because of the pseudoreplication problems inherent in statistical comparisons of single sites inside and outside of an exclosure (e.g., artificially inflated sample sizes; Hurlbert 1984), we present P-values for comparisons of site characteristics only to provide a relative measure of the magnitude of differences, and not to draw conclusions based solely on statistical significance ($P \leq 0.05$).

Fish data were analyzed as estimated density (number/125 m and number/m^2) and biomass (g/125 m and g/m^2) per site with modified t-tests. The estimated number of fish per 125 m was compared between grazed and ungrazed sites with the following t-test formula (Keller and Burnham 1982):

$$t = \frac{|(\text{number}_1) - (\text{number}_2)|}{\sqrt{(\text{SE}_1)^2 + (\text{SE}_2)^2}};$$

subscript 1 refers to the grazed site of a pair, subscript 2 to the ungrazed site; standard errors were calculated by the maximum-likelihood model. Degrees of freedom were calculated with the formula:

$$\text{df} = n\left\{\frac{\left(\frac{1}{n}\sum \text{var}_1 + \frac{1}{n}\sum \text{var}_2\right)^2}{\left(\frac{1}{n}\sum \text{var}_1\right)^2 + \left(\frac{1}{n}\sum \text{var}_2\right)^2}\right\};$$

n = number of sections, and var is variance. Fish weight per 125 m was compared between sites with the t-test formula:

$$t = \frac{|(\text{weight}_1) - (\text{weight}_2)|}{\sqrt{(\text{SE}_1 \cdot X_{\text{wt}1})^2 + (\text{SE}_2 \cdot X_{\text{wt}2})^2}},$$

where (weight$_1$) and (weight$_2$) are the estimated weight of fish in site 1 and site 2, (SE$_1$) and (SE$_2$) are the standard errors of (number$_1$) and (number$_2$) calculated by the maximun-likelihood model, and $X_{\text{wt}1}$ and $X_{\text{wt}2}$ are the average individual fish weights in sites 1 and 2. Multiplying the standard error by the average individual fish weight was necessary to scale the standard error to fish weight. Comparisons of the number of fish per square meter and weight of fish per square meter were made with formulas similar to those given above except that (number$_i$) and (SE$_i$), and (weight$_i$) and (SE$_i \cdot \bar{X}_{\text{wt}i}$) were first divided by the stream surface area of site i to scale these variables to area.

As with the habitat data, analysis of the fish data involved statistical comparisons of single sites inside and outside of an exclosure and are therefore pseudoreplicated. As a result, we present P-values associated with comparisons of fish population characteristics only to provide a relative measure of the magnitude of differences and not to draw conclusions based solely on statistical significance ($P \leq 0.05$).

TABLE 1.—Differences in stream physical characteristics in relation to predicted changes between paired sites inside (ungrazed) and outside (grazed) exclosures in 1993 and 1994.

Exclosure comparison (ungrazed versus grazed)	Difference consistent with prediction	No difference	Difference opposite from prediction
Upper Ramshaw, 1993	6	1	0
Upper Ramshaw, 1994	8	0	0
Lower Ramshaw, 1993	3	2	2
Lower Ramshaw, 1994	7	0	1
Mulkey (versus site below), 1993	7	0	0
Mulkey (versus site below), 1994	5	0	3
Mulkey (versus site above), 1993	7	0	0
Mulkey (versus site above), 1994	8	0	0
All, 1993	23	3	2
All, 1994	28	0	4

Results

Stream Physical Characteristics

All of the protected sites showed differences in stream physical characteristics that were consistent with changes expected following the removal of livestock from streamside zones. In 1993, of 28 comparisons between paired ungrazed and grazed sites for variables that we predicted would change in a particular direction, 23 (82%) differed in the predicted direction, 3 (11%) showed no change, and 2 (7%) changed in directions opposite to what was predicted (Table 1). In 1994, of the 32 comparisons made, 28 (88%) differed in the predicted direction and 4 (12%) changed opposite to predictions (Table 1). The difference between the numbers of confirmed and contradicted predictions was statistically significant in both years (1993, 23 versus 2; $P < 0.001$; 1994, 28 versus 4; $P < 0.001$). Canopy shading and bank water depth showed the greatest number of predicted differences, showing differences in all comparisons in 1993 and 1994. Water depth, stream width, and bank-full height showed differences in the predicted directions in 75–100% of the comparisons, and unvegetated stream width and bank angle showed the predicted differences in 75% of the comparisons. Bank overhang showed the predicted differences in 50–75% of the comparisons.

Of the variables with no predicted direction of change, only channel depth and width showed consistent differences between inside and outside the exclosures. Channel depth was shallower in grazed sites in all of the 1993 comparisons and in 75% of the 1994 comparisons. Channel width was greater in grazed areas in all of the 1993 and 1994 comparisons. Substrate size, percent fine sediment, and water velocity showed no consistent differences between inside and outside of exclosures.

Stream physical habitat results were very similar between 1993 and 1994 (Table 1); therefore, we present detailed stream physical characteristics for each site for 1994 only.

Upper Ramshaw Meadow

In 1994, one of the four variables for which there was no predicted direction of change (channel width, channel depth, substrate size, and water velocity) showed a large difference (defined as one exceeding 20%) between sites inside and outside the upper Ramshaw exclosure (Table 2): channel width was 47% greater below than inside the exclosure. Of the variables that we expected to change in a predicted direction as a result of livestock exclusion, all showed differences consistent with changes expected after the cessation of livestock grazing, and six of the eight differences were larger than 20% (Table 2). Bank-full height, bank overhang, and bank water depth were 75–100% greater inside than outside the exclosure. Canopy shading was 250% greater inside the exclosure than outside.

In 1994, stream width was 34% narrower inside the exclosure than outside (Table 2). To determine whether this difference was the result of a change (i.e., recovery) that had occurred inside the exclosure since 1984, we compared the stream width inside the exclosure obtained from the 1984 channel map to the 1993 and 1994 field measurements of stream width. Stream width inside the exclosure in 1984 was significantly wider than stream width in both 1993 and 1994 (1984, 345 cm; 1993, 271 cm; 1994, 230 cm; ANOVA on log-transformed data; $P < 0.0001$). Differences in stream width between 1993 and 1994 were not significant (Tukey pairwise comparison of means; $P > 0.05$). Therefore, stream width had narrowed significantly inside the exclosure since 1984 (i.e., recovery was occurring).

Willows were much more abundant within the exclosure; 264 willows were counted inside the exclosure and 11 willows were counted below the exclosure (Figure 3A). Willows inside covered a much wider range of heights than those below the exclosure (inside, 5–220 cm; below, 10–70 cm). Also, willows 5–20 cm tall were abundant inside the exclosure but nearly absent below the exclosure.

Lower Ramshaw Meadow

One of the four variables for which there was no predicted direction of change showed a large difference between sites inside and outside the lower Ramshaw exclosure (Table 2): substrate size was 59% larger above than inside the exclosure. Of the eight variables that we expected to change in a predicted direction as a result of livestock exclusion, seven showed differences in the predicted directions, and four of these differences were larger than 20% (Table 2). Unvegetated stream width and total stream width were 31 and 19% narrower inside the exclosure than above the exclosure, respectively, and stream depth, bank water depth, and canopy shading were 30–50% greater inside than above the exclosure. The difference in bank angle was in the opposite direction of what we predicted (bank angle was larger in the ungrazed site), but this difference was small (5%).

As in upper Ramshaw Meadow, willows were much more abundant within the lower exclosure; 222 willows were counted inside the exclosure and 22 willows were counted above the exclosure (Figure 3B). Willows inside the exclosure also showed a much wider size range than those outside the exclosure (inside, 5–170 cm; above, 10–60 cm).

Mulkey Meadow

One of the four variables for which there was no predicted direction of change showed a large difference between sites inside and below the Mulkey exclosure (Table 2): water velocity was 40% higher below the exclosure than inside it. Of the variables that we expected to change in a predicted direction as a result of livestock exclusion, five differed in the predicted direction, and three of these differences were larger than 20% (Table 2). Canopy shading, bank water depth, and stream depth were 25–33% greater inside than below the exclosure. Three variables differed in the opposite direction from our predictions, but all of these differences were smaller than 20% (Table 2). Stream width and unvegetated stream width were 12 and 10% wider inside the exclosure than outside, and bank overhang was 17% less inside the exclosure.

In the comparison of physical characteristics between sites inside and above the exclosure, three of the four variables for which there was no predicted direction of change showed large differ-

TABLE 2.—Means (SEs) of stream physical characteristics outside (grazed) and inside (ungrazed) the upper Ramshaw, lower Ramshaw, and Mulkey exclosures in 1994.

Variable	Grazed	Ungrazed	Percent difference[a]	N	P[b]
Upper Ramshaw exclosure versus below exclosure					
Channel depth (cm)	61 (2)	69 (2)	13	25	0.008
Channel width (cm)	899 (39)	613 (34)	47	25	<0.001
Substrate size (mm)	4.7 (0.4)	4.6 (0.4)	2	125	0.90
Water velocity (cm/s)	17 (1)	19 (1)	12	125	0.30
Stream width (cm)	309 (19)	230 (11)	34+	25	<0.001
Unvegetated stream width (cm)	290 (17)	200 (10)	45+	25	<0.001
Bank-full height (cm)	16 (2)	28 (2)	75+	25	<0.001
Canopy shading (%)	10 (2)	35 (3)	250+	25	<0.001
Bank angle (degrees)	145 (5)	124 (5)	14+	50	<0.001
Bank overhang (cm)	2 (1)	4 (1)	100+	50	0.07
Bank water depth (cm)	8 (1)	16 (1)	100+	50	<0.001
Stream depth (cm)	12 (1)	14 (1)	17+	125	0.03
Lower Ramshaw exclosure versus above exclosure					
Channel depth (cm)	47 (1)	44 (2)	6	25	0.14
Channel width (cm)	569 (40)	583 (38)	2	25	0.14
Substrate size (mm)	3.5 (0.3)	2.2 (0.3)	59	125	<0.001
Water velocity (cm/s)	16 (1)	15 (1)	7	125	0.94
Stream width (cm)	283 (18)	238 (8)	19+	25	0.09
Unvegetated stream width (cm)	276 (19)	210 (9)	31+	25	0.01
Bank-full height (cm)	18 (2)	20 (2)	11+	25	0.42
Canopy shading (%)	14 (2)	21 (3)	50+	25	0.19
Bank angle (degrees)	114 (7)	120 (8)	5−	50	0.38
Bank overhang (cm)	7 (2)	8 (2)	14+	50	0.87
Bank water depth (cm)	10 (1)	13 (1)	30+	50	0.02
Stream depth (cm)	10 (1)	14 (1)	40+	125	<0.001
Mulkey exclosure versus below exclosure					
Channel depth (cm)	56 (2)	64 (1)	14	25	0.002
Channel width (cm)	489 (36)	426 (17)	15	25	0.39
Substrate size (mm)	6.1 (0.9)	6.5 (0.9)	6	125	0.37
Water velocity (cm/s)	14 (1)	10 (1)	40	125	0.01
Stream width (cm)	130 (11)	146 (10)	12−	25	0.17
Unvegetated stream width (cm)	111 (9)	122 (11)	10−	25	0.50
Bank-full height (cm)	15 (2)	18 (2)	20+	25	0.30
Canopy shading (%)	24 (2)	32 (3)	33+	25	0.05
Bank angle (degrees)	114 (6)	112 (6)	2+	50	0.60
Bank overhang (cm)	7 (2)	6 (2)	17−	50	0.55
Bank water depth (cm)	16 (2)	21 (2)	31+	50	0.03
Stream depth (cm)	16 (1)	20 (1)	25+	125	0.01
Mulkey exclosure versus above exclosure					
Channel depth (cm)	41 (2)	64 (1)	56	25	<0.001
Channel width (cm)	759 (33)	426 (17)	78	25	<0.001
Substrate size (mm)	8.2 (1)	6.5 (0.9)	26	125	0.007
Water velocity (cm/s)	9 (1)	10 (1)	11	125	0.72
Stream width (cm)	162 (7)	146 (10)	11+	25	0.04
Unvegetated stream width (cm)	140 (7)	122 (11)	15+	25	0.02
Bank-full height (cm)	11 (1)	18 (2)	63+	25	0.01
Canopy shading (%)	2 (1)	32 (3)	1,500+	25	<0.001
Bank angle (degrees)	152 (4)	112 (6)	36+	50	<0.001
Bank overhang (cm)	1 (1)	6 (2)	500+	50	0.06
Bank water depth (cm)	10 (1)	21 (2)	110+	50	<0.001
Stream depth (cm)	10 (1)	20 (1)	100+	125	<0.001

[a] A "+" after the percent difference indicates the direction of the difference is consistent with changes expected after the removal of livestock; a "−" after the percent difference indicates that the difference is opposite to the changes expected after the removal of livestock; the lack of a symbol indicates that there was no predicted direction of change.

[b] Comparisons are pseudoreplicated. Therefore, P-values are given only for comparison of relative magnitudes of differences.

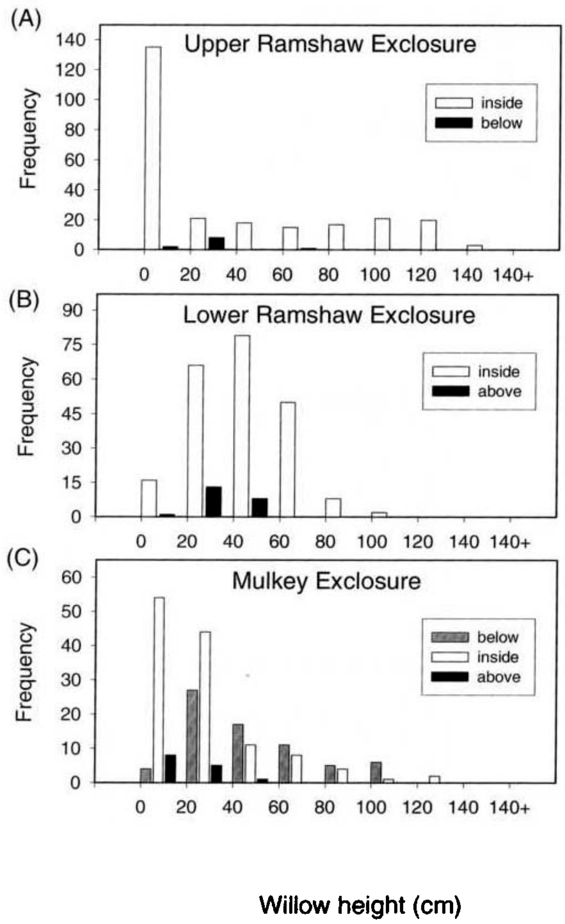

FIGURE 3.—Size-frequency histograms of willows (A) inside and below the upper Ramshaw exclosure, (B) inside and above the lower Ramshaw exclosure, and (C) below, inside, and above the Mulkey exclosure. Frequency of willows is presented in 20-cm size ranges.

ences (Table 2). Above the exclosure, the channel was 56% shallower and 78% wider, and substrate size was 26% larger, than inside the exclosure. Of the variables that we expected to change in a predicted direction as a result of livestock exclusion, all eight showed differences in the predicted directions, and six of the differences were larger than 20% (Table 2). Bank water depth and stream depth were 100–110% greater inside than above the exclosure. In addition, canopy shading was 15 times greater and bank overhang was 5 times greater inside than above the exclosure.

Willows were more abundant within the exclosure than either above or below the exclosure; 124 willows were counted inside the exclosure, 14 above the exclosure, and 70 below the exclosure (Figure 3C). The size range of willows inside and below the exclosure was greater than the size range above the exclosure (inside, 5–140 cm; below, 10–120 cm; above, 5–60 cm). Small (5–20 cm) willows were much more abundant inside the exclosure than either above or below the exclosure.

Fish Population Characteristics

Three electrofishing passes through each 25-m section allowed the capture of nearly all fish in a 125-m site. There were small differences between actual and estimated total fish per section and low SEs around population estimates. The SE associated with the estimated number of fish per site was 3.6% of the estimate in 1993 (N = 7; range, 2.3–4.8%) and 2.8% of the estimate in 1994 (N = 7;

range, 1.3–6.5). The average capture probability was 0.50 in 1993 ($N = 7$; range, 0.40–0.59) and 0.56 in 1994 ($N = 7$; range, 0.41–0.63).

California golden trout were very abundant in the study streams. Densities ranged from 1.3 to 2.7 fish/m^2 (370–692 fish/125 m) and biomasses ranged from 15.8 to 21.2 g/m^2 (3,186–6,779 g/125 m). Fish population characteristics were similar between 1993 and 1994; therefore, we present only data from 1994. The outcome of statistical analyses of fish population characteristics depended in part on whether density and biomass were calculated on a unit-area or unit-stream-length basis. When density and biomass were calculated on a unit-area basis, California golden trout density and biomass were significantly ($P \leq 0.05$) higher inside than outside exclosures in three of the four comparisons and not significantly different in one comparison (Figures 4, 5). In contrast, when density and biomass were calculated on a unit-stream length basis, California golden trout density and biomass were significantly higher inside than outside the exclosure in one comparison, significantly lower inside than outside in one comparison, and not significantly different in the remaining two comparisons (Figures 4, 5). At the upper Ramshaw exclosure, the number and weight of California golden trout per square meter were significantly higher inside than below the exclosure (Figures 4A, 5A). The number and weight of fish per 125 m, however, were significantly lower inside than below the exclosure (Figures 4A, 5A). The number and weight of fish per square meter were significantly higher inside than above the lower Ramshaw exclosure, but the number and weight of fish per 125 m did not differ significantly between sites (Figures 4B, 5B). At the Mulkey exclosure, California golden trout densities calculated on a unit-area and unit-stream-length basis both were significantly higher inside the exclosure than below the exclosure, but not significantly different from the densities above the exclosure (Figure 4C). California golden trout biomasses calculated on a unit-area and unit-stream-length basis both were significantly higher inside the exclosure than above the exclosure but not significantly different from those below the exclosure (Figure 5C).

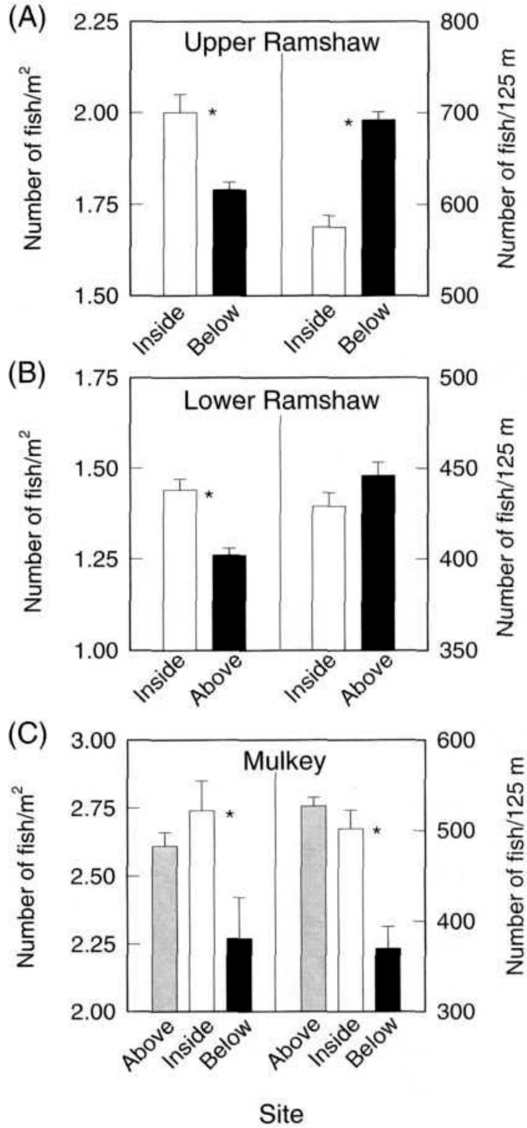

FIGURE 4.—California golden trout population density calculated as number of fish/m^2 (left-side bars) and number of fish/125 m of stream (right-side bars) (A) inside and below the upper Ramshaw exclosure, (B) inside and above the lower Ramshaw exclosure, and (C) below, inside, and above the Mulkey exclosure. An asterisk between paired bars indicates that the difference is statistically significant ($P \leq 0.05$).

Discussion

Our study was hampered by the small number of exclosures available, their nonrandom placement, and the resulting pseudoreplication of our sampling design (Hurlbert 1984). Although we acknowledge that extrapolations from our data to other portions of the study meadows should therefore be made cautiously, the exclosures represent the only possible means of quantitatively assessing current levels of livestock grazing on California golden trout and their habitat at a time when such

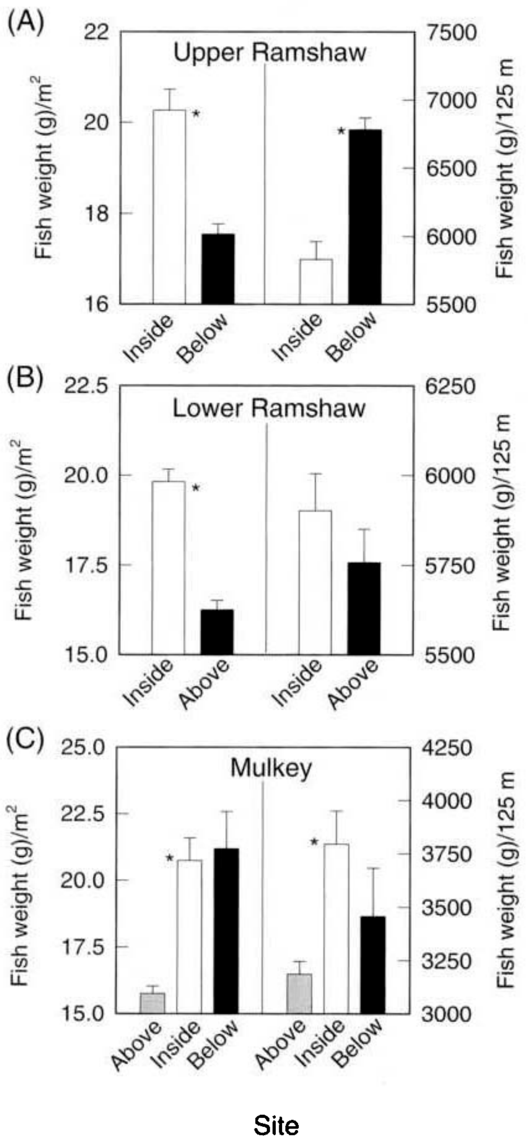

FIGURE 5.—California golden trout population biomass calculated as fish weight (g)/m^2 (left-side bars) and fish weight (g)/125 m of stream (right-side bars) (**A**) inside and below the upper Ramshaw exclosure, (**B**) inside and above the lower Ramshaw exclosure, and (**C**) below, inside, and above the Mulkey exclosure. An asterisk between paired bars indicates that the difference is statistically significant ($P \leq 0.05$).

data are critically needed to aid in determining the status of this subspecies.

Stream Physical Characteristics

Our comparison of stream physical characteristics inside and outside exclosures in Ramshaw and Mulkey meadows strongly suggest that the observed differences are the result of livestock grazing. The consistent differences in numerous physical characteristics between inside and outside sites, including increased streamside vegetation, stream narrowing and deepening, and increased bank stability, are consistent with recovery from grazing-induced damage (Platts 1991). Differences were particularly large at the Mulkey and upper Ramshaw exclosures, where recovery from past grazing is resulting in a more confined, narrow stream (conversion from a type C to a type E channel; Rosgen 1994) lined with abundant mesic and hydric vegetation.

On the basis of the magnitude of differences inside and outside exclosures for each of our measured stream physical characteristics, vegetation apparently responded most rapidly to grazing exclusion, and recovery of streambanks and channel morphology proceeded at a slower rate. Kondolf (1993) reported similar results in a subalpine meadow in the White Mountains, California. Measurements taken inside and outside a 24-year-old exclosure showed significant vegetative recovery, but no detectable recovery in stream channel morphology.

Among the most pronounced differences between protected and unprotected sites were the numbers and sizes of willows. The number of young (5–40-cm) willows was much higher inside than outside all exclosures, suggesting that livestock grazing is impeding willow recruitment. Odion et al. (1988) also found that willow plantings had significantly lower survival outside than inside exclosures in the GTW, and reduction in willow cover appears to be a common result of livestock grazing (Kauffman and Krueger 1984; Platts 1991). Therefore, although willows are currently quite sparse outside grazing exclosures in the GTW, this rarity may be a result of 130 years of livestock grazing and not a natural attribute of these meadows.

Past attempts to establish willows from cuttings have resulted in low survival even inside of GTW grazing exclosures (Odion et al. 1988). However, we observed a large number of young willows inside exclosures. Taken together, these results suggest that the most effective means of establishing willows in the GTW is by natural recruitment following livestock exclusion.

Of the stream physical habitat variables that we measured, several were expected to change only very slowly (20–100 years) or we could not predict in which direction they would change inside versus outside exclosures. These variables were measured to allow post hoc determinations of whether sites inside and outside of exclosures were similar at the time of exclosure construction. Channel depth and channel width both showed consistent differences inside versus outside exclosures; channels were generally more incised and narrower inside exclosures. Although the differences in channel depth and width between some of our grazed and ungrazed sites suggest that some aspects of our sites may have been different at the time of exclosure construction, differences in channel depth and width alone are unlikely to account for the consistent differences in other stream characteristics between inside and outside sites at all exclosures. Instead, these differences are much more likely the result of recovery from livestock grazing inside exclosures.

Changes in stream physical characteristics similar to those found in our study are reported in other studies of grazing impacts on stream and riparian ecosystems. These include increased streamside vegetation (Marcuson 1977; Van Velson 1979; Duff 1983; Platts and Nelson 1985a; Kondolf 1993), stream narrowing and deepening (Duff 1983; Platts and Nelson 1985a), and increased streambank stability (Kauffman et al. 1983; Platts and Nelson 1985b; Rinne 1988).

Fish Population Characteristics

Our comparisons of California golden trout density and biomass show that the magnitude of differences between inside and outside exclosures is influenced by the method used to calculate these variables. When density and biomass were calculated as the number and weight of golden trout per square meter, densities and biomasses were generally significantly higher inside than outside the exclosures. When the density and biomass were expressed as the number and weight per 125 m of stream, however, there were no consistent differences inside versus outside exclosures. The contrasting results obtained when fish density and biomass were calculated based on unit area versus unit stream length are a consequence of the different stream widths (and therefore stream areas) inside and outside exclosures.

Comparisons of fish populations are generally made based on unit-area measurements (e.g., number/m^2, g/m^2, kg/ha; Keller and Burnham 1982; Platts and Nelson 1985b; Beard and Carline 1991; Larscheid and Hubert 1992), but authors have also used unit-volume measurements (e.g., g/m^3; Platts and Nelson 1985a) and unit-stream-length measurements (e.g., number/50 m, g/50 m; Rinne 1988). Because our paired sites were close together and discharges at paired sites should therefore be similar, we found no reason to use unit-volume measurements. Although both unit-area and unit-length measurements are justified in study designs such as ours, unit-area measurements may provide the more accurate portrayal of grazing impacts. Trout population size is frequently limited by the autotrophic food base (Murphy and Hall 1981; Murphy et al. 1981; Hawkins et al. 1983). Because total autotrophic production should increase with increasing stream width but remain constant on a per-area basis, wide stream sections should have a larger total autotrophic food base than narrower stream sections. As a consequence, all other factors being equal, the number of fish per unit stream length should be an increasing function of stream width, whereas the number of fish per unit area should be unaffected by stream width. Therefore, unit-stream-length measurements are potentially confounded by effects associated with stream width, whereas unit-area measurements remove these effects. Based on this reasoning and our results showing that California golden trout density and biomass per square meter were generally greater in ungrazed than grazed sites, we conclude that livestock grazing in the study meadows is having negative effects on California golden trout populations.

Several other studies have also documented the negative consequences of livestock grazing on trout populations (Platts 1991). For example, Marcuson (1977) found that the biomass of brown trout was more than three times higher in an ungrazed stream section than in one that was grazed. Platts (1981) reported that fish densities were more than 10 times higher in a stream section subject to light grazing or no grazing relative to a heavily grazed section. Similarly, densities of rainbow trout *Oncorhynchus mykiss* and brook trout *Salvelinus fontinalis* were higher in an ungrazed than in a grazed stream section (Keller and Burnham 1982). Although these effects are generally attributed to reductions in the quality of physical stream habitat, cumulative effects of grazing can further reduce trout populations by altering stream discharge regimes and by increasing water temperatures (Van Velson 1979; Platts and Nelson 1989; Li et al. 1994).

The results of our fish population surveys show that in spite of the effects of livestock grazing, California golden trout exist at extremely high densities in Mulkey and Ramshaw meadows (1.3–

2.7 fish/m^2). In comparison with salmonid densities in 277 streams reviewed by Platts and McHenry (1988), the California golden trout populations in our study sites were among the densest ever reported for trout in the western United States and were an order of magnitude more dense than the average trout density for all ecoregions in the western United States (0.25 fish/m^2; Platts and McHenry 1988). Because it is unclear whether Platts and McHenry (1988) included age-0 fish in their density estimates, we made the same comparison after removing age-0 fish from our density calculations. Densities of California golden trout in Mulkey and Ramshaw meadows (1.2–2.0 fish/m^2) remained among the greatest in the western United States. Biomass of California golden trout in our study streams (15.8–21.2 g/m^2) is also among the highest recorded and is 3–4 times higher than the average trout biomass of streams in the western United States (5.4 g/m^2; Platts and McHenry 1988).

Livestock Grazing and Wilderness Management

Our study provides evidence that areas in Ramshaw and Mulkey meadows grazed by livestock are in poorer condition than areas inside exclosures and that this degradation is negatively impacting California golden trout. If stream condition in our grazed sites is representative of the condition of streams subjected to grazing throughout the GTW (and we believe it is), our study raises the question of whether such degradation is appropriate in a designated wilderness. Under the Wilderness Act of 1964, "an area of wilderness . . . which is protected and managed so as to preserve its natural conditions and which . . . generally appears to have been affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable" (Kloepfer et al. 1994). Although national forests do not have the authority to curtail livestock grazing solely because of wilderness designation (Kloepfer et al. 1994), they do have the authority to make changes in livestock grazing programs to reduce unacceptable impacts. The Inyo National Forest apparently recognizes the impact of livestock grazing on stream and riparian ecosystems in the GTW, but past efforts to rehabilitate degraded stream habitats in the GTW relied primarily on expensive structural remedies (Laituri et al. 1987; Felando, unpublished report) that have met with very limited success. One of the simplest and most cost-effective means of reducing grazing impacts is to rest areas or reduce livestock numbers (Platts 1991), and we suggest that the Inyo National Forest consider these management options to reduce impacts on stream and riparian ecosystems in the GTW. The restoration of these ecosystems will increase meadow stability (Odion et al. 1988), improve habitat for native California golden trout, and enhance conditions for a wide range of other riparian-dependent species (Johnson et al. 1977; Szaro and Rinne 1988).

Acknowledgments

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Logging Effects on Streamflow: Water Yield and Summer Low Flows at Caspar Creek in Northwestern California

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Streamflow data for a 21-year period were analyzed to determine the effects of selective tractor harvesting of second-growth Douglas fir and redwood forest on the volume, timing, and duration of low flows and annual water yield in northwestern California. The flow response to logging was highly variable. Some of this variability was correlated with antecedent precipitation conditions. Statistically significant increases in streamflow were detected for both the annual period and the low-flow season. Relative increases in water yield were greater for the summer low-flow period than for annual flows, but these summer flow increases generally disappeared within 5 years.

INTRODUCTION

In rain-dominated portions of the Pacific Northwest, annual water yield may be enhanced by the removal of forest vegetation from small upland watersheds. Yet questions and misconceptions linger regarding the effects of logging operations on streamflow under the variety of climatic, physiographic, and vegetative conditions of this region. Timber harvesting impacts have not been fully evaluated for the coastal region of northern California.

Previous studies at the Caspar Creek paired watersheds, near Fort Bragg in northern California, investigated the impacts of selected harvest of a second-growth Douglas fir and redwood forest on peak streamflow [*Ziemer,* 1981; *Wright,* 1985], hydrograph lag time [*Sendek,* 1985], and sediment production [*Rice, et al.,* 1979]. However, the effect of logging and related factors upon summer low-flow quantity and timing at Caspar Creek was not evaluated.

Vegetation affects the proportion of precipitation that is evaporated and transpired and, consequently, the amount available for soil moisture storage, groundwater recharge, and dry weather streamflow. The proportionate contribution of precipitation to streamflow varies by the manner in which interception and evapotranspiration are influenced by vegetation type, development, rooting depth, and health.

Research on upland watersheds indicates that water yield can be augmented by vegetation removal [*Ponce and Meiman,* 1983]. However, responses to treatment are highly variable and depend on the particular watershed system studied [*Hewlett and Hibbert,* 1967].

Logging operations alter the conditions and processes involved in the generation of streamflow. Most notably, evapotranspiration is reduced by the removal of forest vegetation. Also, soil characteristics are inevitably modified by the construction of roads, landings, and skid trails that accompanies timber harvesting, particularly when ground skidding is used [*Stone,* 1977]. Localized soil disturbances associated with this construction include reduced infiltration capacity, increased bulk density, and a conversion of soil macropores to micropores. In addition, soil drainage patterns may be altered. Although the impacts of road construction and tractor logging on soil surfaces have been documented by substantial research, the effects of these activities on the generation of streamflow are not fully understood [*Sendek,* 1985].

In reviewing the results of catchment studies at 11 locations in western Oregon and western Washington, *Harr* [1979] reported annual water yields that increased as much as 62 cm following timber harvest, while summer low flows as much as quadrupled. This difference implies reduced evapotranspiration and greater soil moisture levels on the logged basins. These increased water yields diminished with revegetation, with annual flows returning to pretreatment levels within 4-5 years.

A time duration model predicts streamflow increase for a regrowing eastern forest [*Douglass and Swank,* 1972]:

$$Q_i = a + b\ (\log\ T_i)$$

where Q_i is the increase in flow year *i, a* is the first-year increase, T_i is the *i*th year after treatment, and *b* is a negative coefficient. A similar relationship could hold in other regions. As a gross index of revegetation and renewed interception and evapotranspiration losses, time since logging has been identified as the most important variable in explaining water yield increases in the Pacific Northwest region [*Harr,* 1979].

Harvest Practices

Hibbert [1967] reported increases in streamflow proportional to the amount of cover removed. Partial cutting is less effective than clearcutting at augmenting streamflow [*Rothacher,* 1971]; partial cutting may actually enhance water use by the trees and understory vegetation that remain [*Kittredge,* 1948]. *Greenwood et al.* [1985] concluded that reduced evapotranspiration from overstory vegetation following clearing may be strongly countered by increased evapotranspiration from the understory due to increased availability of energy and soil water.

Site Conditions

Site conditions are an important factor influencing streamflow response to the removal of vegetation. Streamflow increases in watersheds with soil and topographic conditions favorable for tree growth are smaller and diminish faster than those in watersheds with less favorable conditions,

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owing to the prompt recovery of the forest [*Nakano,* 1967]. Water yield increases in the Pacific Northwest are short-lived because of favorable conditions that support rapid regrowth of forest and other vegetation [*Harr,* 1983]. At the H.J. Andrews Experimental Forest east of Eugene, Oregon, annual water yield increases were similar after clearcutting 130 and 450-year-old Douglas fir forests [*Harr et al.,* 1982].

Precipitation

Annual precipitation influences the magnitude of water yield increases that follow timber harvest operations in forested watersheds. Greater increases are found in wetter years [*Harr,* 1979; *Ponce and Meiman,*1983]. *Bosch and Hewlett* [1982] suggest that streamflow response also depends on the mean annual precipitation of the area. Increases are generally greatest in areas of high rainfall, but they are short-lived due to rapid regrowth of vegetation. According to *Bosch and Hewlett* [1982], actual precipitation is influential only in low-rainfall areas, In high-rainfall areas they found water yield changes as a result of treatment were independent of actual precipitation. This view contradicts those of *Harr* [1979] and *Ponce and Meiman* [1983].

Season

As an indicator of potential evapotranspiration, season is an important variable that affects the streamflow response to logging. Seasonal analyses of yield increases on experimental forests in western Oregon by *Rothacher* [1970], *Harr et al.* [1979], and *Ingwersen* [1985] indicate that most of the increases in annual water yield in response to logging occurs in the October-March rainy season. Logging reduces transpiration during the growing season, as well as interception losses. Soil on logged watersheds has a relatively high moisture content at the onset of the rainy season in comparison to uncut watersheds, requiring less rainfall to recharge soil moisture levels, thus allowing more precipitation to become available for streamflow. *Ziemer's* [1981] analysis of peak flows on the Caspar Creek watershed supports this explanation. *Douglass and Swank* [1975] presented the same explanation in their study of eastern forest watershed responses to deforestation, but the timing of yield increases was different. Relative to prelogging summer flow patterns, logging-related streamflow increases in the eastern forests were negligible until June, increased as the growing season advanced, and peaked in September. During the growing season, the East Coast is wetter than the West Coast.

In the Pacific Northwest the greatest relative increases in streamflow have been observed during the summer season, although in absolute terms, larger increases have occurred during the rainy season. These summer increases are short-lived, however, lasting only 2-3 years [*Harr,* 1979]. The number of low-flow days (where streamflow has fallen below some preset threshold value) was used to evaluate flow changes in the Alsea Watershed Study in the Oregon Coast Range; fewer low-flow days were found after logging [*Harr and Krygier,* 1972].

Fog Interception Processes

An important contradiction to the pattern of increased flows after logging was observed in the Fox Creek watershed study within Portland's Bull Run municipal watershed, where a small decrease in annual water yield was noted. After timber harvest, the number of low-flow days increased, suggesting summer flows were actually reduced as a result of logging. *Harr* [1980] hypothesized that this anomaly was the result of reduced fog drip interception after clearing the forest. In a subsequent study, as much as 44% more net precipitation was measured in late spring and summer beneath the forest canopy than in a clearing. During two fall seasons, differences of 18 and 22% were observed [*Harr,* 1982]. Within the forest, fog drip accounted for roughly one third of all precipitation for the May-September period. *Harr* concluded that in addition to offsetting canopy interception and evaporation losses, fog drip at this site may have provided about 50 cm additional water to the forest floor.

Subsequent analysis of recent streamflow data from the Fox Creek experimental watershed indicates that a recovery has occurred from the harvest impacts on summer water yield due to loss of fog drip [*Ingwersen,* 1985]. These results suggest that by the elimination of fog drip through the removal of forest vegetation, anticipated enhancement of summer flows may not be realized in areas where fog occurrence is a frequent source of significant moisture. The occurrence of fog and its role in influencing moisture conditions in coastal California and Oregon has been well documented, lending support to the hypothesis that significant amounts of moisture can be delivered in areas with a high frequency of advected fog [*Byers,* 1953; *Oberlander,* 1956; *Azevedo and Morgan,* 1974; *Goodman,* 1985]. Summer fog is common in the Caspar Creek watersheds.

This paper analyzes streamflow data at Caspar Creek in northwestern California over a 21 -year period to determine the effects of logging and related factors upon low flows and annual water yield.

STUDY AREA AND TREATMENTS

The study watersheds (North and South Forks of Caspar Creek) are located in the Jackson Demonstration State Forest, 11 km southeast of Fort Bragg, California, and about 7 km from the Pacific Ocean (Figure 1). The North and South Forks of Caspar Creek drain watersheds having areas of 483 and 424 ha, respectively. The elevation of the watersheds ranges from 37 to 320 m. Topography of the North and South Fork watersheds runs from broad, rounded ridge tops to steep inner gorges. The median side slope of the watersheds is 20". Watershed soils formed in residuum derived predominantly from sandstone and weathered coarse-grained shale of Cretaceous age. Soils are well drained, having high saturated and unsaturated hydraulic conductivities [*Wosika,* 1981]. The climate is Mediterranean, having dry summers with coastal fog. Summer temperatures are mild, ranging from 10° to 25°C. Winters are mild and wet, with temperatures ranging from between 5° and 14°C and a rainfall average of about 1200 mm per year [*Ziemer,* 1981]. Caspar Creek does not receive any appreciable snowfall.

The North and South Forks of Caspar Creek were originally clearcut logged and burned in the late 1800s, the North Fork about 15 years after the South Fork [*Tilley and Rice,* 1977]. Since then, fairly dense stands of second-growth redwood *(Sequoia sempervirens* (D. Don) Endl.) and Douglas fir *(Pseudotsuga menziesii* (Mirb.) Franco) developed, with some associated western hemlock *(Tsuga heterophylla*

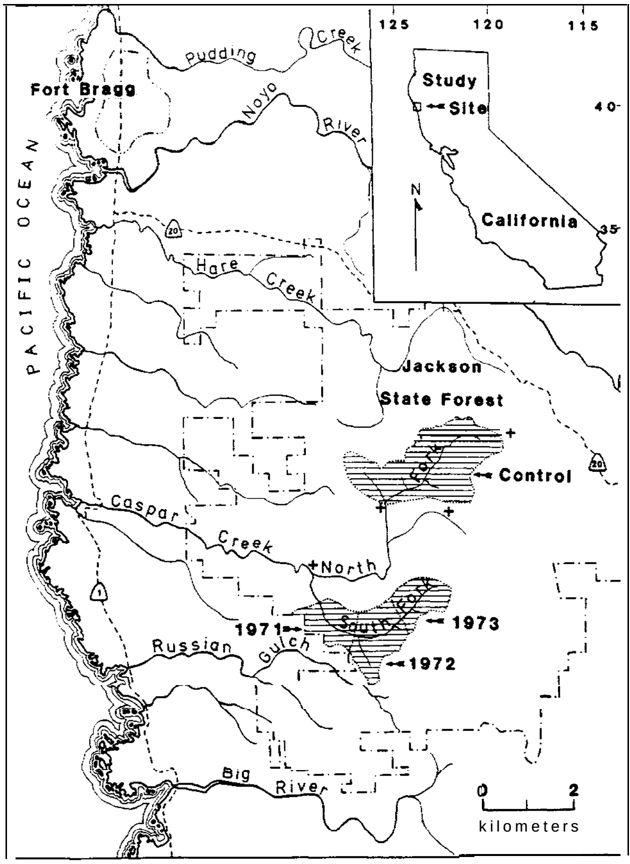

Fig. 1. Caspar Creek experimental watersheds: North Fork (control) and South Fork (areas logged each year are indicated).

(Raf.) Sarg.) and grand fir (*Abies grandis* (Dougl.) Lindl.). At the onset of the study, timber volume of both watersheds was estimated at about 700 $m^3 ha^{-1}$ [*Krammes and Burns,* 1973]. The North Fork was selected as the control watershed because its timber was younger. Road location and construction, and timber harvest practices in the South Fork were designed to meet standards that were considered "state of the art" but also considered commercially acceptable by the local timber contractors.

Construction of stream gauging stations on both forks was completed in the summer of 1962. Flow was measured with compound weirs consisting of a 6.1 m x 0.91 m rectangular sharp-crested weir superimposed upon a 0.61 m 120° v notch weir. Precipitation was estimated from four weighing rain gauges.

Both watersheds were monitored in an undisturbed condition during hydrologic years (October-September) 1963-1967. Road construction in the summer of 1967 was monitored through hydrologic year 1971 when logging began. Logging effects were followed through hydrologic year 1983.

Of the total 6.8 km of road constructed in the South Fork watershed during the summer of 1967, 6 km were within 61 m of the stream. Coarse debris, resulting largely from right-of-way clearing, was removed from the stream and from along the stream banks after road construction [*Krammes and Burns,* 1973]. The roads (including cut and fill slopes) occupied 19 ha (4.5% of the total watershed area) from which 993 $m^3 ha^{-1}$ of timber was removed.

About 110 m of stream bed were disturbed by tractor operation directly in the stream. These areas were primarily around bridge crossings, landings, and in a stretch where the stream was cleared of debris deposited during the road construction. All fill slopes, landings, and major areas of soil exposed by the road building were fertilized and seeded with annual ryegrass in September 1967. The grass was well established before the first rains in November [*Jackman and Stoneman,* 1973].

Logging, which began on the South Fork of Caspar Creek during the summer of 1971, continued over a three-year period. The South Fork watershed was divided into three sale areas. Selective cutting started at the weir and progressed up the watershed on successive years. All logging was done by tractor, but many of the skid trails did not have adequate cross drains installed. By the completion of logging over 15% of the watershed was in roads, landings, and skid trails and considered heavily compacted.

The area of the first sale was 101 ha from which 59% of the stand volume was harvested. The following summer, 69% of the volume was taken from a 128-ha area. During the final summer, 65% of the volume was cut on the remaining 176 ha of the South Fork watershed. In aggregate, road construction and harvesting removed 67% of the timber volume (nearly 200,000 m^3). No cultural measures were taken to foster regeneration. Consequently, most of the regrowth was either true fir seedlings or redwood sprouts. The residual stand did not respond vigorously to the reduced competition resulting from logging.

DATA ANALYSIS

In the present study the low-flow season was defined as the part of each year when the flow response from Caspar Creek was predominated by base flow rather than storm flow or quick flow processes. The great variability in the arrival and cessation of the rainy season each year precludes the use of a constant starting and ending date for the low-flow season throughout the study period. Defining the starting and ending dates of the low-flow season in terms of a moisture index is more meaningful than using arbitrary preset dates.

The source of summer flow was groundwater and soil moisture storage. Lacking actual records of this storage component, an indirect measure of ground and soil water levels was needed.

Soil moisture and runoff levels were indexed by an antecedent precipitation index (API), using precipitation data and the exponential law of decay [*Ziemer,* 1984]. The antecedent precipitation index is defined as

$$\mathrm{API}_i = K * \mathrm{API}_{i-1} + P_i$$

where API_i, is the index value, P_i is the precipitation occurring on the ith day of the calculation, and K is the recession factor ($K \leq 1$). A recession factor of 0.97 was found to satisfactorily predict hydrograph response at Caspar Creek.

The low-flow season began and ended when the daily API fell below and exceeded, respectively, a threshold value of 10 cm. From this definition, each "API year" in the study period began at the end of the preceding low-flow season and continued through the final day of the current low-flow season. The starting dates of the low-flow season were found to correspond with a North Fork flow of approximately 28 L s^{-1}.

TABLE 1. Streamflow Variables (Dependent Variables)

Name	Definition
	Basic Variables
SUMVOL	the total flow volume for the low-flow period, defined in terms of an antecedent precipitation index or roughly corresponding to that period when the North Fork mean daily flow was less than 28 L s^{-1}, 1000 m^3
TOTVOL	the total flow volume for the low-flow season and the preceding rainy season, i.e., the total flow volume for the API year, 1000 m3
PARTVOL	the ratio of total summer flow volume to total API year flow volume, i.e., SUMVOL/TOTVOL
LOFLOZ	the number of days during the low-flow season when the mean daily flow rate is less than 5.66 L s^{-1}, days
START	the mean daily flow rate on the first day of the low-flow season, L s^{-1}
END	the mean daily flow rate on the final day of the low-flow period, L s^{-1}
	Ratio Variables
SUMVOLSN	ratio of difference between seasonal flow volume (SUMVOL) on the South and North Forks ((SF-NF)/NF)
TOTVOLSN	ratio of difference between annual flow volume (TOTVOL) on the South and North Forks ((SF-NF)/NF)
PARTVOLSN	ratio of difference between proportionate seasonal flow volume (PARTVOL) on the South and North Forks ((SF-NF)/NF)
LOFLOZSN	ratio of difference between the number of “low-flow days” for the season (LOFLOZ) on the South and North Forks ((SF-NF)/NF)
STARTSN	ratio of difference between the start-of-season rate of flow (START) on the South and North Forks ((SF-NF)/NF)
ENDSN	ratio of difference between the end-of-season rate of flow (END) on the South and North Forks ((SF-NF)/NF)

Twelve dependent variables were developed to evaluate the streamflow process and changes after logging: six basic variables and six difference ratio variables (Table 1). The data were divided into two classes for analysis: prelogging (1963-1970) and postlogging (1971-1983). The postroad construction years (1967-1970) were included in the prelogging group. Previous Caspar Creek studies found that road construction activities did not alter the hydrologic response of the basin during winter to a statistically detectable extent [Ziemer, 1981; *Sendek,* 1985].

For each of the six basic streamflow variables a simple linear regression model was developed for the prelogging calibration period and the posttreatment period, respectively. Ninety-five percent prediction limits [*Neter et al.,* 1983] were calculated to determine if the postharvest responses were within the range predicted by the calibration relationships. This calculation was done both to test the magnitude of the impacts of the logging operations on streamflow and to determine the duration of statistically significant changes in the summer flow response at Caspar Creek. All statistical tests were performed at the 0.05 significance level.

Multiple regression analysis was used to identify which management and climatic variables might be most influential in affecting the extent and duration of changes in summer flow processes at Caspar Creek. To study the relative change between the two watersheds, a difference ratio was chosen for the dependent variables (Table 1).

A large set of potential independent variables was organized into four categories: logging, precipitation, antecedent precipitation, and general climatic norms. From this set, 15 of the most promising variables were selected for further analysis (Table 2).

An all-possible-subsets regression procedure was used to examine possible regression models and identify “good” models. No single statistical criteria was solely relied upon in choosing the “best” model. Test criteria included Mallow’s C_p [*Daniel and Wood,* 1971], the adjusted coefficient of multiple determination R_a^2, the overall F test for the existence of a regression relation, the partial F test for the marginal reduction in variance associated with each additional variable, and graphical analysis of residuals [*Neter et al.,* 1983]. The correlation matrix derived earlier was also used to check for interdependencies among included independent variables. In addition, the signs of the regression coefficients were considered in relation to the simple correlation coefficient of that variable to the dependent streamflow variable and in relation to the expected directional influence of that variable. By this both objective and subjective process the preferred descriptive regression model was chosen for each of the streamflow ratio variables examined.

RESULTS

Effects of Logging on Streamflow Parameters

The simple linear regressions for the calibration period of the six South Fork basic streamflow variables on those of the North Fork yielded significant relationships at the 0.005 significance level or smaller (Table 3).

For the postlogging period the relationships between the North Fork and South Fork streamflow variables were more variable. Five of these regressions were significant at the 0.025 significance level or smaller (Table 3). No significant relationship could be detected between the North and South Forks for the number of low-flow days (LOFLOZ) after logging.

Posttreatment South Fork observations that fell outside of the prediction limits calculated for the pretreatment calibration regression (at 0.05 significance level) were judged to be significantly different than the expected value (Figures 2-7). While a lack of consistently significant alterations of the streamflow response is apparent, the figures provide some evidence of enhanced streamflow beginning in 1972.

The total annual flow volume (TOTVOL) measured at the South Fork weir (Figure 2) increased during the period from 1973 to 1982, ranging from an additional 7 to 34% of the expected flow. In absolute terms this increase in volume amounted to an additional 2.3 x 10^5 to 9.9 x 10^5 m^3 of water each year (an average of 4 x 10^5 m^3 per year or a 15% increase).

The volume of streamflow recorded at the South Fork weir during the low-flow (summer) season (SUMVOL) ranged from 14 to 55% greater than the predicted value for the

TABLE 2. Independent Variables Developed for Use in the Multiple Regression Analysis

Name	Definition
	Logging Variables
%AC	percent of watershed area compacted; includes road, landing, and skid trail areas
%ADF2	cumulative percent of area harvested by selection method, employing an exponential decay function to model recovery of vegetation water use with time after logging; half-life of recovery estimated at 3.5 years
	Precipitation Variables
PPTSEAS	total measured precipitation for the low-flow season, cm
TRD/TD	number of days with recorded precipitation during the API year divided by the length of the API year in days*
SRD/SD	number of days with recorded precipitation during the low-flow season divided by the length of the low-flow season in days
SRD/TRD	number of days with recorded precipitation during the low-flow season divided by the total number of days with recorded precipitation
	Antecedent Precipitation Variables
LENYR	length of the API year, days
LENSEAS	length of the low-flow season, days
APIYR	cumulative daily antecedent precipitation index for the API year, cm
APISEAS	cumulative daily antecedent precipitation index for the low-flow season, cm
MINAPI	minimum one-day antecedent precipitation index for the low-flow season (and API year), cm
MAXAPI	maximum one-day antecedent precipitation index for the API year, cm
PREAPI	cumulative daily antecedent precipitation index for the preceding API year, cm
	Solar Radiation Variables
HRLIT	estimated total possible daylight for the low-flow season based on times of sunrise and sunset at 39°N latitude, hours†
MLYDAY	estimated normal mean daily incoming radiation for the low-flow season. Total estimated normal incoming radiation for the low-flow season divided by the length of the low-flow season, langleys/day

*The API year begins at the end of the preceding low-flow period and continues through the final day of the current low-flow period. Thus it includes the "winter" or "rainy" season as well as the low-flow season.
†*U.S. Naval Observatory* [1946].

period 1972-1978. The increases during the 1972, 1974, 1975, and 1978 seasons were statistically significant (Figure 3). The greatest percent increase occurred in 1978, although in absolute terms the largest increase, 9 x $10^4 m^3$, occurred in 1974. For the 7-year postlogging period from 1972 through 1978, SUMVOL increased an average of 29%, or an additional 5 x $10^5 m^3$ per low-flow season, compared to the prelogging period. During the 1981 season a statistically significant decrease, 27% in summer flow was detected. A 19% decrease was observed in 1983, but this was not found to be significant.

To investigate change in the seasonal distribution of streamflow volume on the South Fork following logging, the variable PARTVOL (SUMVOL/TOTVOL) was analyzed (Figure 4). The proportionate summer flow volume relative to annual flow volume exceeded the predicted value for the years 1972-1975 and 1978, with 1972 being significantly greater. For the years 1976 and 1979-1983 the observed value fell below the predicted value, with 1981 being significantly lower.

The number of low-flow days (LOFLOZ), days with mean daily flow rates of less than 5.66 L s^{-1}, was consistently fewer than predicted following logging (Figure 5). From 1972 through 1978, LOFLOZ averaged 43 fewer days than predicted, a 40% decrease. Between 1979 and 1983 the number of low-flow days returned to that observed before logging.

The rate of flow at the onset of the low-flow season (START) generally increased after logging the South Fork, with the increases significant for the years 1973-1976, 1982, and 1983 (Figure 6). The maximum increase occurred in 1974 when the flow rate was 46% (13 L s^{-1}) higher than predicted. From 1973 to 1983 the observed rate of flow averaged 25% above the predicted rate.

Beginning in 1973 and continuing through the end of the study period, the observed end-of-season flow rate (END) was greater than the predicted value, but only for the years

TABLE 3. Calibration and Posttreatment Regression Coefficients

	Calibration			Posttreatment		
Variable	b_0	b_1	F	b_0	b_1	F
TOTVOL	120.70	0.85	287*	423.20	0.85	323*
SUMVOL	-39.43	1.51	64*	11.20	1.25	8‡
PARTVOL	0.01	1.12	31†	0.02	0.94	21†
LOFLOZ	-43.20	1.22	63*	10.34	0.57	3 ns
START	8.11	0.53	50*	4.08	0.83	36"
END	-8.02	1.61	1032*	-39.98	3.09	48*

Regression not statistically significant at the minimum 0.050 significance level, ns.
*Regression significant at the 0.001 significance level.
†Regression significant at the 0.005 significance level.
*Regression significant at the 0.025 significance level.

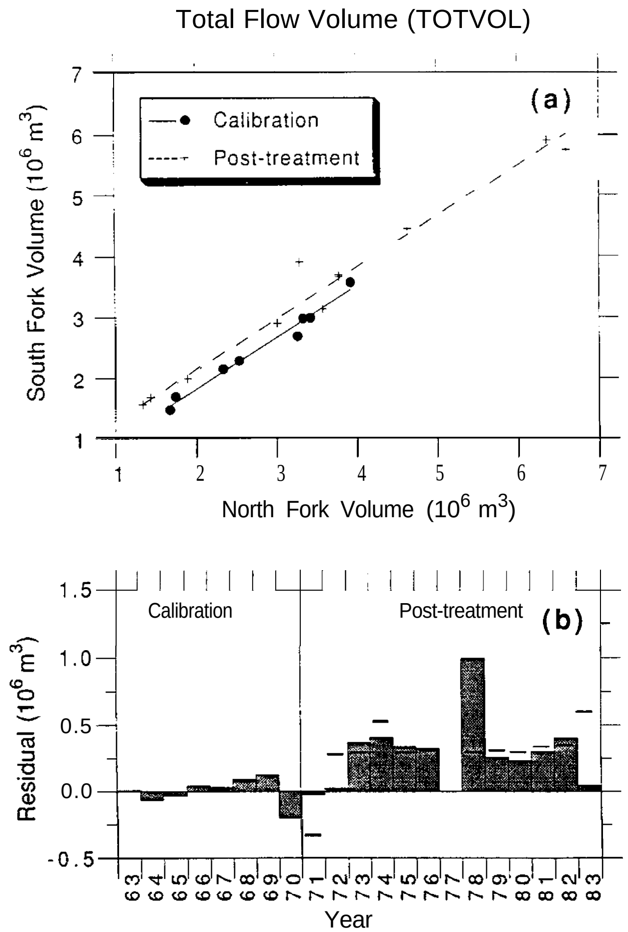

Fig. 2. Results of linear regression for North Fork and South Fork total annual flow volumes: the low-flow season plus the preceding rainy season (TOTVOL). *(a)* Calibration and posttreatment periods. *(b)* Residuals for calibration period and deviations from calibration regression for posttreatment period. Horizontal bars indicate the 95% prediction limits for posttreatment years.

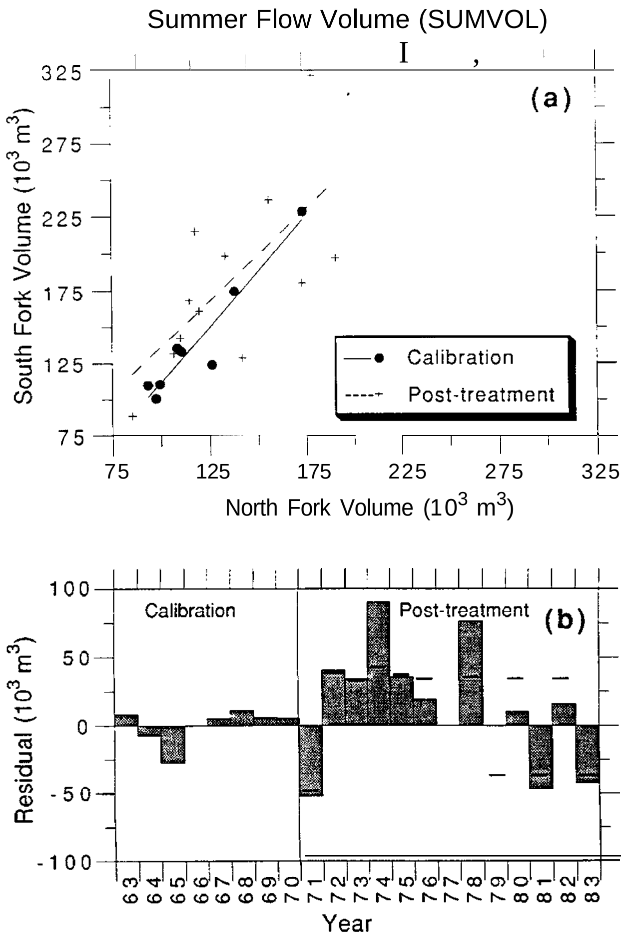

Fig. 3. Results of linear regression for North Fork and South Fork total low-flow season volumes (SUMVOL). *(a)* Calibration and posttreatment periods. *(b)* Residuals for calibration period and deviations from calibration regression for posttreatment period. Horizontal bars indicate the 95% prediction limits for posttreatment years.

1974, 1975, 1977, and 1978 was this increase significant at the 0.05 significance level (Figure 7). The final day of the season occurs when the daily API again exceeds the low-flow season API threshold, that is, substantial precipitation triggers the end of the low-flow season. The increase averaged 87% and ranged from 6% (1979) to 178% (1976). In absolute terms the maximum increase occurred in 1974 when the observed flow rate was 382 L s^{-1} greater than predicted by the prelogging calibration regression. During other years this increase varied from 2 L s^{-1} in 1979 to 250 L s^{-1} in 1978 and averaged 63 L s^{-1}.

Factors Associated With Variations in the Streamflow Response

Multiple linear regression analysis was used to further examine the alteration of the streamflow pattern after timber harvest and to identify factors that were significant in determining differences between the South Fork and North Fork streamflow response. The 0.05 significance level was used for testing the significance of each additional variable (Table 4).

The logging variable, cumulative percent of area logged (%ADF2), was the most significant independent variable associated with five of the six dependent variables. For the sixth dependent variable, annual flow volume differences (TOTVOLSN), the percent of the South Fork watershed area compacted by roads, landings, and skid trails (%AC) was the most significant independent variable. These variables indicated that an enhancement of the South Fork flow was associated with forest harvesting operations.

The antecedent precipitation variables significantly improved the prediction of relative differences between the two streams. High antecedent moisture conditions preceding (PREAPI) and during (APIYR) the hydrologic year were related to an increase in the South Fork flow relative to the North Fork.

When seasonal precipitation (PPTSEAS) during the low-flow season was high, the South Fork flow level was enhanced relative to the North Fork. However, as the proportion of the season's days having recorded precipitation (SRD/SD) increased, which may be viewed as an index of cloud cover, the two streams responded more similarly.

Roads, Landings, and Skid Trails

By the completion of timber harvest operations in 1973, 15% of the South Fork watershed was occupied by roads, landings, and skid trails.

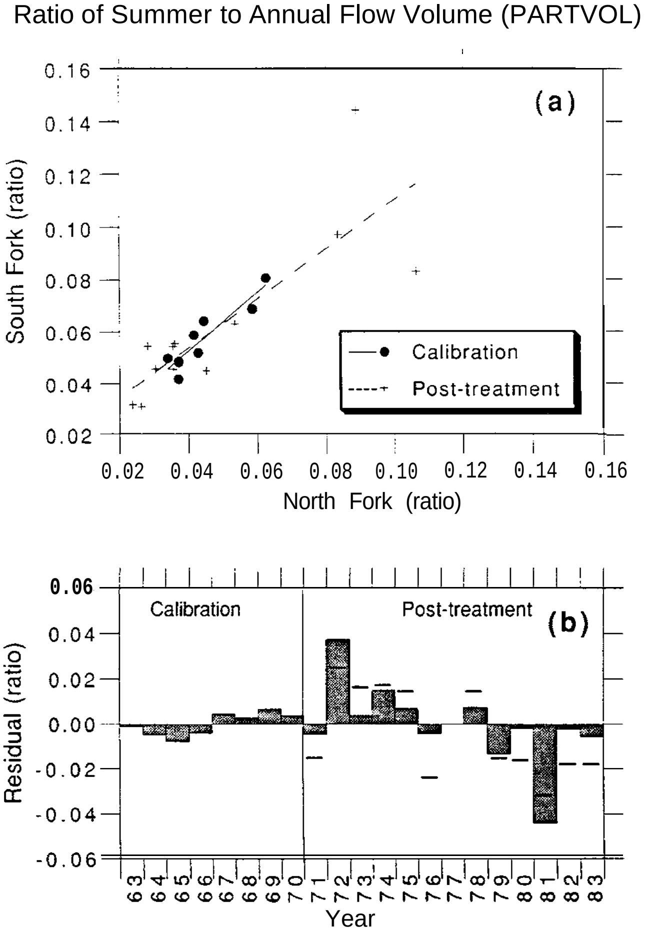

Fig. 4. Results of linear regression for North Fork and South Fork ratios of total low-flow season flow volume to total annual flow volume (PARTVOL). *(a)* Calibration and posttreatment periods. *(b)* Residuals for calibration period and deviations from calibration regression for posttreatment period. Horizontal bars indicate the 95% prediction limits for posttreatment years.

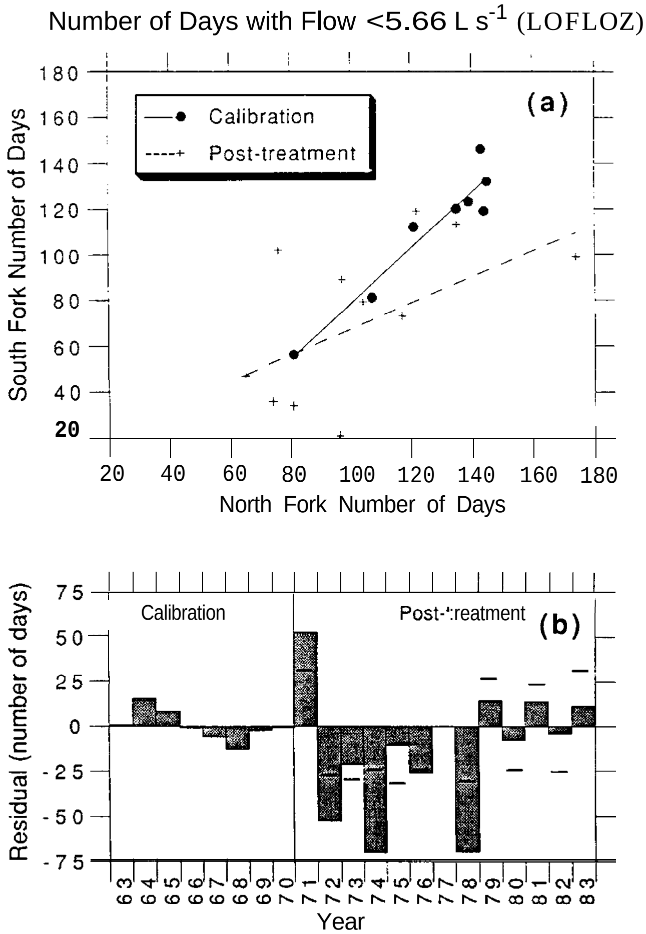

Fig. 5. Results of linear regression for North Fork and South Fork number of low-flow days when the mean daily flow was less than 5.66 L s^{-1} (LOFLOZ). *(a)* Calibration and posttreatment periods. *(b)* Residuals for calibration period and deviations from calibration regression for posttreatment period. Horizontal bars indicate the 95% prediction limits for posttreatment years.

High rates of infiltration, typical of forest soils in the coastal Pacific Northwest, generally preclude the occurrence of overland flow except for areas of bare rock or extremely shallow soil and intermittent channels [*Harr*, 1979]. This condition seems to be characteristic of the Caspar Creek site. Finding no consistent increase in winter storm peak flows, *Ziemer* [1981] reasoned that precipitation continued to infiltrate and supply subsurface flow and that compaction from the construction of the transportation network at Caspar Creek did not result in significantly reduced infiltration for the overall watershed.

Subsequently, *Sendek* [1985] reported evidence that streamflow response to precipitation at Caspar Creek became quicker and more efficient after logging. We found that an increase in annual flow volume, which averaged 15% above the predicted volume, was correlated with the percent of the watershed area converted to roads, landings, and skid trails. Variables representing removal of forest vegetation were modelled to decrease as a function of time since logging. If the increase in rainy season flow volume had been closely associated with a decrease in evapotranspiration as a result of the reduction in forest vegetation, one of the alternate logging factors that represented such forest influences should have correlated more highly with this change. A probable explanation is that the increase in annual flow volume associated with the winter season was mainly the result of a reduction in interception losses from the roads, landings, and skid trails accompanied by a minimal reduction in soil moisture storage.

Alteration of Forest Vegetation

Timber harvest operations selectively removed 67% of the South Fork timber volume between 1971 and 1973. Removal of forest vegetation reduces evapotranspiration and canopy interception losses, thereby increasing soil moisture storage. During the growing season, substantial differences in soil moisture can develop between a logged and unlogged watershed. Although lower evapotranspiration rates characterize the winter period, it is possible for an "interstorm" difference in soil moisture to develop between a logged and an unlogged watershed during the winter. Such dissimilarity may be indicated by differences in base flow recession. The enhancement of streamflow at Caspar Creek can be explained in light of these principles.

Proportionately larger increases in mean daily flow rate relative to prelogging predicted rates occurred from the first day of the low-flow season to the final day of the low-flow

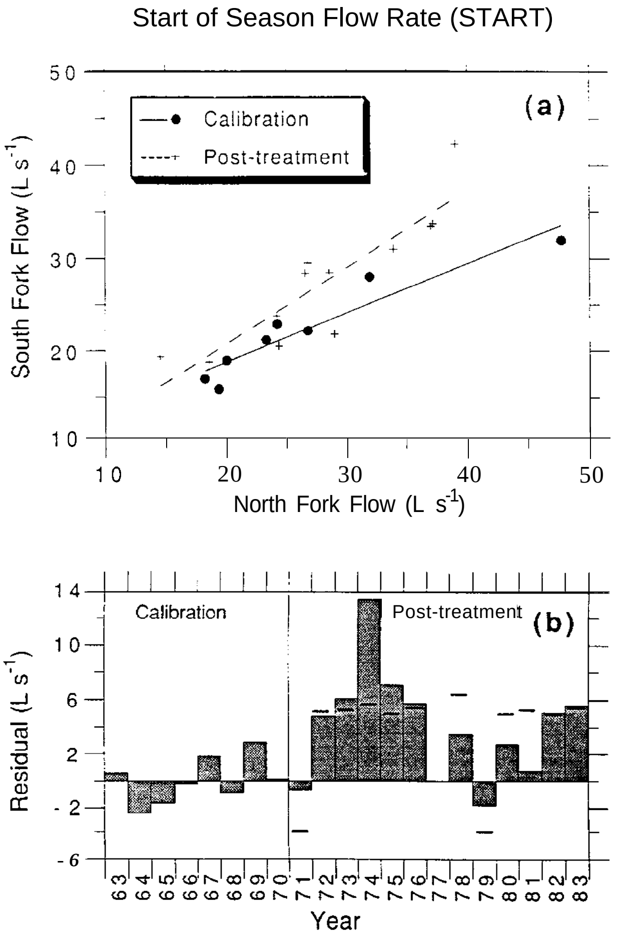

Fig. 6. Results of linear regression for North Fork and South Fork mean daily flow on the first day of the low-flow season (START). *(a)* Calibration and posttreatment periods. *(b)* Residuals for calibration period and deviations from calibration regression for posttreatment period. Horizontal bars indicate the 95% prediction limits for posttreatment years.

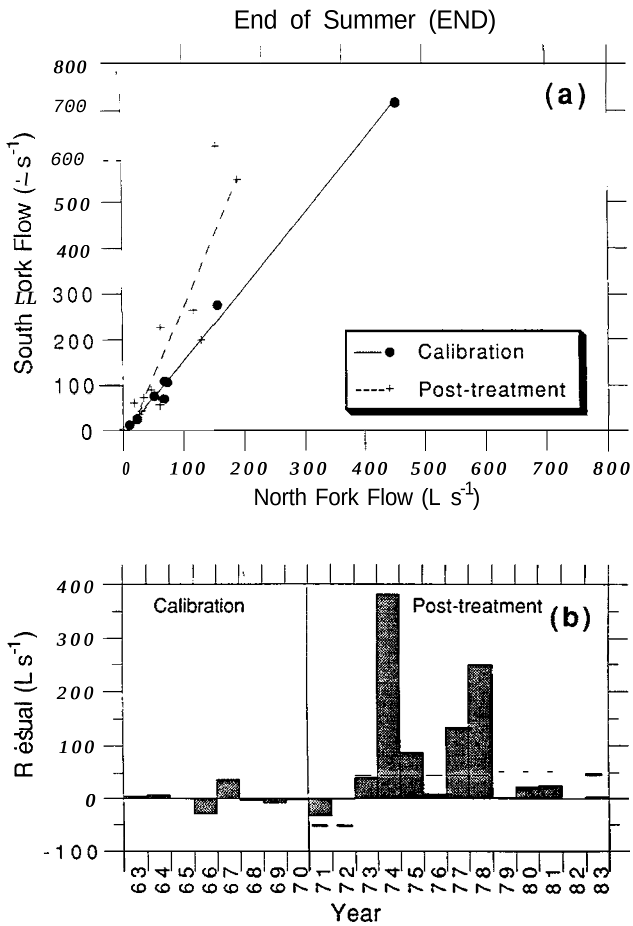

Fig. 7. Results of linear regression for North Fork and South Fork mean daily flow on the final day of the low-flow season (END). *(a)* Calibration and posttreatment periods. *(b)* Residuals for calibration period and deviations from calibration regression for posttreatment period. Horizontal bars indicate the 95% prediction limits for posttreatment years.

season. This trend suggests that soil moisture differences were developing between the two Caspar Creek watersheds as the growing season progressed.

Canopy interception is determined by total leaf area and the amount and intensity of precipitation. During a major winter storm the proportion of intercepted precipitation is probably inconsequential. However, during a light rain or fog, a considerable proportion of the total can be intercepted by a dense coniferous forest [*Dunne and Leopold,* 1978]. Along California's north coast, numerous small storms occur, each separated by a prolonged rainless interval. During such low-intensity precipitation events, interception loss on the North Fork was probably substantially greater than that on the South Fork after logging, and differences in interception between the two watersheds may have contributed to the increase in water yields detected in the logged area.

As seasonal precipitation (PPTSEAS) increased, summer flows on the South Fork increased significantly in comparison to those on the North Fork. This difference suggests that a greater proportion of summer precipitation reached the soil surface and was available to offset evapotranspiration by the remaining vegetation on the logged watershed. On the control watershed, more of this precipitation was probably intercepted and evaporated directly from leaf surfaces.

Regrowth

After 1978, increases in South Fork flow were detected only for the variables directly related to winter streamflow processes and meteorological conditions (TOTVOL and START). A significant increase was not detected for those variables reflecting summer conditions. In 1981 and 1983 there was a significant decrease in total summer season flow (SUMVOL).

The removal of mature timber by selective harvest operations was designed to improve the growth potential of the younger trees. By creating openings in the canopy and reducing competition for sunlight and water, the growth and water use of the remaining vegetation and invading trees, brush, and forbs may have been accelerated, as has been documented in other forest environments [*Bogatyrev and Vasil'eva,* 1985; *Jarvis,* 1985; *Greenwood et al.,* 1985]. This mechanism would explain the rapid reduction of summer flow enhancements and the possibility of decreased summer flows after growth has been stimulated in the remaining vegetation.

Antecedent Moisture Influences

Multiple regression analysis indicated that antecedent moisture conditions, represented by antecedent precipita-

TABLE 4. Standardized Coefficients and Statistics for Six Multiple Regression Models

Independent Variables and Statistics	Dependent Variables					
	TOTVOLSN	SUMVOLSN	PARTVOLSN	LOFLOZSN	STARTSN	ENDSN
%ADF2		0.886	0.999	-0.562	0.651	0.803
%AC	0.821		-0.822			
PREAPI	-0.416	-0.334			-0.390	-0.446
APIYR	-0.630		0.527			
APISEAS						0.215
PPTSEAS		0.350		-0.460		
SRD/SD				0.591		
R^2	0.766	0.692	0.608	0.476	0.463	0.847
Overall *F*	16.3*	11.2*	8.3†	8.9†	6.9‡	27.7*
Standard Error	0.055	0.157	0.169	0.180	0.113	0.372

*Regression significant at the 0.001 significance level.
†Regression significant at the 0.005 significance level.
‡Regression significant at the 0.025 significance level.

tion (API), significantly influenced the magnitude of flow differences between the two Caspar Creek watersheds (Table 4). During years with high cumulative API values or preceded by years with high values, the logging effects on streamflow were reduced. A logical explanation is that during these wetter years, soil moisture deficits are small for much of the year, and thus both basins show similar summer recession characteristics. In contrast, during drier years, extensive differences in soil moisture may develop between the basins, owing to reduced evapotranspiration on the logged watershed.

The cumulative API for the preceding year (PREAPI) was a significant variable in four of the six regressions (Table 4). This suggests that the carryover effect of past antecedent moisture regimes may be substantial. The adequacy of soil moisture during critical growth periods can influence subsequent transpiration and growth rates depending on nutrient conditions and other growth requirements, but the extent of this effect is unclear [*Russell,* 1973]. The carryover effect of the antecedent moisture conditions of the preceding year possibly is an indirect reflection of variations in vegetation vigor and overall efficiency of water use rather than a direct indication of persistent soil moisture differences.

Additional Climatic Factors

The proportion of rainy days variables (SRD/SD, SRD/TRD, and TRD/TD) were screened as gross indicators of cloud cover. The variable SRD/SD contributed significantly to variance reduction in the regression between the South and North Fork watersheds for the number of low-flow days (LOFLOZSN). This variable suggests that cloud cover and light summer rainfall or fog reduced evaporative demand that, in turn, caused a minor reduction in flow difference between the two streams.

Management Implications

This research indicates that the potential exists for increasing water yield from a second-growth Douglas fir and redwood forest by selective harvest operations. On the average a 15% increase in annual water yield would be expected for the first decade after logging. However, several important characteristics of this expected increase lessen its utility to water managers. First, the timing of the augmented yield is displaced from the time of peak demand. At Caspar Creek, 90% of the flow increase was realized during the rainy high-flow season. Water demand is usually greatest during low-flow periods in the summer. In addition, that portion of the flow increase that occurred during the low-flow season diminished rapidly in the years following logging. Beyond 5 years after the completion of logging, no significant flow increases were detected, and a possible decline in summer flows relative to prelogging levels was noted. Persistent summer flow augmentation would not be expected without continued vegetation reduction in the logged watershed. Also, the sizeable variation in flow enhancements detected in the postlogging years at Caspar Creek suggests that water yield increases could not be depended upon by planners and managers to meet specific water demand levels. This lack of certainty would reduce the utility of flow increases. The potential side effect of increased sediment yields accompanying streamflow enhancements realized in logged watersheds would counter the possible benefits. At Caspar Creek, *Rice et al.* [1979] found that stream sediment increased by 80% with road building and 275% with logging.

In the Pacific Northwest a large proportion of the forest land is managed for timber production. As a result, it appears probable that forest management decisions involving cutting cycle, volume harvested, and postharvest silvicultural practices will continue to be based on timber management objectives. Forest practices that encourage the rapid regrowth of trees are not likely to result in prolonged water yield increases related to harvesting. In contrast, forest operations designed to maximize water yield augmentations by inhibiting regrowth may pose problems related to slope stability, sedimentation impacts, and timber growth and yield.

We found some indication of a reduction in summer flows beyond 5 years after logging. This reduction might be the result of accelerated transpiration and growth by the residual vegetation. Reduced summer Aows may, in turn, affect the aquatic ecosystem by reducing in-stream habitat capacity for fish.

CONCLUSIONS

Selective logging of an 85-year-old second-growth Douglas fir and redwood forest at the Caspar Creek watershed resulted in the alteration of the amount and seasonal distribution of streamflow. Streamflow was augmented both for the low-flow season and the annual period. Increases were greatest in the year after logging was completed, 1974, and diminished irregularly thereafter. Increases in summer flow volume were detected between 1972 and 1978, although not all of these increases were considered statistically significant. Enhancement of the summer flow volume was less persistent than annual flow increases.

The prospects of increasing water yield by selective harvesting of second-growth forest along California's north coast are not promising for two important reasons. First, the difficulty of reliably predicting the timing and extent of streamflow increases resulting from logging would make this supply undependable. Second, although the quantity of available water may be increased, the quality of the supply could be adversely affected by increases in suspended sediment and turbidity. This study suggests that water yield increases resulting from selection harvesting along the northern coast of California will be of minimal importance when compared to other forest management and production goals.

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Environmental impacts of ORVs on the Rubicon Trail

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Photo: Little Rubicon Crossing. All Photos taken by Monte Hendricks.

Introduction

There are a multitude of scientific studies that provide sufficient evidence and support for carefully controlled and managed off-road vehicle use on public lands. These studies illustrate the demonstrated, detrimental and interconnected effects of off-road vehicles on wildlife, habitat, vegetation, soil, air, water and other users. The following is an overview of some of the major impacts to species and habitat that are most important to understand when discussing the situation on Rubicon Trail in order to fully grasp the urgent need to rein in abuse on the trail.

Photo 1. County's Rubicon Trail map. Note that the locations of the outflow from Spider and the wet crossing before Buck are on private land.

1. Riparian Zones, Meadows and Wetlands and Associated Wildlife and Plants

Sachet (1988) identified "sensitive" habitats where backcountry recreation is a concern due to 1) the ecological uniqueness of the habitat, 2) the essential habitat it provides for a key species and 3) the potential extreme sensitivity of the habitat to recreation. Two of these "sensitive" habitats are riparian zones and meadows (Sachet, 1988). The severity and extent of off-road vehicle damage can be greater in areas of uncommon habitat. Many species are dependent on riparian zones for their survival. Thus, due to a paucity of these habitats in certain areas in

California, as well as their fragility and importance, damage inflicted by ORVs can have extremely detrimental, expansive and persistent effects on riparian zones and meadows and the creatures that depend on these ecosystems.

Riparian habitat areas or corridors include the vegetated areas along streams, rivers or lakes. Research shows that riparian ecosystems, as well as meadows and wetlands are vital to the health of aquatic and terrestrial ecosystems because they filter out pollutants from land runoff, prevent erosion, and provide shelter and food for many aquatic and terrestrial animals (United States Department of Agriculture, 2002; Sanders and Flett, 1989). These areas are, however, sparse in certain areas (i.e., desert ecosystems in California) and fragile to disturbance from motorized recreation.

Riparian zones attract wildlife because they provide food and water, breeding and rearing areas, and hiding and resting opportunities. Riparian areas do not only provide direct sources of food to animals; they also support the needs of aquatic insects, a key food source for fish, and also provide an area for the growth of a prey base that will feed hawks, eagles, owls, falcons, bears and wolves (Brown, 1985). Riparian areas are ideal for breeding and rearing because of the diversity of resources they provide, including food, water and cover for newly born fawns and calves. In addition to providing nesting sites for raptors and their prey, riparian zones and associated vegetation help maintain the quality of spawning grounds for fish. Riparian areas are often used as corridors for movement because they provide cover and resources while connecting areas of critical habitat (i.e. deer often migrate through riparian cover to get from higher elevation areas in the summer to lower elevation areas in the winter).

Meadows are also extremely important to wildlife because they provide habitat for foraging and other necessary activities; meadow edges often have extremely high species diversity and richness. Deer and elk come to meadows to find nutritious vegetation that is not available in coniferous forests and may also use these areas for rutting and mating. Bear also rely on meadows for food and even sheep will forage in meadows that are associated with rocky terrain and cliffs.

There are countless television and magazine advertisements depicting a motorcycle, ATV or full-size vehicle crashing through a stream, driving up and along a river bank or racing across a muddy, wet meadow. Portrayals that glorify this type of behavior are misleading and only serve to perpetuate this type of action. In actuality, the operation of vehicles in streams and pools has the potential to destroy riparian vegetation and habitat for a range of animals, including invertebrates, fish, amphibians, reptiles, mammals and birds (Manning, 1979; Bury, 1980). ORV use often causes damage to streambanks that leads to increased erosion and sedimentation in streams and rivers. Studies have found that ORV use in wetlands, meadows, bogs and swamps can create ruts which ultimately alter hydrological patterns as they change the path of water (Heede, 1983;Lodge 1994;, Duever et al. 1981; Duever et al. 1986). These impacts can alter entire ecosystem processes, affecting the ecology of an entire area. DeMayndier and Hunter (1995) cite studies regarding the importance of riparian zones and streams to amphibians. They argue that impacts to “streamside vegetation and soils may have important effects on both stream-dwelling and upland streamside amphibian fauna (DeMayndier and Hunter, 1995).” For example,as we see on the Rubicon trail mismanagement or abuse of a trail leads to increased erosion and associated increased sedimentation that degrades the quality of habitat for larval and adult animals; this has been shown to lead to decreased amphibian diversity and abundance (Swanson et al., 1990).

Photo 2. Near wet crossing before Buck Island Lake (at ~ 39 degrees 00.872’ N and 120 degrees 16.009’ W - all coordinates are WGS84 datum).This is an ephemeral stream that flows down the granite east and down into the Little Rubicon.

There is evidence that ORV use in riparian areas can negatively affect birds that depend on this habitat (Sanders and Flett, 1989). Damage to riparian plant communities by ORV use has also had an impact on the ranges and distribution of other western subspecies (Wildlands CPR, 1999). Weinstein (1978) found that off-road vehicles changed the behavior of birds in riparian areas by causing them to move away from certain critical areas, to be flushed more readily and to alter their use of habitat (Weinstein, 1978). In a study in Northeastern California, Barton and Holmes (2007) found that off-road vehicles had negative impacts on nesting success and the abundance of breeding birds. They found that “areas within 100m of OHV trails may provide reduced-quality habitat to nesting songbirds, particularly for species that suffer significant losses of annual fecundity due to abandonment or desertion of individual breeding attempts

(Barton and Holmes, 2007)." As a result, they suggested that managers limit ORV trails in breeding areas of rare or endangered birds (Barton and Holmes, 2007)

Off-road vehicles also directly affect birds by damaging riparian habitat, water caches and vital cover for wildlife. For example, wetlands and riparian habitats provide necessary resting areas, food, and water to resident and migratory birds (Sanders and Flett, 1989). Because off-road vehicles have eased human access to stopover sites used by migrating birds, these critical areas are being destroyed by pollution and direct damage to plants as a result of crushing and cutting for firewood (Bury 1980). In addition, if the passage of off-road vehicles through riparian habitat does not directly destroy the area, it may still disrupt and restrict wildlife use. Sheridan (1979) warns that "...Any rare species inhabiting such uncommon ecosystems may be in danger of local or total extinction as a result of ORV use.

ORVs have increased the accessibility of remote areas, thus, creating the potential for damage in places that were previously protected. For example, when campsites in riparian areas are created and become established, riparian vegetation is often cut for fuel, erosion of heavily used areas occurs and litter and water pollution become common (Bury, 1980). This is very evident on the Rubicon Trail. In addition, when riparian areas are used by ORVs, there is also a good chance for gas and oil pollution from leaks and spills, as is evidenced by the need for the CAO issued by the Regional Board.

In 1999, Wildlands CPR released a document petitioning for a change in regulations regarding off-road vehicles on national forests. This petition identified riparian zones as a habitat requiring special attention because in many instances, the government has failed to meet its own legal obligation to protect riparian areas where "no management practices causing detrimental changes in water temperature or chemical composition, blockages of water courses, or deposits of sediments shall be permitted ... which seriously and adversely affect water conditions or fish habitat (36 C.F.R. §219.28(e))."

The most valuable management strategies will prevent damage by avoiding the creation of recreation opportunities in riparian zones and will mitigate damage by closing critical riparian, wetland and meadow areas like those found on the Rubicon trail to use.

Photo 3. Outflow from Spider Lake (39 degrees 01.080' N and 120 degrees 16.182' W.) This drops into an alpine pond in the flat where the RubiconTrail crosses. This ephemeral stream also flows east and down into the Little Rubicon.

2. Species Affected by ORV use on the Rubicon Trail

In the above discussion of riparian habitat it is clear that the use of ORVs in riparian areas can have impacts on many species ranging from birds to amphibians to plants. The Draft Environmental Impact Report: Rubicon Master Trail Plan (DEIR) supports this fact stating that high mountain lakes and streams in the project area provide: *"important habitat and sustenance for plants and wildlife…Amphibians, insects, and small invertebrates such as fresh-water shrimp (Syncaris pacifica) dominated these high-elevation aquatic ecosystems….In addition to aquatic species, large and small terrestrial mammals and avian species depend on these resources for forage, cover, nursery and nesting habitat, and migration corridors (ESP, 2007)."*

Clearly, a complete discussion of the impacts of the Rubicon trail must include the way in which a range of species – from amphibians to birds to plants – are affected by ORV use.

Herpetiles (Amphibians and Reptiles)

Herpetofauna are very important players in the food web because as a group, they are more abundant, they make-up more biomass and they contribute more significantly to the transfer of energy along the food web than mammals and birds. These creatures have an impact on communities at each stage of their development; amphibian larvae structure aquatic communities, lizards and metamorphosing amphibians provide a link between aquatic and terrestrial food webs and adults play a key role in maintaining the efficiency of terrestrial food webs. Because of these important roles and the fact that amphibians, and some reptiles, serve as indicators of the health of our environment, the impacts of routes and trails and off-road vehicle activity on herpetiles should be a management concern (Welsh and Ollivier, 1998). Those found in the Rubicon trail area include frogs (Rana spp.), alligator lizard (Gerrhonotus multicarrinatus), garter snake (Thamnophis sirtalis) and western rattlesnake (Crotaus viridis).

Off-road vehicle use can lead to the death of reptiles and amphibians due to direct kills, however, the elimination and degradation of vegetation and critical habitat by ORVs has a larger, long-term impact on these animals.

In addition to loss of vegetation and destruction of habitat, road traffic and the use of off-road vehicles can cause increased sedimentation and chemical contamination (as outlined in the CAO issued by the Regional Board) that can be detrimental to adjacent aquatic systems; large amounts of sediment can prove detrimental and even lethal to amphibians. Welsh and Olliver (1998) found a lower density of Tailed frogs (*Ascaphus true,* a Species of Special Concern in California*),* Pacific giant salamanders (*Dicamptodon tenebrosus*) and southern torrent salamanders (*Rhyacotriton variegatus,* a Species of Special Concern in California) in streams adjacent to road construction in Redwood National Park. Contaminated sediment and runoff from roads or campgrounds can also negatively affect amphibians and should be considered in management of places like the Rubicon trail where similar species exist.

Routes, trails and the use of off-road vehicles can create barriers to necessary movement (i.e., movement for migration, breeding, foraging). Studies have found a higher proportion of dead frogs and toads on routes with higher traffic volumes. Although this may result from higher direct mortality, it may also occur because traffic changes movement patterns and interrupts anuran behavior (Fahrig et al. 1995).

In a literature review discussing the impacts of forest management practices on amphibians in North America, deMayndier and Hunter (1995), contend that the forest roads can lead to long-term changes in habitat because routes increase fragmentation and decrease the permeability of the landscape. Marcot et al. (1997) also reports that roads can fragment some reptile habitats. Routes and trails that serve as barriers to amphibian and reptile movements can cause populations to become isolated and therefore, more susceptible to detrimental genetic and environmental consequences. Barriers also cause difficulties for herpetiles populations that migrate between aquatic breeding ranges and upland home ranges and may prevent populations from successfully breeding (i.e., Red-legged frog, *Rana aurora*, California Species of Special Concern, Federally Threatened).

Herpetiles are susceptible to direct mortality from off-road vehicle use, especially during dispersal and migration; however, they are more greatly affected by the associated loss of vegetation that causes the degradation of critical habitat. Marcot et al. (1997) state that "off-road vehicle use has become a major threat to reptiles" while various studies suggest that ORVs are also a threat to amphibians (DeMaynadier and Hunter, 1995; Maxell and Hokit, 1999). Managers should be concerned about "the potential impacts of secondary roads on sensitive species and should construct fewer and narrower roads with little or no verge clearance (DeMaynadier and Hunter, 2000)." Maxell and Hokit (1999) recommend that roads and trails avoid water bodies, wetlands and areas that are key habitat for amphibians and reptiles.

Birds

Riparian areas and associated species are, however, not the only ones affected by the Rubicon trail. Many birds that depend on conifer trees for their homes live in the forested lands along the Rubicon trail. These include cavity nesters such as the yellow-bellied sapsucker (Sphyrapicus varius),pileated woodpecker (Dryocopus pileatus), white-headed woodpecker (Picoides albolarvatus), hairy woodpecker (Picoides villosus), and northern flicker (Colaptes auratus). Other bird species observed within the project area include red-breasted merganser (Mergus serrator), mountain quail (Oreortyx pictus), Steller's jay (Cyanocitta stelleri), mountain bluebird (Sialia currucoides), warblers (Dendroica spp.), and mountain chickadee (Parus gambeli).

The fact that these animals depend on habitat along the Rubicon trail is important b/c of what Bury (1980) concludes from previous studies, "Birds apparently are the vertebrates most sensitive to ORV influence." Compared to areas subject to ORV use, he found 1.5 times the number of birds and twice the biomass and species of birds in control plots (Bury et al., 1977). A further analysis of the impact of ORVs on birds found that birds are susceptible to direct and indirect effects of off-road vehicles. By destroying nests, crushing individuals, harassing individuals and creating noise, off-road vehicles can directly impact birds (Luckenbach, 1978). Indirectly, off-road vehicle use can alter habitat and decrease the amount of shelter and forage available (Luckenbach, 1978; Severinghaus, 1982). ORVs can also effect the breeding success of birds. It is estimated that roads and motorized trails can influence the reproduction of forest birds located up to 200 meters from a trail (Foppen and Reijnen, 1994).

Mammals

The DEIR states that mammal species found on the Rubicon Trail include the striped skunk (Mephitis mephitis), chipmunk (Tamias spp.), western gray squirrel (Sciurus griseus), yellow-bellied marmot (Marmota flaviventris), coyote (Canis latrans), gray fox (Urocyon cinereoargenteus), and mule deer (Odocoileus hemionus). It also states that black bear (Ursus americanus) and other large mammals, including mountain lion (Felis concolor), range throughout the project area.

Studies have been done showing that small mammals are also prone to effects of off-road vehicle use; they are subject to direct mortality, disturbance and habitat loss and fragmentation as a result of ORV use and the creation of routes and trails. Small mammal distribution, abundance, behavior and movements are highly influenced by the volume of vegetation present because this represents the amount of food available in a certain area. Off-road vehicles easily damage vegetation quickly destroying critical food sources and habitat for small mammals. The destruction and conversion of habitat (i.e., the poor cover on forest roads) leaves small mammals vulnerable to predation because even routes that are small and of low use may act as barriers and may inhibit movement (Merriam et al, 1989; Burnett, 1992).

In addition to effects on habitat, ORVs also impact the abundance of small mammals by directly killing or crushing individuals or trapping one in a collapsed burrow (Luckenbach and Bury 1983).

Unlike small mammals, carnivores are not under stress from predators (except humans). They are still sensitive to disturbance from recreationists and are directly affected by the damage that ORVs do to the soil, air, water, and other animals. Carnivores will suffer if these resources are degraded because they, and their prey animals, are dependent on all of them for survival. Carnivorous creatures, such as the black bear and the mountain lion need large amounts of space within which to live, hunt, mate and breed. Thus, they are extremely vulnerable to the impacts of habitat fragmentation and loss of connectivity due to roads, trails and off-road vehicle activity. The creation and use of routes and trails, as well as cross country travel by motorized vehicles (which is prevalent along the Rubicon trail) is influential in the distribution and abundance of many carnivores (McReynolds and Radtke, 1978; Claar et al., 1999). In addition, the increased access provided by off-road vehicles and associated trails can be detrimental to the survival of carnivores because it may allow for over hunting or over trapping and illegal poaching, as well as for harassment of individuals.

Photo 4. Outflow from Buck Island Lake found below the Little Rubicon crossing (39 degrees 00.338' N and 120 degrees 15.377' W).

Special Status Species

The Initial Study done in March 2006 by Eldorado County to determine if an EIR was needed for the Rubicon Trail Master Plan concluded that there were *potentially significant impacts to species and habitat associated with continued ORV use along the Rubicon Trail.* This study and the DEIR conclude that there are also sensitive animal and plant species that occur in the area that may be affected by ORV use on the Rubicon trail stating: "vehicle operations outside of the primary route result in substantial increased potential for specifies or habitat disturbance. The illegal creation of bypasses or variants has the potential to modify habitat and adversely affect candidate, sensitive, and/or special status species as recognized by the U.S. Fish and Wildlife Service and the California Department of Fish and Game (CDFG)."

The following is a brief discussion of the potential impacts to some species based on the information available (See Attached Table 3-8.1 and 3-8.2 for the list of sensitive species from the DEIR and the attached list showing the species found on the Homewood Quad from a search of the CNDDB.)

ORV use has the potential to directly or indirectly affect special-status plant species or other sensitive natural communities.

Several sensitive plant species have the potential to exist in the Rubicon Trail vicinity. The DEIR states that the area supports potential habitat for Stebbin's phacelia and shore sedge and limited or marginal habitat for northern adder's-tongue and marsh skullcap. This may not be a comprehensive list due to the fact that the DEIR focused only on the El Dorado County portion of the Rubicon trail. The DEIR states that the primary threat is not ORV operation along the trail, but the proliferation of off-trail travel that will result in substantial increased potential damage to species or habitat, including sensitive, candidate or special-status species.

ORV use on the Rubicon Trail has the potential to directly or indirectly affect mountain yellow-legged frog.

Mountain yellow-legged frog (MYLF; *Rana muscosa*) occurs in streams, lakes, and ponds, in montane riparian, lodgepole pine, subalpine conifer, and wet meadow habitat types at elevations above 6,000 feet. MYLF prefers habitat with rocks and vegetation at the shallow perimeter. They typically crouch on rocks or in vegetation within 30 feet of the aquatic habitat and takes refuge in vegetation, under rocks, or at the bottom of ponds. MYLF adults hibernate beneath ice covered streams, lakes, and ponds during the winter months. Reproduction takes place after mountain lakes and streams are free of ice when MYLF eggs are laid in shallow water attached to gravel or rocks..

The DEIR identifies the fact that ORV use on the trail has the potential to significantly affect the mountain yellow legged frog (*Rana muscosa).*The DEIR also states that "potentially suitable habitat was observed along the Rubicon Trail for the mountain yellow-legged frog (MYLF) and this species was also reported in theCalifornia Natural Diversity Database (CNDDB) with recorded occurrences within the vicinity of the Rubicon Trail (ESP, 2007 at 3.8-7)." In the absence of a clear and enforceable management plan, the MYLF is at risk if ORV use is allowed to continue unchecked.

ORV use on the Rubicon Trail has the potential to directly or indirectly affect Yosemite toad.

The Yosemite toad (*Bufo canorus*) occurs in wet meadows and seasonal ponds in the central high Sierra Nevada Range at elevations between 6,400 and 11,300 feet. They typically prefer quiet pools in alpine meadows and seek cover inside abandoned rodent burrows or adjacent forested areas. They typically remain near water where they retreat if threatened. The DEIR considers the impact on the

Yosemite toad, but dismisses it as less than significant saying that because the Rubicon Trail is generally beyond the northern extent of the range for Yosemite toad, it is unlikely that use on the Rubicon would impact it. This is, however, based on the fact that no individuals were found in surveys and is therefore inconclusive.

ORV use on the Rubicon Trail has the potential to directly or indirectly affect the Marten, Fisher, and Wolverine

The DEIR identifies all 3 of these species of federal concern as those that occur in the vicinity of the Rubicon Trail. The primary factors influencing this group are direct mortality from trapping, habitat alteration (largely as a result of logging and development) and disturbance responses. Because of their life history and behavior, all 3 of these animals are commonly caught and over harvested (Powell, 1979; Thompson, 1994; Witmer et al. 1998). Recreational trails and roads increase access for humans, thereby increasing the susceptibility of forest carnivores to trapping (Hodgman et al., 1994; Witmer et al., 1998; Claar et al., 1999). In addition, animals crossing trails or wide open areas where ORVs travel can lead to direct mortality. The combination of over harvesting and road kill can be even more significant in small populations found in fragmented habitat because movement and dispersal is limited.

Animals respond physiologically to disturbance (MacArthur et al., 1982; Yarmoloy et al., 1988; Gutzwiller, 1995). These responses can include change in heart rate, body temperature, respiration rate, etc. Claar et al. (1998) state that it is likely that human disturbance, including off-road vehicle use, evokes similar responses and an expenditure of energy in martens, fishers and wolverines. Forest carnivores are especially vulnerable to disturbance caused by recreational activities because they need large home ranges, have specific habitat requirements and have a low reproductive potential. Thus, the preservation of areas of undisturbed habitat without roads, off-road vehicle use, hunting or trapping is necessary for the persistence of forest carnivore populations, especially those who are, like the pacific fisher who are facing increased pressure and who need further protections (hence its status as a candidate for listing).

3. Extent of Impacts

We contend that it is not biologically sound to evaluate impacts to resources, such as riparian vegetation and wildlife habitat on a restrictive, area- specific basis. We present the following information based on peer-reviewed scientific research to support the idea that the effects of motorized trails, like the Rubicon, extend well beyond the actual area that they occupy on the ground. Based on the following scientific concepts: the virtual footprint, indirect effects, road effect zone and road avoidance zone, it can be argued that the Rubicon trail has the potential for much more significant impact on the environment than just those impacts associated with the length and width of the trail.

A true assessment of environmental impacts would reference and apply the myriad scientific literature that exists regarding road ecology and cumulative environmental impacts. A scientifically valid and ecologically representative analysis of the Rubicon Trail must consider all of the cumulative impacts of the route at a broader scale to accurately determine the true effects of the route within the broader context of the landscape.

Virtual Footprint

Forman et al. (2003) state all roads not only have a physical footprint, but also a "virtual footprint" surrounding their actual location. This virtual footprint includes the "accumulated effect over time and space of all of the activities that roads induce or allow, as well as all of the ecological effects of those activities (Forman et al. 2003)." For example, the United States has 6.4 million km of public roads that are used by over 200 million vehicles (FHWA, 2003). Road corridors cover approximately 1% of the United States; however, the ecological impacts of these roads are not restricted to this area alone. It is estimated that 19% of the land surface in the U.S. is directly affected by roads, while in total, 22% of the U.S may be ecologically altered by the road network (Forman 2000). This concept extends to forest roads, as they have been shown to cause fragmentation, habitat loss, damage to riparian ecosystems and soil degradation well beyond their actual footprint (Gucinski et al., 2000).

Because a larger virtual ecological footprint is associated with the physical footprint of roads, "road planners//builders and environmentalists need to be concerned with the broad landscape rather than the one-dimensional road corridor (Forman et al. 2003)." The environmental evaluations completed for the Rubicon (i.e. in the Draft Environmental Impact Report) largely focus on the one-dimensional road corridor, thus they are is not complete and/or accurate in their evaluation of the actual ecological impacts made by the virtual footprint.

Indirect and direct effects

Many scientists suggest that motorized recreation is the greatest threat to wildlife on our public lands because it can alter habitat, cause disturbance and lead to the direct death of animals (Luckenbach, 1975, 1978; Bury and Luckenbach, 1983, 2002; Sheridan, 1979; Berry, 1980; Brattstrom and Bondello, 1983; Boyle & Samson, 1985; Havlick, 1999; Joslin and Youmans, 1999; Lovich and Bainbridge, 1999; Lawler, 2000; Belnap, 2003).

Lovich and Bainbridge (1999) acknowledge the significance of direct mortality but argue that the more detrimental repercussions of linear recreation corridors include habitat fragmentation, restriction of wildlife movements and gene flow, and increased human access to remote areas. They also explore other consequences of off-road vehicles, including destruction of soil stabilizers, soil compaction, reduced water infiltration rates, destruction of vegetation, and increased erosion (Lovich and Bainbridge, 1999).

To the casual observer, the impacts of forest roads and motorized recreation on wildlife may not be as evident as their effects on the surrounding physical environment (i.e. loss of trees, damage to ground surface, etc.). In reality, however, wildlife are affected beginning when a route is first cut (legally or illegally) and continue to be even after the route is no longer being used. As ORVs affect soils, air, water and vegetation, they also impact wildlife species because animals depend on all of these other factors for their survival. Thus, ORV activity and associated routes have both direct and indirect effects on animals (Davenport et al, in press).

Animal mortality, a significant direct effect, can occur when off-road vehicles hit ground-dwelling animals, destroy birds or small mammals by crushing ground nests or vegetation that contains nests, or cause the collapse of needed burrows. Although animal mortality is an obvious and familiar direct effect, displacement, avoidance and disturbance at specific sites, often associated with breeding and raising young, are the most commonly reported direct effects of motorized trails on wildlife (Bury et al. 1977; McReynolds and Radtke, 1978; Bury, 1980; Luckenbach and Bury, 1983; Sachet, 1988; United States General Accounting Office, 1995 Youmans, 1999).

Off-road vehicle activity and harassment can stress animals, resulting in a measured physiological stress response or increase in energy use (Schultz and Bailey, 1978; King and Workman, 1986; Canfield et al., 1999). Changes in animal behavior, (i.e., the abandonment of important activities like hunting, foraging and mating), have been attributed to the passage of off-road vehicles. These behavioral and physiological responses to motorized human disturbance may not only impact individuals, but also entire populations. It has been suggested that the impacts associated with disturbance from ORVs can increase the risk of individual mortality and decrease the productivity and viability of an entire population (Knight and Cole, 1991). For example, if the passage of an ORV causes a male yellow warbler in a canyon to change his habitat use pattern

While the consequences of direct effects (i.e. a road kill) may be more evident, indirect effects on wildlife are significant and often impact habitat in areas subject to motorized recreation. For example, ORV activity that destroys vegetation by crushing it and exposing roots, also disturbs soil, thereby negatively effecting future plant growth and the potential for healthy habitat for many animals. The destruction of habitat can increase fragmentation and decrease connectivity, breaking previously suitable habitat into smaller patches which may make it less usable and can jeopardize the survival of certain species. “Edge effects” increase and are magnified in areas with small, isolated patches of habitat. Increased edge effects can impact wildlife that need interior habitat for foraging, hunting or establishing home ranges (i.e., mountain lions, martens, black bears). Research also shows that fragmentation and increased edge habitat support the invasion of non-native, noxious and weedy species that eventually displace native interior species. The destruction of native vegetation and changes in the density and diversity of plant communities as a consequence of prolonged off-road vehicle use can even further change the composition of desert reptile and small mammal communities (Bury, 1980).

Indirect effects often have such broad implications because the “road effect zone,” or the outer limit of a significant ecological effect, extends much further than the actual road, route or trail (Forman 2000). Disturbance due to noise, pollution, ground impact, and speed will travel beyond the actual surface of any route. In addition, ecological effects will ripple, expanding well beyond the perimeter of a route and potentially affecting an entire ecosystem. For example, in aquatic areas like the Rubicon Trail, off-road vehicles can increase the amount of silt and turbidity in a stream by increasing erosion (Moyle and Leidy, 1992). If this causes degradation of habitat to the point where spawning sites are not available and food sources are destroyed, less fish will survive and so will those creatures that depend on the aquatic ecosystem for survival.

In an evaluation of threats to biodiversity, Wilcove et al. (1998) ranked habitat destruction and the spread of alien species as the two greatest threats; off-road vehicles contribute to both of these. There are a number of causes of habitat destruction, including land conversion, agriculture, development and outdoor recreation. From their study of these causes, they reported that 15 % of all endangered species are affected by roads. Twenty seven percent (27%) of all endangered species, including plants and animals, are harmed by outdoor recreation while 13% of endangered species have been specifically, negatively impacted by the use of off-road vehicles (Wilcove et al. 1998).

Studies with similar findings regarding the impacts of off-road vehicles on wildlife and their habitat abound. Bury et al. (1977) studied the impacts of ORV use on wildlife in creosote shrub habitat in the California desert. The authors found a negative effect on desert wildlife wherever ORVs were used. In a comparison with control areas, they reported significantly less species diversity, fewer individuals present and lower biomass of mammals and reptiles in areas used by ORVs. Diversity, abundance, and biomass of avian species were also significantly greater in undisturbed areas than in those used by ORVs (Bury et al., 1977). Results also support the idea that a decrease in fauna is correlated with the level of off-road activity. The authors conclude that activity related to ORV use negatively affects desert wildlife and creosote shrub habitat, both of which they argue are irreplaceable (Bury et al., 1977).

Luckenbach and Bury (1983) conducted a study to determine the ecological impacts of ORV use on biota by comparing presence and density of vegetation, rodents, arthropods and lizards on plots with and without use by off-road vehicles in sand dunes in south eastern California. They found that ORV activity in the Algodones dunes reduced the biota; in areas of ORV use, there were less herbaceous and perennial plants, arthropods, lizards and rodents. Researchers found almost no native plants or wildlife in areas of heavy ORV use and also cited negative impacts to the biota in areas with low levels of ORV activity. They argue that ORV activities very negatively affect dune biota and even low levels of use can cause a reduction in the biota of ecosystems.

Although we discuss them separately, the actual environmental effects of these factors are not individual. Rather, they are cumulative and synergistic because seemingly may result in large scale changes in the reproductive success and survival of organisms, thereby altering the entire ecology of an area. The combination of these impacts has the potential to cause disturbance at the landscape level (McLellan and Shackleton, 1988; Eaglin and Hubert, 1993). Few species or habitats are completely immune to the effects of off-road vehicle recreation and many are threatened by similar impacts: habitat loss or fragmentation, disturbance, displacement and direct mortality.

Clearly, any complete analysis of off-road vehicle activity in a delicate riparian ecosystem like the Rubicon Trail is not complete if it only quantifies direct effects based on specific acreage of impact; , it must also take into account the far reaching indirect effects.

Road effect zone

Roads are responsible for a suite of indirect effects that impact species dynamics, soil characteristics, water flow regimes, and vegetation cover (Bashore et al. 1985; Reijnen et al. 1996, Forman et al. 2003). The degree of indirect effect varies in relation to the distance from a road, extending to what is known as the "road effect zone" or the outer limit of significant ecological effect (Forman et al. 1997; Forman and Deblinger 1998, 1999). Forman and Deblinger (2000) found that the effects of all nine ecological factors studied extended more than 100 m from the road, with some extending outwards of 1 km of the road. The road-effect zone was asymmetric, had convoluted boundaries and a few long fingers and averaged approximately 600m in width.

Road-avoidance zone

Native wildlife species are less common or absent near roads, suggesting the existence of a road-avoidance zone (Forman and Alexander, 1998). Evidence of a road- avoidance zone exists for deer, elk, coyote, small mammals, birds, amphibians, snakes and caribou. Road-avoidance zones, extending outwards tens or hundreds of meters from a road, generally exhibit lower population densities compared with control sites. Forman et al. (2005) conclude that the ecological impact of road avoidance probably exceeds the impact of either road-kills or habitat loss in road corridors.

Clearly, most of the ecological effects of road systems are negative and their cumulative effect covers an extensive area (Forman 2000). Landscape ecologists and scholars of related fields increasingly recognize ecological flows across the landscape as critical for long-term nature protection (Forman 1995, 1999; Harris et al. 1996). Forman suggests that because of this, the road effect zone should be the basis for transportation planning, implying that a landscape perspective is necessary to maintain spatial and biological diversity.

A report by Gucinski et al. (2000) suggests that this type of full analysis is necessary and is within the realm of possibility for government agencies charged with managing our public lands:

This overview of scientific information leads us to conclude the following: The emerging science of the effects of roads as networks in the landscape requires considerable new research. Because of the high degree of variability of roads from place to place and region to region, a framework for evaluating benefits, problems, risks, and tradeoffs among them would provide a powerful decision-making tool. We believe such a framework is now in place (USDA 1999). Conducting these analyses is well within the grasp of capable specialists, planners, and managers to bring their expertise to bear on the problem of reducing risks from past, current, or planned roads, and targeting future road-restoration activities. The science pieces for analyzing and integrating road systems and their effects are already developed.

Despite the fact that this capability exists, assessments like the one completed for the Rubicon Trail in the DEIR continue to neglect to consider the virtual footprint of a route or the road-effect zone. Angermeier et al. (2004) argue that assessments of environmental impacts of roads are inadequate to ensure informed decision making because direct, localized or acute impacts are emphasized whereas indirect, dispersed or chronic impacts are neglected. This bias reflects the typical level of analysis wherein attention is narrowly focused on points of impact or species rather than ecosystems, on site-specific scale rather than regional scale and on short-term rather than long-term environmental impacts. This is particularly disturbing due to the fact that: "The mismatch between scales of assessment and impact is especially problematic for roads because there is compelling scientific evidence that long-term, large-scale impacts are the greatest threats to biota (Angermeier et al. 2004)."

We agree with the above scientific evidence and argue that further analysis of the impacts of the Rubicon Trail must be completed before use is allowed to continue. The current lack of a management plan means that there is insufficient consideration of the far-reaching cumulative effects of off-road vehicle travel on a riparian ecosystem and a complete failure to address the indirect effects of the mere presence of trail alongside and in a creek. When the virtual footprint of a route is considered, along with the road-effect zone, the road avoidance zone and the further reaching indirect effects, the Rubicon Trail has the potential for a much more significant impact on the environment than has yet been considered.

Photo 5. Pond below Little Sluice Box (39 degrees 01.288' N and 120 degrees 16.517' W). This receives the run off in the ephemeral stream that runs down the Little Sluice Box. The County's DEIR for the Rubicon Trail Master plan reported cadmium and copper in the sediments. Taken May 9, 2007.

Photo 6. Same pond as photo 5, but taken Oct 13, 2007. This pond receives everything that comes down the Little Sluice Box drainage and joins with the flows which come down the Trail from west of this spot. The inflow is on the left side of the photo and the sediment filling that end of the pond is evident.

4. Pollution

Research suggests that off-road vehicles, including motorcycles, all-terrain vehicles (ATVs), snowmobiles, etc. contribute greatly to the pollution of water and air in the United States (Gucinski et al., 2000). They increase pollution by depositing unburned fuel into the soil, snow or water and by emitting pollutants into the air. This directly alters the composition of soil and snow while indirectly affecting vegetation and aquatic systems. Off-road vehicles also emit dangerous levels of toxins, including carbon monoxide (CO), nitrogen oxides (NO), and hydrocarbons (HC). In addition, off-road vehicles release compounds that are known human carcinogens (particulate matter (PM), benzene and polycyclic aromatic hydrocarbons, (PAHs)), and a suspected carcinogen (methyl tertiary butyl ether, MTBE). Thus, the effects of pollution generated by ORVs are pervasive as they extend well beyond any route or trail, affecting the health of humans, wildlife, vegetation and entire ecosystems.

A significant amount of damage can be attributed to the unburned fuel that ORV engines deposit into the environment. Off-road vehicle engines may be either two-stroke or four-stroke; two-stroke engines use fuel less efficiently and emit more unburned hydrocarbons (HC) and particulate matter (PM) than four-stroke engines. The Environmental Protection Agency estimates that 25 to 30% of the fuel in a 2-stroke motor remains unburned and is released into the air and water (Natural Trails and Water Coalition, 2005b). Because they are more powerful, lighter weight and are less expensive, two-stroke engines can be found in 60-65 % of off-highway motorcycles and 10 to 15 % of all ATVs in the United States (Unites States Environmental Protection Agency, 2001a). In 1993, the California Air Resources Board found that motorcycles with 2-stroke engines release 10 times the amount of hydrocarbon emissions as 4-stroke motorcycles. As a result of the amount of emissions released, use of 2-stroke engine motorcycles are now responsible for 90% of the emissions from ORVs that contribute to the formation of smog in California (California Air Resources Board, 2001). This is of interest on the Rubicon trail because the Eldorado National Forest is within a designated non-attainment area for state standards PM10 and ozone.

In 1994, the Environmental Protection Agency (EPA) definitively announced that non road engines "are significant contributors to ozone or carbon monoxide concentrations." Durbin et al. (2004) report that off-road vehicles are "one important source of emissions that make a disproportionately high contribution to the emissions inventory." For instance, between 1989 and 1998, pollution due to off-road vehicles grew from 17 to 22 percent of the total produced by mobile sources in the U.S, while pollution from cars decreased from 62 to 56 percent despite the fact that the number of these vehicles and the miles driven increased (United States Environmental Protection Agency, 2001b). This may be due to the fact that the hydrocarbon and carbon monoxide emissions released by a new passenger car are much lower than those released by 2-stroke or 4-stroke engines.

A significant proportion of the research conducted on ORV pollution relates to its impact on air quality and human health. However, pollution emitted by ORVs can have severe impacts on aquatic and terrestrial systems. The substantial amount of unburned fuel released by ORVs may be deposited into the soil where it has the potential to penetrate into underground water, adversely impact vegetation or run off into the aquatic system. A rapid pulse of these toxins into a system can quickly increase the acidity of a stream or waterway, causing the death of aquatic insects and amphibians (Hagen and Langeland, 1973). Acidification due to atmospheric deposition and pollution has been shown to effect the survival and distribution of amphibians, including tiger salamanders, boreal toads, and northern leopard frogs (Freda and Dunson, 1985; Harte and Hoffman, 1989; Corn and Vertucci, 1992). By releasing hydrocarbons and volatile organic compounds into streams and lakes, off-road vehicles can also disrupt the biological functions of fish, disrupting their ability to maintain their metabolism and immune system while also jeopardizing their reproductive success and survival (Tjarnlund et al. 1995, 1996; Juttner et al., 1995a,b). In addition, there is evidence that low levels of PAHs released by ORVs are toxic to zooplankton, restricting the reproductive success of zooplankton and many fish (Giesy, 1997;Oris, 1998).

The consideration of pollution is especially important on the Rubicon because the Regional Board found these types of impacts as a result of ORV use on the Rubicon trail; low levels of oil and grease were identified in water and soil samples and low levels of copper and cadmium were identified in soil samples. The Regional Board concluded that this contamination is due to motor oil, grease, and other petroleum-based fluids spilling and leaking from ORVs that have overturned or have damaged mechanical components while traversing rocky segments of the trail.

Off-road vehicles release detrimental pollutants, including carbon monoxide and particulate matter that work their way into the air, water, soil and snow, affecting human and environmental health. The same toxic chemicals and compounds that impact human resources and health can also affect the health and survival of wildlife and vegetation that are exposed to polluted air, water and/or food sources. Although these impacts may be silent or unnoticeable to the eye, governmental organizations and land management agencies have a legal and ethical responsibility as set out by the Clean Air Act and Clean Water Act to address the overwhelming amounts of pollution that currently threaten our lands and the people and wildlife who use them.

Figures

Figure 1. Table 3.8-1 From DEIR. Special-status wildlife species potentially occurring in the vicinity of the Rubicon Trail.

Table 3.8-1
Special-Status Wildlife Species Potentially Occurring in the Vicinity of the Rubicon Trail

Common Name (Scientific Name)	Status	Habitat
Northern goshawk (*Accipiter gentiles*)	FSC	CF, RF
Sierra Nevada mountain beaver (*Aplodontia rufa californica*)	FSC	RF, CF
California wolverine (*Gulo gulo luteus*)	FSC, ST	CF,
Bald eagle (*Haliaeetus leucocephalus*)	FT	CF, LK
American marten (*Martes Americana*)	FSC	CF
Pacific fisher (*Martes pennanti pacifica*)	FSC	CF, RF
Mountain yellow-legged frog (*Rana muscosa*)	FE	ST, PO
Yosemite toad (*Bufo canorus*)	FC	
Sierra Nevada red fox (*Vulpes vulpes necator*)	ST	CF

Status:
- FE = Federal Endangered
- FT = Federal Threatened
- FSC = Federal Species of Concern
- SE = State Endangered
- ST = State Threatened
- FC = Candidate for Federal Listing

Habitat Codes:
- CF = Coniferous Forest
- ST = Stream
- PO = Pond
- RF = Riparian Forest
- LK = Lake (permanent water body)

Figure 2. Table 3.8-2. From DEIR. Sensitive plant species potentially occurring in the Vicinity of the Rubicon Trail.

Table 3.8-2
Sensitive Plant Species Potentially Occurring in the Vicinity of the Rubicon Trail

Common Name (Scientific Name)	Status	Blooming	Habitat	Elevation	Duration	Growth Form
Stebbin's phacelia (*Phacelia stebinsii*)	1B	June to July	W, CF, M, S	2,000 to 6,600'	Annual	Herb
Northern adder's-tongue (*Ophioglossum pusillum*)	CNPS-2	July	G, WT	3,300 to 6,600'	Perennial	Herb
Marsh skullcap (*Scutellaria galericulata*)	CNPS-2	June to Sept.	CF, M, S, WT	0 to 6,900'	Perennial	Herb
Shore sedge (*Caren limosa*)	CNPS-2	June to August	CF, M, S, WT	4,000 to 8,900'	Perennial	Herb
Tahoe yellow cress (*Rorippa subumbellata*)	1B	May to Sept.	M, S, CF	6,250 to 6,270'	Perennial	Herb
Alpine dusty maidens* (*Chaenactis douglasii* var. *alpina*)	CNPS-2	July to Sept.	BR	9,900 to 11,200'	Perennial	Herb
Fell-fields claytonia* (*Claytonis megarhiza*)	CNPS-2	July to August	BR, CF	8,500 to 10,900'	Perennial	Herb
Long-petaled lewisia* (*Lewisia longipetala*)	1B	July to August	BR, CF	8,250 to 9,650'	Perennial	Herb

* Not likely to occur within the project area due to elevation range.

Habitat codes:

G	=	Grassland
W	=	Woodland
S	=	Seep
BR	=	Alpine Boulder/Rock Field
WT	=	Wetland
CF	=	Coniferous Forest
M	=	Meadow

Figure 3. List of Species found in the Homewood Quad. Accessed from CNDD.

California Department of Fish and Game
Natural Diversity Database
Selected Elements by Scientific Name - Portrait

	Scientific Name/Common Name	Element Code	Federal Status	State Status	GRank	SRank	CDFG or CNPS
1	***Accipiter gentilis*** northern goshawk	ABNKC12060			G5	S3	SC
2	***Botrychium crenulatum*** scalloped moonwort	PPOPH010L0			G3	S2.2	2.2
3	***Botrychium montanum*** western goblin	PPOPH010K0			G3	S1.1	2.1
4	***Capnia lacustra*** Lake Tahoe benthic stonefly	IIPLE03200			G1	S1	
5	***Carex praticola*** northern meadow sedge	PMCYP03B20			G5	S2S3	2.2
6	***Empidonax traillii*** willow flycatcher	ABPAE33040		Endangered	G5	S1S2	
7	***Helisoma newberryi*** Great Basin rams-horn	IMGASM6020			G1Q	S1	
8	***Lepus americanus tahoensis*** Sierra Nevada snowshoe hare	AMAEB03012			G5T3T4Q	S2?	SC
9	***Martes americana sierrae*** Sierra marten	AMAJF01014			G5T3T4	S3S4	
10	***Martes pennanti (pacifica) DPS*** Pacific fisher	AMAJF01021	Candidate		G5	S2S3	SC
11	***Myotis volans*** long-legged myotis	AMACC01110			G5	S4?	
12	***Rorippa subumbellata*** Tahoe yellow cress	PDBRA270M0	Candidate	Endangered	G1	S1.1	1B.1
13	***Scutellaria galericulata*** marsh skullcap	PDLAM1U0J0			G5	S2.2?	2.2

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FUEL TO BURN

THE CLIMATE AND PUBLIC HEALTH IMPLICATIONS OF OFF-ROAD VEHICLE POLLUTION IN CALIFORNIA

A CENTER FOR BIOLOGICAL DIVERSITY REPORT

Fuel to Burn:
The Climate and Public Health Implications of Off-road Vehicle Pollution in California

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Motorcycle in Jawbone Canyon, California
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The Clean Air Initiative, a project of the American Lung Association of San Diego & Imperial County, is dedicated to improving the air quality and health of residents in Imperial County and the Mexicali border region through education, advocacy, and support. The Clean Air Initiative coalition members include health care agencies, nonprofit organizations, environmental agencies, and Imperial Valley community members.

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Executive Summary

In 2006, California took a giant leap forward in addressing the threats posed by global climate change by passing landmark legislation, the Global Warming Solutions Act. Under this law, the state commits to reducing its emissions of greenhouse gases to 1990 levels by the year 2020 — a reduction of approximately 29 percent compared to the projected business-as-usual scenario. In addition, Governor Schwarzenegger's Executive Order S-3-05 commits the state to reducing greenhouse gas emissions to 80 percent below 1990 levels by 2050. Currently, the California Air Resources Board is crafting rules to achieve the new greenhouse gas emission reductions targets.

As described below, because off-road vehicles produce significant greenhouse gases, California should ensure that emissions from this source are reduced at the same pace as other sources. At a minimum, emissions from off-road vehicles should be reduced to at least 1990 levels by 2020 with further reductions to 80 percent below 1990 levels by 2050.

The gas used annually by California off-road vehicles equals that used in 1.5 million car trips between San Francisco and Los Angeles.

The state has also made a commitment to protecting the quality of the air that California residents breathe. California has among the poorest air quality in the nation and is home to 13 of 20 counties nationwide most at risk to adverse health impacts from smog.

In addressing the twin goals of reducing greenhouse gas emissions and protecting public health from the adverse effects of poor air quality, California needs to immediately address the pollution and greenhouse gas emissions from off-road vehicles. These emissions, while a relatively small component of the overall transportation sector, are a significant and growing source of greenhouse gases. Due to the meteoric rise in the number of off-road vehicles, these emissions will climb significantly if steps are not taken to curb them.

Off-road vehicles in California currently emit more than 230,000 metric tons — or 500 million pounds — of carbon dioxide into the atmosphere each year. This is equivalent to the emissions created by burning 500,000 barrels of oil. The 26 million gallons of gasoline consumed by off-road vehicles each year in California is equivalent to the amount of gasoline used by 1.5 million car trips from San Francisco to Los Angeles.

Because of the significant pollution caused by off-road vehicles, a reduction in emissions will have important health benefits for Californians. Off-road vehicles emit considerably more pollution than automobiles. According to the California Air Resources Board, off-road motorcycles and all-terrain vehicles produce 118 times as much smog-forming pollutants as do modern automobiles on a per-mile basis.

In the past 15 years, pollution from off-road vehicle use has increased significantly. Emissions of total organic gases and reactive organic gases — which are important precursors to smog — have doubled. Carbon monoxide emissions have increased by 56 percent. Emissions from current off-road vehicle use statewide are equivalent to the carbon dioxide emissions from 42,000 passenger vehicles driven for an entire year or the electricity used to power 30,500 homes for one year. If left unchecked, the emissions from off-road vehicles will continue to increase; as California addresses the difficult problems posed by global warming, emissions from off-road vehicles must be addressed.

This pollution is having a significant impact on the health of Californians. Imperial County, for example, is one of the most popular off-road vehicle recreation destinations in the state. It also has among the worst air quality in California. Childhood asthma rates in Imperial County are far higher than the statewide average. Air pollution is a contributor to the high rates of asthma, bronchitis, pneumonia, and allergies in this region, especially among children younger than 14 years old.

Despite these serious climate and health implications, the State of California has failed to seriously address the greenhouse gas emissions and pollution associated with off-road vehicle recreation. The California Air Resources Board currently allows the continued sale and use of polluting off-road vehicles that do not meet state emissions standards. And the Department of Parks and Recreation spends tens of millions of dollars each year promoting and supporting off-road vehicle use on state and federal public lands.

The significant reduction in greenhouse gas emissions mandated by the Global Warming Solutions Act applies to all greenhouse gas sources throughout the state. However, not all sources are able to realize reductions to the same degree at the same economic and societal costs. Because

Dusty trail in dirt-bike and all-terrain vehicle park. Dust, a component of particle air pollution, makes unpaved roads the largest single source of particulate matter.
Photo by Laurel Hagen

recreational off-road vehicle use is entirely discretionary, emissions reductions in this source to levels at or significantly below 1990 levels may be used to offset other sources that are less discretionary or that involve higher costs. For the policy recommendations below, we urge the Air Resources Board to assess the benefits of using each policy mechanism to achieve much greater reductions in this source. In all cases, a reduction to 1990 levels by 2020 should be considered only as the minimum reduction alternative.

Limiting overall off-road vehicle emissions will ensure that recreational polluters are reducing emissions at the same pace as other sectors of the population. Consistent with Assembly Bill 32 and the governor's executive order, emissions from off-road vehicles should be reduced to at least 1990 levels by 2020 with further reductions to 80 percent below 1990 levels by 2050. In order to meet this target, we offer the following recommendations:

• **The California Air Resources Board, in cooperation with the Department of Parks and Recreation, should limit greenhouse gas emissions from off-road vehicle use in state vehicular recreation areas and other state lands to at least 1990 levels.**

The Department of Parks and Recreation should develop a statewide plan to reduce statewide off-road vehicle emissions to the maximum extent possible. The plan should include options to reduce greenhouse gas emissions from discretionary recreational off-road vehicle use to at least 1990 levels by 2020. No new state off-road vehicle sites should be established unless they are consistent with such a plan. An initial analysis of the amount of greenhouse gases currently being emitted from off-road vehicle use within state vehicular recreation areas and other state lands is crucial in developing a statewide plan and individual management plans to reduce off-road vehicle emissions from these areas.

• **The State of California should ensure that federal agencies managing off road recreation in California are limiting greenhouse gas emissions from off-road vehicles to at least 1990 levels and should withhold financial support and permits from federal agencies that do not meet this target.**

Because significant greenhouse gas emissions arise from off-road vehicle use on federal lands, the State of California must ensure that those emissions are reduced along with emissions from other sources. This means that:

o *The California Air Resources Board should reject applications for continued or expanded off-road vehicle use by federal agencies that are not reducing emissions.*

The California Air Resources Board should adopt rules that require rejection of applications for new, continued, or expanded off-road vehicle recreation on federal lands from federal agencies or districts that do not have an adequate plan to reduce overall off-road vehicle emissions from their jurisdiction to at least 1990 levels.

Off-road motorcycle sending up a cloud of dust
Photo by George Wuerthner

o *The Department of Parks and Recreation should reject applications for funding from federal agencies that are not reducing emissions.*

The California Department of Parks and Recreation provides tens of millions of dollars to federal agencies to promote and manage off-road vehicle recreation. The Off-Highway Motor Vehicle Recreation Division should adopt rules that disallow applications for funding from federal agencies or districts that do not have a sufficient plan to reduce overall off-road vehicle emissions from their jurisdiction to at least 1990 levels.

o *The State of California should provide substantive comments on federal land-use plans and proposals that will result in increased greenhouse gas emissions.*

The State of California has several opportunities to significantly reduce greenhouse gas emissions from off-road vehicle use on federal lands. The California Air Resources Board, the state, and appropriate state agencies should participate in the public planning process for proposed federal land management plans, travel management plans, and individual projects to actively promote the position that each plan or project must be consistent with an overall plan by the federal land management agency to reduce off-road vehicle emissions to the maximum extent possible. Such plans should include options to reduce greenhouse gas emissions from discretionary recreational off-road vehicle use to, at a minimum, 1990 levels by 2020.

Off-road vehicle destruction in the Mojave Desert. Besides creating ugly tracks like these, California off-road vehicles together emit as much carbon dioxide as 42,000 passenger vehicles driven for a year.
Photo by Perry Hoffman

• The Department of Motor Vehicles should cap the number of registrations issued for off-road vehicles in California.

The Department of Motor Vehicles should cap the number of registrations issued for off-road vehicles in California. The cap should be scaled to achieve, at least, a reduction of emissions to 1990 levels by 2020. Because registration enforcement is currently lax, additional resources may be required for effective enforcement.

Also, the California Air Resources Board should immediately address the adverse public health effects and climate implications of non-conforming off-road vehicles.

• The California Air Resources Board should eliminate loopholes that allow continued use of polluting off-road vehicles that fail to meet state emission standards.

Just as California does not allow the continued use of automobiles that do not meet state emission standards, the state should not allow the use of off-road vehicles that do not comply with state standards. The California Air Resources Board should eliminate the "red-sticker" loophole that allows continued use of polluting off-road vehicles.

• The California Air Resources Board should disallow continued or expanded off-road vehicle use on federal lands in areas that do not meet air quality standards.

California must certify that proposed land uses on federal lands conform to the state's enforcement of the Clean Air Act. To date, the state regularly approves these uses — even in non-conforming areas like Imperial County — without significant evaluation. The California Air Resources Board should reject proposals to continue or expand off-road vehicle use on federal lands in areas that do not meet air quality standards.

Dust plume from off-road vehicle staging. Meeting California's ambitious goals of reducing greenhouse gas emissions means that *all* emissions sources must be addressed.

Photo courtesy Community ORV Watch

Global Climate Change: Overall Impacts and California's Response

In 2007, the Intergovernmental Panel on Climate Change once again warned that human-induced global warming is already causing physical and biological impacts worldwide.[1] Global temperatures have already risen by 0.74°C (1.3°F) over the past century, and the rate of warming in the last 50 years was nearly double the rate observed over the last 100 years.[2] Temperatures are certain to go up even further in the future, and the most recent scientific work demonstrates that climate changes are occurring earlier and more quickly than expected.

Fossil fuel combustion is producing a critical mass of greenhouse gases that has already shifted the planet's climate system into new and dangerous territory. The impacts of this shift are already apparent and are predicted to intensify.

On a global level, we are seeing and will continue to see increases in average air and ocean temperatures, widespread melting of snow and ice, and rising mean sea levels. On continental, regional, and ocean-basin scales, numerous long-term changes in climate have also been observed. These include loss of Arctic ice and habitat, loss of Antarctic ice, melting of glaciers and related glacial-lake outburst flows, loss of snowpack in California and elsewhere, changes in precipitation patterns, increased hurricane intensity, sea-level rise and coastal flooding, public health harms such as increased heat-related illness and smog, harm to habitats, widespread species extinction, and the potential for substantial social upheaval resulting from significant environmental changes. Further, there will continue to be warming due to the amount of heat-trapping greenhouse gases already in the air, even if we completely stop new emissions immediately.[3]

Very few species will escape the burn of climate change. A landmark study surveying 20 percent of the Earth's land area offered a stark prediction: 35 percent of species will be committed to extinction by the year 2050 if greenhouse gas emission trends continue.*

What does this temperature change mean for California? The California Climate Change Center has evaluated the present and potential future impacts of climate change to the state and demonstrated that climate change poses enormous risks to California.[4] Predicted impacts to the Golden State include:

- A six- to 30-inch rise in sea level, leading to increased coastal flooding.
- A 200- to 400-percent increase in the number of heat-wave days in major urban centers.
- An increase of up to 53 percent in wildfire risk.
- An increase in storm intensity, precipitation, and the proportion of precipitation as rain versus snow.
- A 30- to 90-percent reduction of the Sierra snowpack during the next 100 years, as well as earlier melting and increased runoff.
- An increase in the number of days conducive to ozone (O_3) formation.
- Profound, and potentially catastrophic, impacts to ecosystems and species, including changes in the timing of life events, shifts in range, and community-abundance shifts.[5]

*C.D.C Thomas et al., "Extinction risk from climate change," *Nature* 427 (2004):145-148.

Curbing greenhouse gas emissions to limit the effects of climate change in California and the world is one of the most urgent challenges of our generation. Recent peer-reviewed works emphasize the urgent need to reduce greenhouse gas emissions immediately: Just ten more years of "business-as-usual" emissions may commit us to climate feedbacks and impacts that would entirely transform the planet as we now know it.[6] As noted in a report commissioned by the California Environmental Protection Agency:

Because most global warming emissions remain in the atmosphere for decades or centuries, the choices we make today will greatly influence the climate our children and grandchildren inherit. The quality of life they experience will depend on if and how rapidly California and the rest of the world reduce greenhouse gas emissions.[7]

In response to this monumental threat, in 2006, the California legislature passed the Global Warming Solutions Act, known as Assembly Bill 32, which requires the state air resources board to limit statewide greenhouse gas emissions by 2020 to 1990 levels.[8] Assembly Bill 32 recognizes California's leadership in furthering environmental protection. Despite leading the nation in energy efficiency, the state of California — compared to entire nations — remains the 12th-largest emitter of greenhouse gases worldwide.

Under Assembly Bill 32, the California Air Resources Board must establish rules and regulations to achieve the maximum technologically feasible and cost-effective greenhouse gas emission reductions from any "greenhouse gas emission source." This is defined in the statute as any "source, or category of sources, of greenhouse gas emissions whose emissions are at a level of significance, as determined by the state board, that its participation in the program established under this division will enable the state board to effectively reduce greenhouse gas emissions and monitor compliance with the statewide greenhouse gas emissions limit."[10]

The California Air Resources Board is currently in the process of crafting the rules and regulations in an effort to meet its goal of cutting greenhouse gas emissions to 1990 levels by 2020. This will require an approximately 29-percent reduction from a business-as-usual approach.

A primary focus of efforts to curb greenhouse gas emissions is likely to remain on passenger vehicles, which includes the sedans, trucks, sport utility vehicles, and mini-vans that most of us drive to work, school, or the grocery store every day. But while passenger vehicles contribute the majority of greenhouse gas emissions, off-road vehicles emit the same greenhouse gases as passenger vehicles and have even more detrimental impacts on human health. Limiting off-road vehicle emissions will help the state meet its goal of reducing greenhouse gas emissions while simultaneously protecting public health. Regulation of emissions from off-road vehicles must be a priority for the California Air Resources Board as it implements Assembly Bill 32.

Dust from off-road vehicles. Off-road vehicle dust can disperse harmful air contaminants well beyond a designated off-road vehicle-use area.

Photo by Kevin Emmerich

The Significant Greenhouse Gas Emissions of California Off-road Vehicles

In 2006, the Off-Highway Motor Vehicle Recreation Division of California's Department of Parks and Recreation commissioned a survey to estimate fuel usage by off-road recreation in California.[11] The survey concluded that overall use of off-road vehicles on public lands consumes more than 26 million gallons of gasoline each year in California (Figure 1).[12] This equates to more than 500,000 barrels of oil. The gasoline consumption from off-road vehicle use in California is equivalent to the gasoline consumed by more than 1.5 million passenger vehicles driving from San Francisco to Los Angeles.[13]

Figure 1. Estimated Gallons of Gasoline Used by Off-road Vehicles on Public Lands in California, April 2004 – March 2005

VEHICLE TYPE*	MEAN	LOWER BOUND	UPPER BOUND
Registered off-road vehicles	20,014,590	17,081,031	22,948,148
Illegal, unregistered off-road vehicles	6,207,327	4,186,151	8,218,148
Total	**26,221,917**	**21,267,182**	**31,166,650**

Source: California Department of Parks and Recreation, *Estimating the State Fuel Tax Paid on Gasoline Used in the Off-highway Operation of Vehicles for Recreation*, September 2006

***Registered off-road vehicles** include dirt bikes, all-terrain vehicles, snowmobiles, and dune buggies that have been legally registered with the state.

Illegal, unregistered off-road vehicles include dirt bikes, all-terrain vehicles, snowmobiles, and dune buggies that

The Environmental Protection Agency standard estimation is approximately 8,800 metric tons of carbon dioxide from each million gallons of gasoline burned.[14] By this estimate, annual emissions from California off-road vehicle use equal 230,000 metric tons of carbon dioxide. This equates to more than 500 million pounds of carbon dioxide emissions each year. Emissions from current off-road vehicle use statewide are equivalent to the carbon dioxide emissions from 42,000 passenger vehicles driven for an entire year or the electricity used to power 30,500 homes for one year.

Worse, the figure used here does not include emissions from travel to and from off-road vehicle recreation sites, which is likely substantial. According to a comprehensive survey of recreation in California, the mean travel time to a recreation area is 45 minutes.[15] Many off-road vehicle recreation sites are remote from urban population centers, leading to even longer travel times. Further, the trucks used to tow off-road vehicles often have very low fuel efficiency, leading to increased emissions. When emissions from travel to and from off-road vehicle recreation sites are considered, total greenhouse gas emissions from off-road recreation are likely to be much higher.

In addition, off-road vehicle recreation consumes 5.5 million gallons of diesel fuel each year,[16] and although diesel engines are generally more fuel efficient than gasoline

Figure 2. Annual Recreational Gasoline Usage by Vehicle Type

VEHICLE TYPE	GALLONS OF FUEL USED
Off-road motorcycles	10,003,506
Off-road all-terrain vehicles	12,013,896
Off-road four-wheel vehicles	2,658,841
Snowmobiles	1,444,087
Other off-road vehicles	101,585
Total gasoline usage	**26,221,915**

Source: California Department of Parks and Recreation, *Estimating the State Fuel Tax Paid on Gasoline Used in the Off-highway Operation of Vehicles for Recreation*, September 2006 engines, they emit 25 to 400 times the amount of particulate black carbon and organic matter (soot) than gas-burning vehicles.[17] The warming from soot may offset any benefits from reduced carbon dioxide emissions, and scientists have increasingly focused on the need to control black carbon in conjunction with carbon dioxide reductions in order to slow global warming.[18]

The Continued Growth of Off-road Vehicle Emissions in California

Transportation is the largest single contributor of greenhouse gases in California, accounting for 38 percent of the state's total greenhouse gas emissions.[19] Off-road vehicle emissions account for a small but significant fraction of the state's overall greenhouse gas emissions, and emissions from this sector, if left unchecked, will continue to grow.

Motorcycle ascending scarred hillside in Jawbone Canyon, California. California off-road motorcycles together release more emissions than all other types of off-road vehicles in the state.
Photo by Howard Wilshire

Because only a small fraction of the population — about 15 percent — participates in off-road vehicle recreation, reductions in use will have no impact on a majority of Californians.[20] And because recreational off-road vehicle use is entirely discretionary, reductions in this source to levels at or significantly below 1990 levels may be used to offset other sources that are less discretionary or that involve higher costs. In a survey of Californians, walking was the activity with the highest participation percentage (91 percent) and trail hiking ranked ninth out of 55 (69 percent), while driving four-wheel-drive vehicles ranked 31st (19 percent) and riding all-terrain vehicles and dirt bikes ranked 38th (17 percent).[21]

Finally, as described in greater detail below, reducing greenhouse gas emissions from off-road vehicles will have important public health benefits for all Californians. It is only fair that reductions in emissions associated with an optional recreational pursuit contribute towards meeting the state's greenhouse gas emissions reductions targets. Meeting the state's ambitious goals of reducing greenhouse gas emissions means that all emissions sources must be addressed, and the Air Resources Board must acknowledge this fact by addressing the emissions associated with off-road vehicles.

Motorcycle in dune recreation area. Off-road motorcycles released an astounding average of 143 tons of emissions per day in 2006. (California Air Resources Board, http://www.arb.ca.gov/app/emsinv/emssumcat.php.)

Photo by George Wuerthner

Reducing Off-road Emissions by Reducing Overall Usage

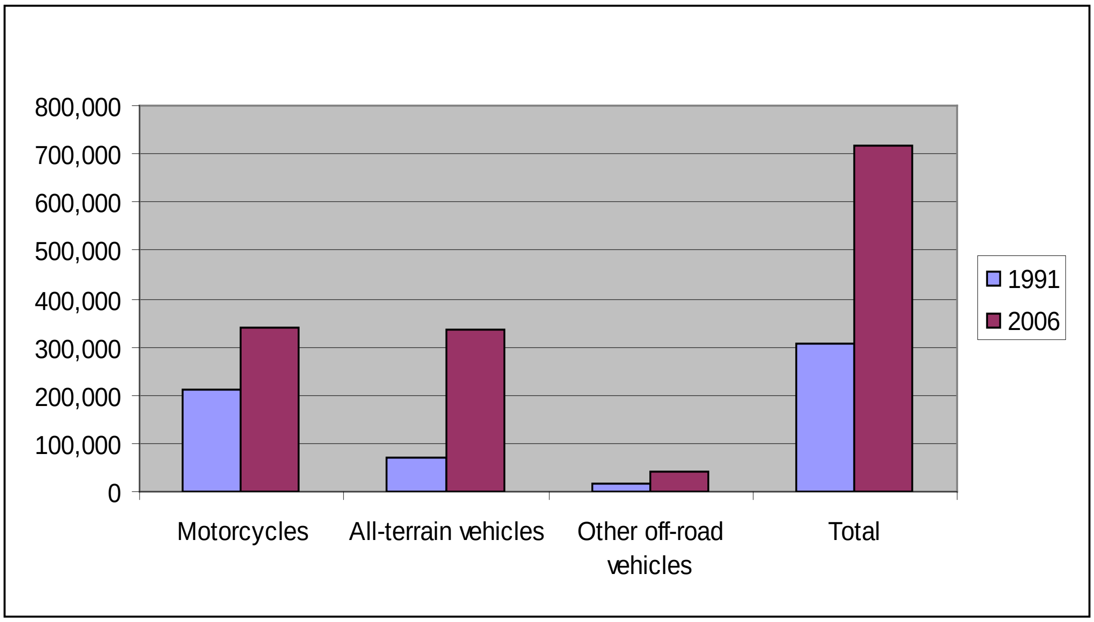
There are currently no regulations directly addressing the greenhouse gas emissions of off-road vehicles in California. In 2002, the United States Environmental Protection Agency issued final regulations setting new standards for emissions from off-road vehicles and snowmobiles.[22] However, this rule focused on carbon monoxide, nitrogen oxides, and volatile organic gases, and did not regulate greenhouse gas emissions. In 2004, the California Air Resources Board adopted regulations to comply with Assembly Bill 1493, California's Clean Vehicle Law, which commits the state to achieving the maximum feasible and cost-effective reduction of greenhouse gas emissions from passenger cars and light trucks sold in California. However, the Bush administration has so far blocked these regulations by refusing to provide Environmental Protection Agency approval. Most recently, the State of California petitioned the federal government for rule-making to address the greenhouse gas emissions from all non-road vehicles, including off-road vehicles,[23] but the Bush Administration is not expected to act on this petition.

Dust plume from off-road vehicle staging near public lands. Evidence shows that coarse particle pollution, most associated with off-road vehicles, is detrimental to human health.

Photo courtesy Community ORV Watch

While more efficient vehicles would produce less greenhouse gas per miles traveled, efforts to increase efficiency would generally apply only to new vehicles and would therefore fail to address the greenhouse gas emissions of all of the off-road vehicles already in use in California. At the same time, the use of off-road vehicles in California continues to increase. Registrations of all-terrain vehicles, snowmobiles, dune buggies, sand rails, and dirt bikes in California have more than doubled in the last 20 years.[24] In addition, there has been a 74-percent increase in street-licensed four-wheel-drive vehicles in California since 1994, and a more than 60-percent increase in the sale of sport-utility vehicles in the state from 1996 to 2002 (Figure 3).[25] Furthermore, California contains more than 1.1 million legally registered and illegal, unregistered off-road vehicles, and millions more sport-utility vehicles and motorcycles that are driven off road.[26]

Figure 3. Increase in Off-road Vehicle Registration, 1991-2006



Source: California Department of Parks and Recreation, *Estimating the State Fuel Tax Paid on Gasoline Used in the Off-Highway Operation of Vehicles for Recreation*, ICF International, September 2006, at 5-20; Memorandum from Department of Transportation to State Controller's Office, June 9, 1992

All told, the large number of off-road vehicles already in use in California, coupled with the expected increase in the number of users, makes it highly unlikely that higher efficiency requirements for new off-road vehicles alone could bring about a decrease in greenhouse gases. In addition, considering the ongoing political obstacles to regulations to increase vehicle efficiency, efforts to reduce greenhouse gas emissions from off-road vehicles as a group must focus on measures to limit their use and proliferation.

The Serious Public Health Effects of Off-road Vehicle Emissions

Off-road vehicles are typically powered by two-stroke engines that are highly inefficient and produce relatively high emissions of gases that harm the environment and can adversely affect human health.[27] The pollutants released in off-road vehicle exhaust include carbon monoxide, ozone, hydrocarbons, oxides of nitrogen and sulfur, and particulate matter.[28] Kasnitz and Maschke report: "One two-stroke off-road motorcycle or all-terrain vehicle emits as much hydrocarbon pollution per mile as 118 passenger cars, while relatively cleaner four-stroke engines still emit more than seven times the level of carbon monoxide as new cars."[29] Other studies report similar results.[30]

According to the Environmental Protection Agency, recreational vehicles account for nearly 10 percent of national mobile-source hydrocarbon emissions and about 3 percent of national mobile-source carbon monoxide emissions. If left uncontrolled, by 2020, these engines will contribute 33 percent of national mobile source hydrocarbon emissions, 9 percent of carbon monoxide emissions, 9 percent of oxides of nitrogen emissions, and 2 percent of particulate matter emissions.[31]

Dirt bike in all-terrain vehicle park. On an hour-by-hour basis, a motorcycle can emit as much pollution as more than 30 automobiles.

Photo by Laurel Hagen

On an individual basis, these vehicles have very high pollution rates. A two-stroke all-terrain vehicle or motorcycle can emit as much pollution (hydrocarbons, carbon monoxide, and nitrogen oxides) in one hour as more than 30 automobiles operating for one hour, and a snowmobile can emit as much as nearly 100 automobiles.[32] This pollution from emissions of hydrocarbons, carbon monoxide, and nitrogen oxides — as well as particulate matter — has been linked to respiratory disease, cancer, and premature death.[33] Pollution from off-road vehicles in California has continued to rise over the last several decades (Figure 4).

Ozone

Ground-level ozone, the primary and most health-damaging component of smog, is a toxic gas formed from ozone precursors including industrial emissions and gasoline vapors and can affect health even when found in small amounts. According to the California Air Resources Board, off-road motorcycles and all-terrain vehicles produce 118 times as much smog-forming pollution as modern automobiles on a per-mile basis.[34]

Figure 4. Increase in California Off-road Vehicle Pollution, 1990-2006

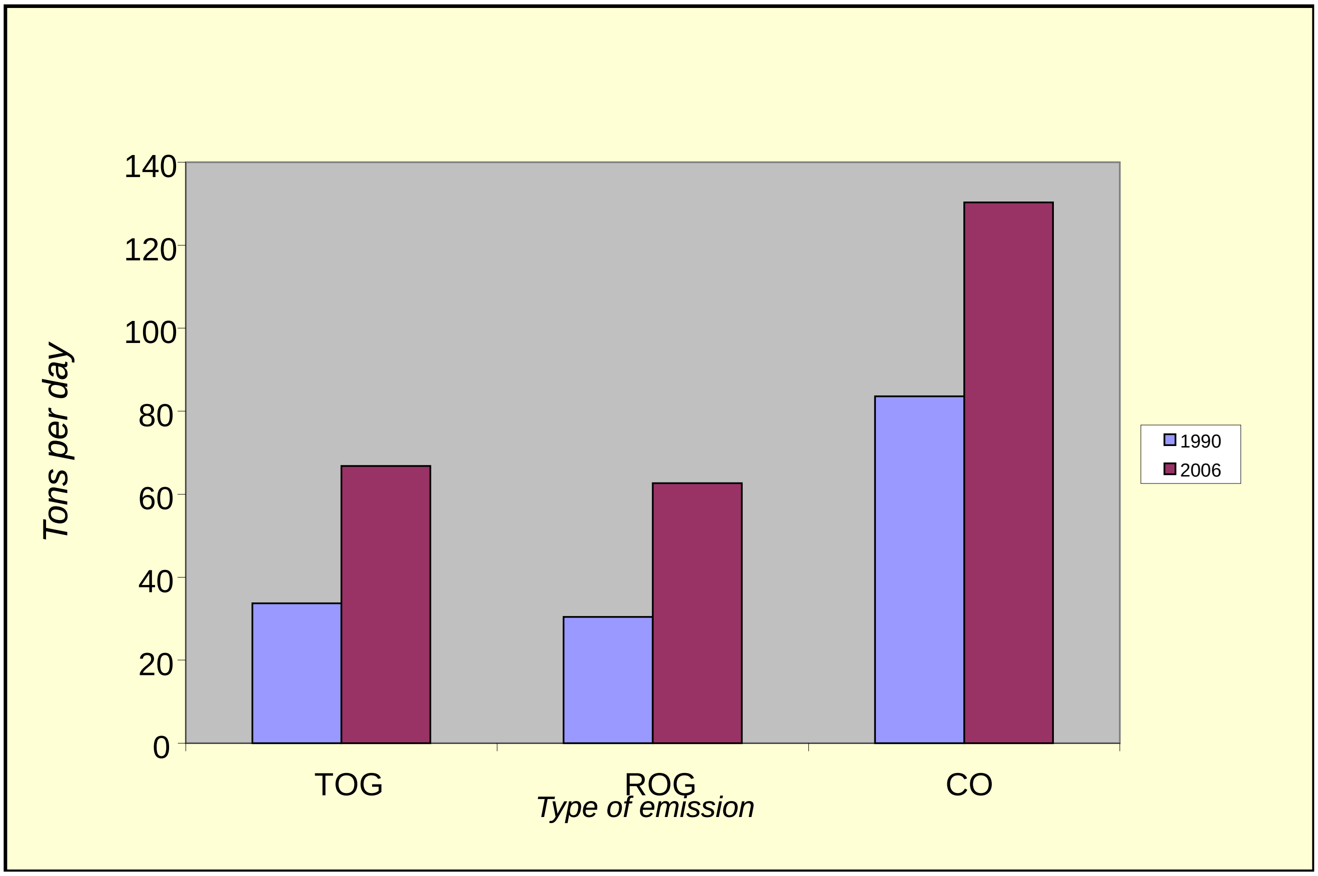


Figure 4: In California, emissions from off-road vehicles (all-terrain vehicles, dirt bikes and snowmobiles) of total organic gases (TOG) and reactive organic gases (ROG) have approximately doubled in the last 15 years while carbon monoxide (CO) emissions have shown a 56-percent increase.[43] Some of these pollutants are precursors to other pollutants. For example, oxides of nitrogen and reactive organic gases are precursors to ground level ozone and other greenhouse gases. Data from California Air Resources Board, http://www.arb.ca.gov/app/emsinv/emssumcat.php.

Ozone is a respiratory irritant and increased concentrations have been associated with reduced lung function and increased hospitalizations for asthma, especially for children or those with compromised respiratory systems.[35] Ozone can also have detrimental impacts on healthy populations. Studies of two healthy groups, outdoor postal workers in Taiwan and college freshmen who were lifelong residents of Los Angeles or the San Francisco Bay area, found that exposure to elevated ozone decreases lung function,[36] and chronic exposure may cause permanent lung damage.[37] Ozone has been linked to increased hospital admissions for respiratory conditions including respiratory infection, asthma, chest pain, cough, and significant decreases in lung function.[38]

Elevated ozone concentrations pose a serious health concern. The American Lung Association reports that one-third of the U.S. population lives in areas with unhealthy levels of ozone nationwide.[39] One in three Americans lives in a county where the monitored air quality places them at risk for decreased lung function, respiratory infection, and lung inflammation.[40] California is home to 13 of 20 counties nationwide where residents are at the greatest risk from ozone pollution.[41] This includes the six counties most at risk nationwide from ozone pollution: San Bernardino, Kern, Riverside, Los Angeles, Tulare, and Fresno.[42] Many of these counties contain popular off-road vehicle areas like San Bernardino County's Johnson and Stoddard valleys and Dumont Dunes (among many others).

According to the California Air Resources Board, dirt bikes and all-terrain vehicles produce 118 times as much smog-forming pollutants as cars.

Particulate Matter

The subset of particulate matter known as PM_{10} consists of fine particulate matter of 10 microns or less that is a mixture of airborne solid particles and liquid droplets from both man-made and natural sources. It is generally caused by wind-blown sources of dust or the interaction of sulfur oxides, nitrous oxides, and volatile organic compounds. Particle air pollution is the biggest and most pervasive air pollution risk humans face.[43] Particulate matter can be emitted directly into the atmosphere by combustion sources, including off-road vehicles, or it can be created by the combination of gases such as nitrous oxide and sulfur dioxide, both of which are also released by off-road vehicles. Like ozone and carbon monoxide, nitrogen oxides and sulfur dioxide are associated with decreased lung function.[44] When inhaled, particulate matter irritates the respiratory tract.[45] Due to the small size of some particles, they are easily inhaled and can lodge in the lungs, causing respiratory and cardiovascular health consequences, as well as increased hospital admissions of the elderly and children when particulate-matter levels increase.[46, 47]

Dust is also a component of particle pollution, making unpaved roads the largest single source of particulate matter.[48] Off-road vehicles disturb soil crusts, crush soil, and generate wind that results in the creation and release of dust into the air. Because wind can disperse suspended particulates over long distances, dust raised by off-road vehicle traffic can disperse contaminants carried by dust well beyond a given off-road vehicle-use area. In 1973, for example, satellite photos detected six dust plumes in the Mojave Desert covering more than 656 square miles, all attributable to off-road vehicle activities.[49]

Particle pollution is a significant threat nationwide. The American Lung Association reports that one in three people in the United States lives in an area where they are subject to short-term exposure to particle pollution, while one in five people lives in an area where they are subject to exposure to unhealthy year-round levels of particle pollution.[50] Even at low levels, exposure to particles over time can increase risk of hospitalization for asthma, damage to the lungs, and — most significantly —the risk of premature death.[51]

Particle pollution is particularly serious in California when compared to other states. According to the Environmental Protection Agency, 16 California counties exceed accepted levels of particulate matter.[52] In fact, the state is home to four of the five most polluted counties nationwide for both short-term and year-round particle pollution.[53]

While the health affects associated with particulate matter are especially severe for fine particles ($PM_{2.5}$), there is evidence that coarse particle pollution (PM_{10}), most often associated with off-road vehicles, is also detrimental to health. Studies have found that for each 10 microgram-per-cubic-meter increase in PM_{10}, there was a 1-percent increase in hospital admissions for cardiovascular disease, and about a 2-percent increase in admissions for pneumonia and chronic obstructive pulmonary disease. Investigators concluded that their analysis provided "new and strong evidence" linking PM_{10} air pollution to adverse health effects.[54]

Another study reported that deaths from respiratory diseases were associated with PM_{10} and total suspended particulates. They found that relative risks for coarse particles were similar to those for fine particles and even higher in the case of ischemic heart disease and stroke. The authors concluded that "the finding of elevated and significant effects for $PM_{10-2.5}$ suggests that there may still be a rationale to consider the health effects of the coarse fraction as well as the fine fraction of particulate matter."[55]

Other studies support the idea that coarse particles contribute to respiratory diseases and cardiovascular hospitalizations.[56]

Although many peer-reviewed studies have examined the effects of particulate matter on health, relatively few have specifically addressed coarse particles, and those that have often focus on short-term exposures. The impacts of long-term exposure to coarse particles is an area in which more research is likely needed.

Carbon Monoxide and Oxides of Nitrogen

In addition to its serious impacts on the environment, carbon monoxide poses serious health risks because it strongly binds to hemoglobin in the blood, thereby reducing the amount of oxygen that reaches the organs. Exposures to low levels affect the most oxygen-sensitive organs of the body — the heart and the brain — and can result in fatigue, angina, reduced visual perception and dexterity, and even death. Further, though not a greenhouse gas itself, carbon monoxide can increase the lifespan of greenhouse gases, increase the production of ground-level ozone, and worsen climate change.[57] Transportation accounts for the majority of carbon monoxide released nationwide and in 2000, the Environmental Protection Agency determined that recreational vehicles cause or contribute to ambient carbon monoxide in more than one nonattainment area, including Los Angeles.[58]

In 2001, the Environmental Protection Agency found that all-terrain vehicles, a subset of off-road vehicles, emit more than 381,000 tons of hydrocarbons, 1,860,000 tons of carbon monoxide, and 11,000 tons of oxides of nitrogen each year across the country.[59] The emissions of oxides of nitrogen alone are equivalent to the annual greenhouse gas emissions from 566,575 passenger vehicles.[60] The Environmental Protection Agency has adopted National Ambient Air Quality Standards for some air pollutants that are of particular concern from a health perspective — including particulate matter, nitrogen oxide, carbon monoxide, sulfur dioxide, and ozone — which define maximum concentrations of these substances that are allowed in the air. However, many areas in California are not yet in compliance with these standards.[61]

The Environmental Protection Agency estimated that its 2002 rules regulating emissions from off-road vehicles and snowmobiles would avoid 1,000 premature deaths, prevent 1,000 hospital admissions, reduce 23,400 cases of asthma attacks, and reduce 200,000 days of lost work.[62] It is estimated that these health benefits will equal a total of $8 billion in 2030.[63]

Still, even with the new regulations, unhealthy emissions from all types of recreational vehicles continue to increase in California (Figure 6). By regulating emissions from these vehicles, California will help protect the health of its residents.

Figure 6. Increase in Pollution by Vehicle Type in California, 1990-2006

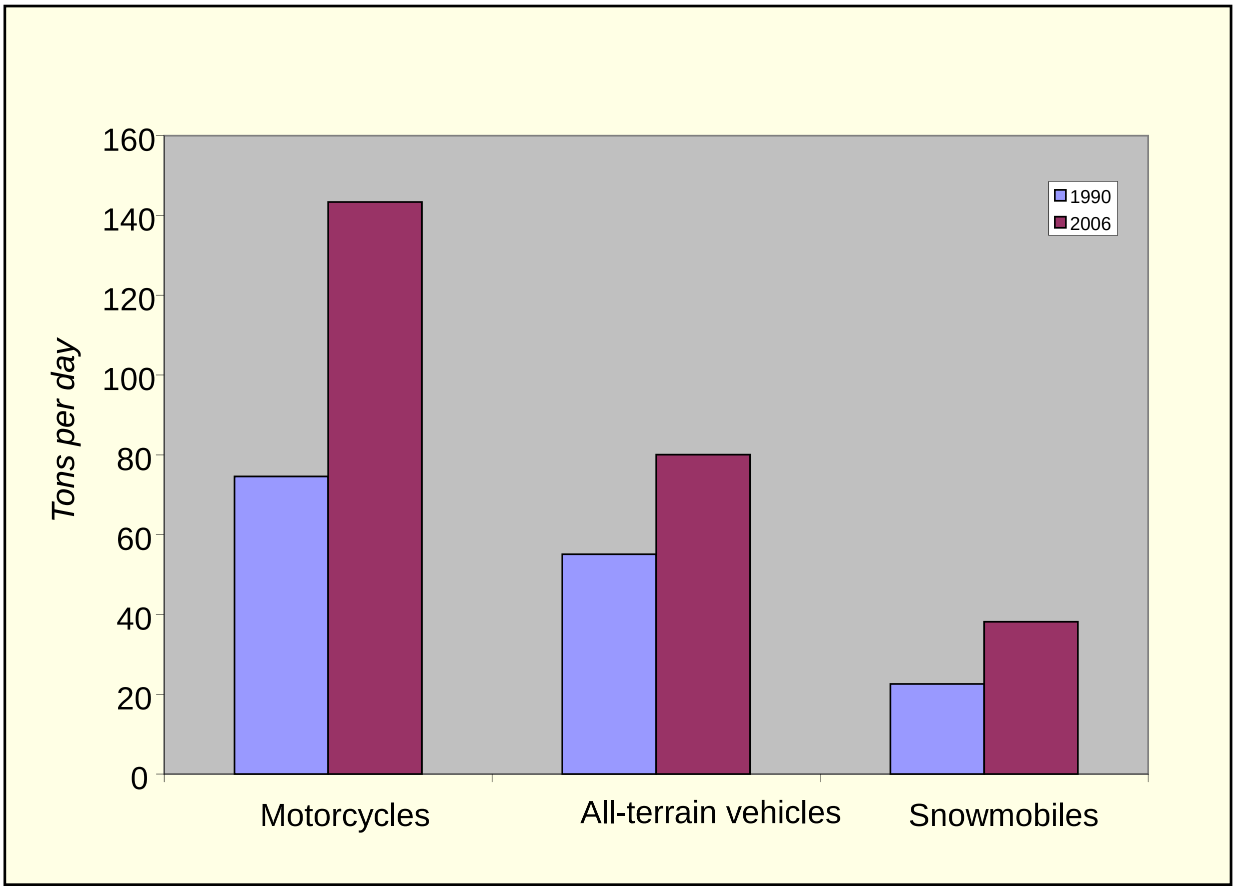


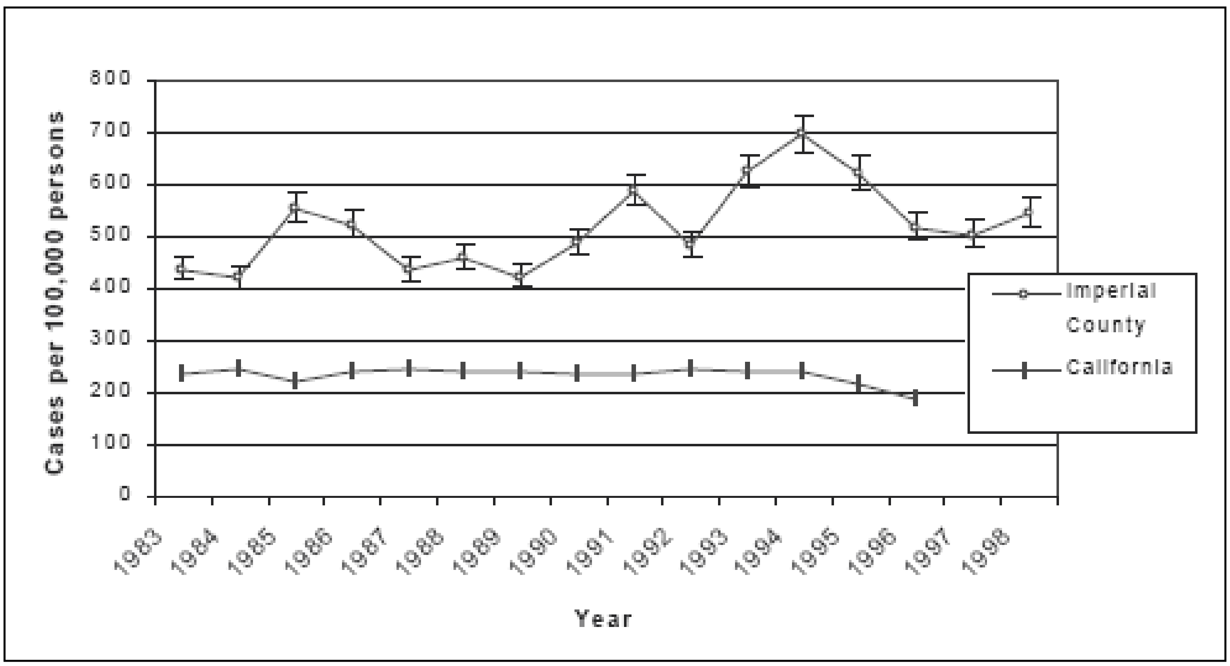
Figure 6: Each type of off-road vehicle showed an increase in total emissions (ROG, TOG, CO, NOx and SOx) over the past 15 years. Off-road motorcycles (dirt bikes) release the most, averaging about 143 tons (equivalent to the weight of 103 Toyota Priuses) of emissions per day in 2006. This was nearly double the average emissions (an increase of 95 percent) from dirt bikes in 1990. Over the same time period, all-terrain vehicles had an approximately 45-percent increase in total emissions, while snowmobiles had a 72-percent increase. Regulations require that the state of California cut overall greenhouse gas emissions to return to 1990 levels by the year 2020. Data from California Air Resources Board, http://www.arb.ca.gov/app/emsinv/emssumcat.php.

Case Study: Imperial County

Imperial County in southern California covers more than 4,597 square miles, bordering on Mexico to the south, Riverside County to the north, San Diego County on the west, and the state of Arizona to the east. The region currently exceeds federal standards for the particulate matter PM_{10} and both federal and state standards for ozone, and it has exceeded federal and state standards for both pollutants since 1996.[64,65,66] Local surveys report that some locations measure more than 10 times the maximum allowable federal standard for particulate matter and that Imperial County suffers from the worst particulate air pollution in California.[67] In fact, particulate matter concentrations in Imperial Valley have been measured at double the level deemed by the Environmental Protection Agency to cause significant harm to health.[68] The American Lung Association gave Imperial County an "F" for its failure to meet ozone standards and a "D" for its performance in terms of particle pollution.[69]

As described above, the adverse health effects from particulate matter and ozone pollution are severe — and their impacts on Imperial County's residents are readily apparent. Asthma is a serious problem, and Imperial has the highest child asthma rate of any county in California (Figure 7).[70] Asthma rates in Imperial County increased by 59 percent from 1983 to 1994. The county's maximum ozone levels increased by 64 percent, and particulate-matter

Figure 7. Age-adjusted Childhood Asthma Hospitalization Rates and 95-percent Confidence Intervals for Imperial County and California, 1983-1998



This graph shows the childhood asthma rate intervals for Imperial County and California from 1983 to 1998. Overall, the state's rate is fairly constant and is much lower than Imperial County's, which shows much more fluctuation and an overall upward trend. The statewide rate is decreasing; however, the county's rate is once again on the rise at the end of this study period and to the present.

Source: Imperial County Public Health Services levels are four times higher in Imperial than in neighboring San Diego County.[71]

The California Department of Public Health Services recently found that Imperial County has the highest asthma hospitalization rates in the state for all race/ethnicity groups among all ages and for most race/ethnicity groups among children.[72] Rates of respiratory diseases continue to worsen.[73] Air pollution is blamed as a contributor to the high rates of asthma, bronchitis, pneumonia, and allergies in this region, especially among children between the ages of one and 14 years.[74] Children are especially at risk, as are the elderly, asthmatics, and those with chronic pulmonary disease or heart disease (Figure 7).

Off-road vehicle use on public lands in Imperial County is a major contributor to the county's air quality problems. In fact, the federal Bureau of Land Management has stated that off-road vehicles are one of the county's most significant sources of the harmful pollutants ozone, oxides of nitrogen, carbon monoxide, and particulate matter.[75] Off-road vehicle emissions also contribute to the county's increased levels of reactive organic gases.

Figure 7. Imperial County Public Health Statistics

CONDITION	NUMBER OF CASES
Pediatric asthma	4,201
Adult asthma	7,813
Chronic bronchitis	4,335
Emphysema	1,731
Cardiovascular disease	31,151
Diabetes	7,437
Total population with any of above conditions	**155,823**
Population younger than 18	47,199
Population 65 and older	**16,035**

Source: American Lung Association, *State of the Air: 2007*

Still, federal and state agencies continue to encourage off-road vehicle use throughout Imperial County. On holiday weekends, the Imperial Sand Dunes Recreation Area, run by the federal Bureau of Land Management, can be used by hundreds of thousands of off-road vehicle users. Other popular federal off-road vehicle areas include Superstition, Plaster City, Heber Dunes, and parts of the California Desert Conservation Area. State-run areas allowing off-road vehicle use include the Ocotillo Wells State Vehicular Recreation Area on the border of San Diego and Imperial counties, Desert Cahuilla, and portions of Anza-Borrego Desert State Park.

The high concentration of off-road vehicle use in Imperial County, coupled with the poor public health of its residents — which studies partially correlate to air pollution — implies that there is a need for further research. This research should focus on the contribution of off-road vehicles to pollution in the county and should seek to parse out the impacts that off-road vehicle pollution is having on poor public health. In the meantime, considering Imperial County's record-high childhood-asthma rates together with its massive off-road vehicle use — and the severe health implications of its violation of federal and state air-pollution standards — isn't it time for the state and federal governments to rein in anything that may be contributing to these increased levels, including off-road vehicle pollution?

Off-road Vehicles' Exemption From California Emission Standards

In the 1990s, the California Air Resources Board attempted to address the air-quality impacts of recreational pollution by adopting emission-control regulations for new off-road recreational vehicles, including off-road motorcycles (dirt bikes) and all-terrain vehicles.[76] The regulations require that all off-road recreational vehicles sold in California, model year 1998 and later, are certified by the Board to meet state emissions standards.[77] But manufacturers and off-road vehicle groups, while initially supportive, soon balked at the new regulations, claiming that the requirements decrease off-road vehicle sales.[78] Off-road vehicle user groups and industry representatives mounted an intense lobbying campaign urging the Board to weaken the new regulations.

Red stickers allow off-road vehicles that do not meet state emission standards to be used throughout much of California for most of the year.

In 1998, the California Air Resources Board succumbed to industry pressure and approved amendments to the new emission regulations that allow the continued operation of especially polluting off-road vehicles.[79] This clause distinguished types of off-road vehicle registration based on compliance (or noncompliance) with California's exhaust emission standards. Emission-compliant dirt bikes and all-terrain vehicles were (and still are) eligible for a "green-sticker" registration that allows them to be operated year round. Noncompliant dirt bikes and all-terrain vehicles were (and still are) eligible for a "red-sticker" registration and are subject to usage restrictions Despite violating emissions standards, polluting "red-sticker" vehicles may still be ridden in many places during many months of the year (Appendix A).[80] A red sticker merely limits recreational use in certain places to those months of the year determined by the California Air Resources Board to have the lowest levels of ozone pollution — mainly, the months of fall, winter, and spring. To make matters worse, the California Air Resources Board grandfathered in all off-road vehicles manufactured before 2003. A press release from California State Parks explains: "Because of the confusion as to which vehicles required which stickers … to start with a clean slate, the DMV will provide Green Stickers to all 2002 model year and older OHVs, regardless of emission standards."[81]

Instead of re-evaluating each vehicle to ensure compliance, the Board revised its regulations once again so that all 2002 model year and older off-road vehicles would receive green stickers, even if these same vehicles had previously been certified as noncompliant based on their emissions.

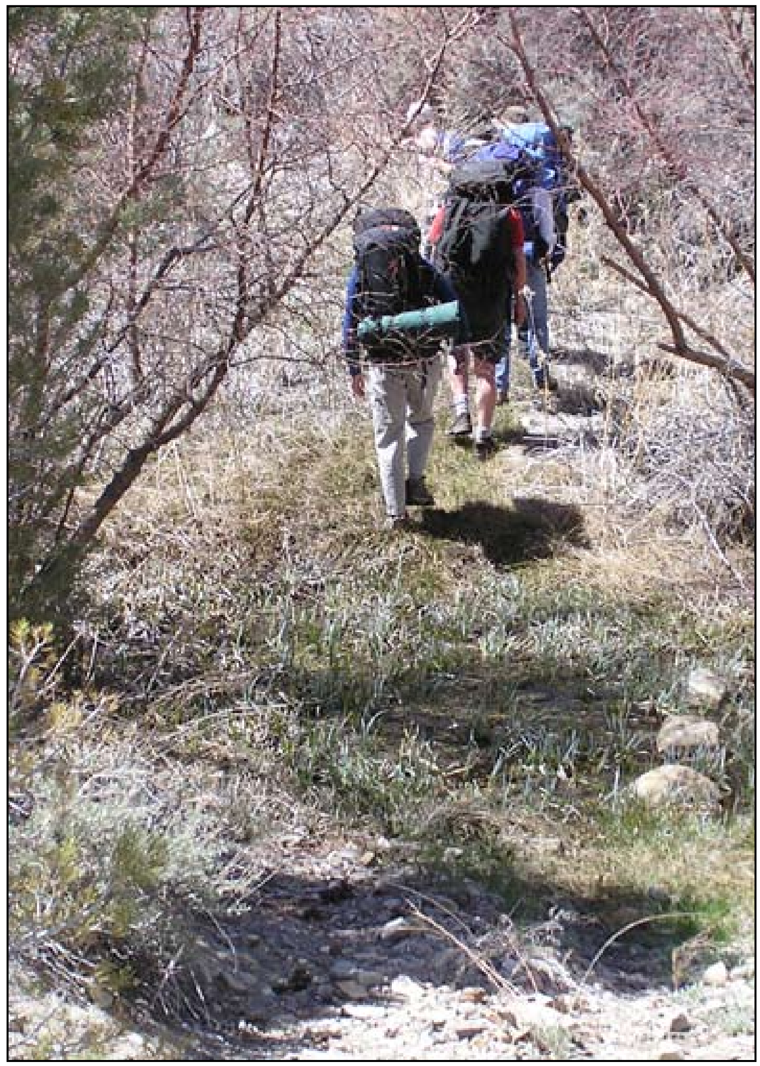
To date, off-road vehicles that do not comply with state emission standards may still be sold in the state and used throughout much of the year in California, creating a loophole in the state's emissions regulations that undermines its commitment to protecting the public health of its residents.

Biological and Cultural Benefits of Limiting Off-road Vehicle Use

The impacts of off-road vehicles on the environment have been well documented. Off-road vehicle use impairs water quality, degrades wildlife habitat, threatens California's archeological heritage, and destroys the peace and quiet that Californians want and expect from the great outdoors.[82]

Encouraging Quiet Outdoor Recreation

Most people visit the outdoors seeking peace and quiet, wanting to recharge their batteries by taking a break from the ever-increasing pace of modern life. Walking, hiking, wildlife viewing, camping, and picnicking are among the most popular outdoor recreation activities of Californians,[83] while off-road vehicle use, riding dirt bikes, and snowmobiling are among the least popular.[84] Muscle-powered recreationists, including hunters, anglers, bird watchers, cross-country skiers, and hikers — which make up the largest user group on public lands and overall in California — are losing access to places where ecological integrity is intact and quiet prevails. Reducing off-road vehicle traffic would mean more opportunities for quiet recreationists to enjoy peaceful and undisturbed experiences in nature.



Hikers near Furnace Creek. Hiking ranked ninth out of 55 in a study of the most popular outdoor activities in California. Off-road vehicle use ranked 38th.

Photo by Daniel Patterson

Improving Water Quality

Off-road vehicle use near streams, rivers, and lakes can degrade water quality, both negatively impacting the creatures who live there and creating a serious threat to the quality of our drinking water.[85]

Off-road vehicles expel 20 to 30 percent of their oil and gasoline unburned, releasing it into air and water.[86] With off-road vehicle use exceeding 80 million visitor days in national forests alone, tens of millions of gallons of gasoline and motor oil likely enter the soils and waters of our public lands each year as a result of inefficient combustion and emissions.[87] This is significant because national forests are the largest and most important water source in the United States: more than 60 million Americans in 3,400 communities from 33 states get their drinking water from watersheds that originate on our national forests and grasslands.[88]

Areas on the Inyo National Forest and surrounding lands managed by the Bureau of Land Management show evidence of degraded water quality and habitat due to off-road vehicles. The Bureau of Land Management found that Furnace Creek in the White Mountains does not meet the Bureau of Land Management's standards for a properly functioning riparian system. They report:

Presently, portions of the Furnace Creek drainage are considered ''functional-at risk.''

Riparian-wetland areas are considered ''functional-at risk'' when an existing soil, water, or vegetation condition makes them susceptible to degradation. Presently, there are seven locations in Furnace Creek where the existing vehicle route crosses the stream. Significant erosion and sedimentation of the stream are occurring at two stream crossings. Erosion in both locations is contributing excessive sediment to the adjacent riparian area. Moreover, headcutting is forming at both locations. Headcuts are a fluvial geomorphic feature indicative of unstable conditions. The proposed closure order is consistent with protecting and restoring Furnace Creek to a "properly functioning" riparian system.[89]

Although Furnace Creek is not a key source of drinking water, it is a good example of what recreational off-road vehicle use can do to a stream and surrounding habitat. Both air and water pollutants have been shown to have measurable impacts to stream environments.[90] In addition to releasing pollutants, off-road vehicles cause erosion and sedimentation that pours dirt directly into streams and rivers, also degrading water quality and habitat for animals that are key to functioning riparian ecosystems.[91]

Another example of a degraded waterway is the Pajaro River, listed as an "impaired water body"

Mud pit created by off-road vehicles near Furnace Creek.
Besides emitting greenhouse gases, off-road vehicles can do tremendous damage to public lands.
Photo by Paul McFarland

under the federal Clean Water Act, which flows into the protected Monterey Bay National Marine Sanctuary and which faces a number of problems including erosion, sedimentation, and pollution.[92] Off-road vehicle activity has directly impacted water quality in this watershed and has exacerbated sediment migration and degraded habitat along riparian corridors and in the Clear Creek area.[93] Reducing off-road vehicle use in California has the potential to increase the quality of drinking water for Californians and the creatures that need this habitat for survival.

Reducing Wildlife Habitat Degradation

Off-road vehicle recreation has severe impacts on wildlife and habitat. It is the third-leading cause of species endangerment — behind only direct habitat destruction and invasive species — and 43 percent of California's threatened and endangered species are declining in whole or in part because of off-road vehicles, including the Peninsular bighorn sheep, Mojave fringe-toed lizard, snowy plover, and Peirson's milk-vetch.[94] A reduced number of off-road vehicles would provide these and other species an opportunity to survive, thus preserving an important part of California's natural legacy. On a larger scale, the greatest impacts of off-road vehicles to species and habitats may be the greenhouse gases that contribute to global warming.

Peirson's milk-vetch in flower. Off-road vehicle use is one of the biggest obstacles to the recovery of the threatened Peirson's milk-vetch.
Photo by Andreas Chavez

Helping Preserve Archeological Sites

California's lands are rich with cultural and archeological resources that also can be destroyed by off-road vehicles. According to the Bureau of Land Management, prehistoric sites in the California desert have been "run over and ridden through, and tires have been spun on them," leading to the degradation or complete destruction of archeological sites thousands of years old.[95] For example, "donuts" or off-road vehicle tracks, were recently found through ancient sleeping circles in the Desert Cahuilla area adjacent to Anza Borrego State Park. Not only would reducing off-road vehicle use help protect California's land, air, and water — it would also contribute to the preservation of the state's cultural heritage for future generations to enjoy.

California's Continued Support for Off-road Vehicle Use — Despite the Consequences

Currently, more than 200 private, county, state, and federal sites in California are open to off-road vehicle use, and regardless of these vehicles' significant impacts to public health, the global climate, and local ecosystems, the state continues to encourage expanded off-road vehicle recreation on public lands.

Sport utility vehicles churning up dust
Photo by Larry Hogue

Off-road Vehicle Use on State Lands

The Off-Highway Motor Vehicle Recreation Division of California's Department of Parks and Recreation manages six state vehicular recreation areas to provide off-road vehicle opportunities.[96] Attendance at these areas increased by 52 percent between 1985 and 2000 — with a corresponding increase in greenhouse gas emissions.[97] Still, the Off-Highway Motor Vehicle Recreation Division is calling for an increase in new off-road vehicle facilities in the coming years.[98]

Other state lands also allow off-road vehicle use. Anza Borrego Desert State Park, for example, contains more than 500 miles of roads for four-wheel-drive and all-terrain vehicles and dirt bikes. Overall, the state of California provides thousands of miles of routes for off-road vehicle use throughout its state parks and more than 105,000 acres in state vehicular recreation areas. The use in state vehicular recreation areas shows an overall increase from 1992 to 2001 with estimated visitors in 2001 reaching more than 2.3 million (see Appendix B).[99]

Despite this, there has been little effort to consider the impacts of this growth on global climate change and emissions by the Department of Parks and Recreation. The Department's two-page "Response to Climate Change" devotes only a single paragraph to the Off-Highway Motor Vehicle Recreation Division, stating simply that the agency will take actions consistent with the Department's direction and will retrofit its trucks to comply with new California Air Resources Board standards.[100] There is virtually no mention of the significant climate and health effects of continued and expanded off-road vehicle use and no evidence of effort to avoid or mitigate greenhouse gas emissions associated with state-supported off-road vehicle use.

Off-road Vehicle Use on Federal Lands

California's federal lands offer millions of very accessible acres and thousands of miles of trails for off-road vehicle use. According to the Government Accounting Office, California boasts twice as many areas offering off-road vehicle recreation opportunities than in any other state[101] — and the state itself is the primary supporter of off-road vehicle recreation on these lands. (See Appendix C for a breakdown of federal lands open to off-road vehicle use.)

The Angeles National Forest in southern California, for example, is within an hour's drive of Los Angeles and currently provides 364 miles of designated off-road vehicle routes and more than 10,000 acres for open off-road vehicle use. Off-road vehicles contribute to poor air quality in Los Angeles, a non-attainment area, by releasing carbon monoxide and other contaminants into the air.[102] Still, the State of California spent $5.6 million between 1983 and 1998 to support off-road vehicle recreation on the Forest, including $401,720 to construct 36 miles of off-road vehicle trails in 1983 and $361,000 to develop another 58 miles of off-road vehicle routes in 1988. [103]

All told, the state provided the U.S. Forest Service with more than $58 million to support off-road vehicle recreation between 1983 and 1999.[104] Funding has continued for the past 25 years and in fact has expanded in recent years. For example, in 2006 and 2007, California sent the federal government more than $25 million to support off-road vehicle recreation and management on federal lands in the state.[105] Despite California's goals of reducing greenhouse gas emissions and protecting public health, its support for off-road vehicle recreation on federal lands continues.

Currently, all California national forests are undergoing travel-management planning to designate roads, trails, and areas where off-road vehicles are specifically allowed. [106] The Bureau of Land Management occasionally revises management guidelines for its lands open to off-road vehicles. But to date, the State of California has taken no substantive position regarding the climate change and public health implications of state-supported off-road vehicle recreation on public lands.

Tracks near Anza-Borrego Desert State Park. The California Department of Parks and Recreation has done little to address the climate change implications of off-road vehicles' increasing greenhouse gas emissions.

Photo by Larry Hogue

Recommendations

In order to prevent needless off-road vehicle pollution from further imperiling the global climate and public health, the California Air Resources Board must limit overall greenhouse gas emissions from off road vehicles to the maximum extent possible. Consistent with Assembly Bill 32 and the governor's executive order, a reduction to 1990 levels by 2020 should be considered only as the minimum reduction alternative. Such a limitation will ensure that recreational off-road vehicles are reducing emissions at the same pace as are other sectors of the population and will have important health benefits for Californians.

Off-road motorcycle in dune recreation area. No new state off-road vehicle sites should be established unless they fit an overall scheme to reduce off-road vehicle emissions to 1990 levels.
Photo by George Wuerthner

There are two ways to effectively limit greenhouse gas emissions from off-road vehicles: capping the number of vehicles registered and limiting their use. The surest way to limit overall off-road recreation emissions is to reduce the amount of off-road recreation allowed on both state and federal public lands throughout California. Specifically, the following should be achieved:

• **Reduction of greenhouse gas emissions from off-road vehicle use in state vehicular recreation areas and other state lands to at least 1990 levels**

The California Air Resources Board must analyze the amount of greenhouse gases being emitted from off-road vehicle use within state vehicular recreation areas and other state lands, while the Department of Parks and Recreation ensures that, at a minimum, off-road vehicle emissions from these areas are reduced to 1990 levels. Further emission reductions may be required if federal agencies do not reduce emissions from off-road vehicles on federal lands.

No new state off-road vehicle sites should be established unless they are consistent with an overall scheme to reduce total off-road vehicle emissions to 1990 levels.

• **Enforced federal management of California off-road recreation limiting off-road vehicle emissions to, at a minimum, 1990 levels**

Hillside scarred by off-road vehicle use. The California Department of Parks and Recreation now provides tens of millions of dollars to federal agencies to promote and manage polluting and damaging off-road vehicles. Photo by Chris Kassar

Because significant greenhouse gas emissions arise from off-road vehicle use on federal lands, the State of California must ensure that those emissions are reduced along with emissions from other sources. The state should ensure that federal agencies managing off road vehicles in California are limiting greenhouse gas emissions from discretionary recreational off-road vehicle use; a reduction in this source to 1990 levels by 2020 should be considered as the minimum reduction alternative. The state should also deny financial support and permits to federal agencies that do not meet this target.

First, this requires that the California Air Resources Board adopt rules that call for the rejection of applications for new, continued, or expanded off-road vehicle use on federal lands from federal agencies or districts that do not have an adequate plan to reduce overall off-road vehicle emissions from their jurisdiction to, at a minimum, 1990 levels.

Second, this means that the Off-Highway Motor Vehicle Recreation Division should adopt rules requiring the rejection of applications for funding from federal agencies or districts that do not have a sufficient plan to reduce overall off-road vehicle emissions from their jurisdiction to the maximum extent possible — at a minimum, to 1990 levels by 2020. Currently, the California Department of Parks and Recreation provides tens of millions of dollars to federal agencies to promote and manage off-road vehicles.

Finally, the State of California should provide substantive comments on federal land-use plans and proposals that will result in increased greenhouse gas emissions. The California Air Resources Board and other state agencies should take substantive positions on proposed federal land management plans — including pending travel-management plans — and projects that urge federal land management agencies to ensure that each plan or project is consistent with an overall plan to reduce off-road vehicle emissions to at least 1990 levels. The state should encourage a cap on off-road vehicle use on federal lands that is scaled to achieve maximum emission reductions.. To date, California has not offered consistent substantive comments on federal land-use proposals that will impact global climate change.

• A cap on the number of registrations issued for off-road vehicles in California.

The Department of Motor Vehicles should cap off-road vehicle registrations to achieve an emission reduction to, at a minimum, 1990 levels, which should be adjusted depending on the effectiveness of limitations on use described above. Because registration enforcement is lax, additional resources will be required for effective enforcement. Additionally, the California Air Resources Board should immediately address the adverse public health effects and climate implications of non-conforming off-road vehicles.

• Elimination of loopholes that allow continued use of polluting off-road vehicles that fail to meet state emission standards.

Just as California does not allow the continued use of automobiles that do not meet state emission standards, the state should not allow use of off-road vehicles that fail to comply with state standards. The California Air Resources Board should eliminate the “red-sticker” loophole that allows continued use of polluting off-road vehicles that do not meet state emission standards.

• Rejection of continued or expanded off-road vehicle use on federal lands in areas that do not meet air quality standards.

California must certify that proposed land uses on federal lands conform with the state’s enforcement of the Clean Air Act. To date, the state regularly approves these uses — even in non-conforming areas like Imperial County — without significant evaluation. The California Air Resources Board should reject proposals to continue or expand off-road vehicle recreation on federal lands in areas that do not meet air quality standards.

Off-road vehicle tracks with run-over sign. Enforcement of rules is lax on public lands. The Department of Motor Vehicles will need more resources to institute and enforce a cap on off-road vehicle registrations.
Photo by Larry Hogue

Conclusion

The State of California has developed laudable goals to reduce greenhouse gas emissions and protect the health of California residents. Exhaust from off-road vehicles contains the same greenhouse gases as emissions from cars — and significantly more pollution. In addition, just as the number of cars on the road has increased, off-road vehicle use has skyrocketed in the last 20 years. The continued rise of off-road vehicle recreation — and the pollution and greenhouse gas emissions associated with it — threaten to undermine the state's goals for reversing climate change and improving public health.

The California Air Resources Board must place recreational off-road vehicle pollution on the table with emissions from automobiles, smokestacks, and other polluters. The state must take immediate action to prevent off-road vehicle pollution from continuing to jeopardize the public health of California residents, contributing to disastrous changes in climate, and otherwise harming California's natural and cultural heritage.

Dust plume from off-road vehicle staging near public lands
Photo courtesy Community ORV Watch

Appendix A: Off-road Vehicle Riding Areas Open to Non-compliant Vehicles

California Air Resources Board (CARB) Non-compliant OHV (Red Sticker) Riding Season Schedule		Map Area ID	Red Sticker Riding Season	
			Riding Starts	Riding Ends
State Vehicular Recreation Areas (SVRA)				
SVRA	Clay Pit	38	1-Sep	30-Jun
State Recreation Area (SRA)	Mammoth Bar	40	Year round	
SVRA	Prairie City	53	1-Oct	30-Apr
SVRA	Carnegie	65	1-Oct	30-Apr
SVRA	Hollister Hills	75	1-Oct	31-May
SVRA	Oceano Dunes	87	Year round	
SVRA	Hungry Valley	102	1-Oct	30-Apr
SVRA	Ocotillo Wells	124	1-Oct	31-May
SVRA	Heber Dunes	128	Year Round	
Bureau of Land Management (BLM)				
Northern California				
BLM Arcata Field Office	Samoa Dunes	6	Year round	
BLM Redding Field Office	Chappie-Shasta ORV Area	8	1-Oct	30-June
BLM Eagle Lake Field Office	Fort Sage OHV Area	16	Year round	
BLM Ukiah Field Office	South Cow Mountain Recreation Area	36	Year round	
BLM Ukiah Field Office	Knoxville Recreation Area	37	Year round	
Bakersfield District				
BLM Hollister Field Office	Clear Creek Management. Area	76	1-Oct	31-May
BLM Bishop Field Office	Bishop Resource Area	82	Year round	
California Desert District				
BLM Ridgecrest Field Office	Olancha Dunes	96	Year round	
BLM Ridgecrest Field Office	Jawbone Canyon, Dove Springs	103	1- Sep	31-May
BLM Ridgecrest Field Office	Spangler Hills	104	1 Sep	31-May
BLM Barstow Field Office	Dumont Dunes	105	Year round	
BLM Barstow Field Office	El Mirage	109	1-Oct	30-Apr
BLM Barstow Field Office	Stoddard Valley	110	1-Sep	31-May
BLM Barstow Field Office	Rasor	111	1-Sep	31-May
BLM Barstow Field Office	Johnson Valley	115	1-Sep	31-May
BLM Needles Field Office	Eastern Mojave Desert Areas	118	Year round	
BLM Lake Havasu Field Office	Parker Strip	120	Year round	
BLM Palm Springs Field Office	Colorado Desert Areas	122	1-Oct	30-Apr
BLM El Centro Field Office	Lark Canyon	127	1-Oct	30-Apr
BLM El Centro Field Office	Arroyo Salado	125	1-Oct	31-May
BLM El Centro Field Office	Superstition Mountain	129	1-Oct	31-May
BLM El Centro Field Office	Plaster City	130	1-Oct	31-May
BLM El Centro Field Office	Imperial Dunes-Mammoth Wash	131	Year round	
BLM El Centro Field Office	Imperial Dunes-Glamis/Gecko	132	Year round	
BLM El Centro Field Office	Imperial Dunes-Buttercup Valley	133	Year round	

United States Forest Service (USFS)				
Shasta-Trinity National Forest				
Mc Cloud Ranger District	McCloud Area	5	Year round	
Hayfork Ranger District	Hayfork Area	7	Year round	
Plumas National Forest				
Mt. Hough Ranger District	Deadman Springs, Snake Lake	18	Year round	
Mt. Hough Ranger District	Big Creek, Four Trees, French Creek	20	Year round	
Feather River Ranger District	Cleghorn Bar, Poker Flat	22	Year round	
Beckworth Ranger District	Gold Lake	25	Year round	
Beckworth Ranger District	Dixie Mountain	27	Year round	
Mendocino National Forest				
Upper Lake Ranger District	Lake Pillsbury	33	Year round	
Upper Lake Ranger District	Elk Mountain Area	34	Year round	
Grindstone Ranger District	Davis Flat	35	Year round	
Tahoe National Forest				
Downieville Ranger District	Downieville Area	23	Year round	
Foresthill Ranger District	Foresthill OHV Area	49	Year round	
Foresthill Ranger District	China Wall	50	Year round	
Nevada City Ranger District	Nevada City District Areas	41	Year round	
Nevada City Ranger District	Fordyce	42	Year round	
Sierraville Ranger District	Sierraville Area	30	Year round	
Truckee Ranger District	Truckee District Area	43	Year round	
Truckee Ranger District	Prosser Hills Area	44	Year round	
Lake Tahoe Basin Management Unit	Kings Beach	47	Year round	
Eldorado National Forest				
Georgetown Ranger District	Mace Mill, Rock Creek	51	Year round	
Pacific Ranger District	Barrett Lake	52	Year round	
Stanislaus National Forest				
Calaveras Ranger District	Corral Hollow, Spicer	58	Year round	
Summit Ranger District	Niagara Ridge Area	60	Year round	
Mi-Wuk Ranger District	Crandall Peek, Deer Creek Area	62	1-Oct	31-May
Mi-Wuk Ranger District	Hunter Creek	63	1-Oct	31-May
Mi-Wuk Ranger District	Hull/Trout Creek	64	1-Oct	31-May
Groveland Ranger District	Date Flat, Moore Creek Area	69	1-Oct	31-May
Sierra National Forest				
Mariposa/Minarets Ranger District	Hites Cove	70	1-Oct	31-May
Mariposa/Minarets Ranger District	Miami Motorcycle Trails	71	1-Oct	31-May
Kings River-Pineridge Ranger District	Huntington Lake	77	1-Oct	31-May
Kings River-Pineridge Ranger District	Eastwood	78	1-Oct	31-May
Kings River-Pineridge Ranger District	Shaver Lake Area	79	1-Oct	31-May
Kings River-Pineridge Ranger District	Kings River, Pineridge	81	1-Oct	31-May
Hume Lake Ranger District	Quail Flat	83	1-Oct	31-May

Sequoia National Forest				
Greenhorn Ranger District	Frog Meadow Area	90	1-Oct	31-May
Tule River Ranger District	Tule River Area	93	1-Oct	31-May
Cannell Ranger District	Kennedy Meadows	95	Year round	
Inyo National Forest				
White Mountain Ranger District	Poleta	97	Year round	
Los Padres National Forest				
Santa Lucia Ranger District	Black Mountain	88	Year round	
Mt. Pinos Ranger District	Ballinger Canyon	98	1-Oct	30-Apr
Mt. Pinos Ranger District	Alamo Mountain	99	1-Oct	30-Apr
Santa Barbara Ranger District	Santa Barbara	100	1-Oct	30-Apr
Ojai Ranger District	Ortega Trail	101	1-Oct	30-Apr
Angeles National Forest				
Santa Clara/Mojave Rivers Ranger District	Drinkwater Flats	106	1-Oct	30-Apr
Santa Clara /Mojave Rivers Ranger District	Rowher Flat	107	1-Oct	30-Apr
Santa Clara/Mojave Rivers Ranger District	Littlerock	108	1-Oct	30-Apr
San Gabriel River Ranger District	San Gabriel	112	1-Oct	30-Apr
San Bernardino National Forest				
Front Country Ranger District	Lytle Creek Area	113	1-Oct	30-Apr
Mountain Top Ranger District	Lake Arrowhead Area	116	1-Oct	30-Apr
Mountain Top Ranger District	Big Bear Lake Area	117	1-Oct	30-Apr
San Jacinto Ranger District	San Jacinto Area	121	1-Oct	31-May
Cleveland National Forest				
Trabuco Ranger District	Wildomar	123	1-Oct	30-Apr
Descanso Ranger District	Corral Canyon	126	1-Oct	30-Apr
Other Jurisdictions				
Army Corps of Engineers	Black Butte Lake	32	Year round	
City of Marysville (Riverfront)	Eugene Chappie OHV Park	39	Year round	
Santa Clara County	Metcalf Motorcycle Park	66	1-Oct	30-Apr
Stanislaus County	Frank Raines-OHV Park	67	1-Oct	30-Apr
Stanislaus County	La Grange	68	1-Oct	30-Apr
San Bernardino County	Park Moabi	119	Year round	

This list was provided by the California Air Resources Board (CARB). It will be updated periodically and you may contact CARB at (800) 242-4450 for more information.

Map available from California State Parks OHMVR Division that corresponds to Map Area ID.

Appendix B: State Vehicular Recreation Area Visitation, 1992-2006

This chart shows the number of visitors to state vehicular recreation areas. Data results from a combination of estimates based on field observations and paid entrance fees and conversion factors. Data includes both paid and free entries.

SVRA	1992	1993	1994	1995	1996	1997	1998
Carnegie	46,986	45,547	48,740	38,446	35,302	41,976	69,918
Heber Dunes	---	---	---	---	---	---	---
Hollister Hills	92,098	93,180	86,460	81,235	89,464	99,757	109,694
Hungry Valley	113,157	112,827	93,477	152,075	143,889	96,492	107,988
Oceano Dunes	1,173,019	1,090,522	925,131	1,106,221	1,090,223	1,075,621	1,013,728
Ocotillo Wells	288,800	301,092	298,418	306,874	323,414	302,607	236,722
Prairie City	43,730	36,278	42,349	44,800	56,802	56,926	55,652
Total visitation	1,757,790	1,679,446	1,494,575	1,729,651	1,739,094	1,673,379	1,593,702

SVRA	1999	2000	2001	2002	2003	2004	2005	2006
Carnegie	102,488	118,687	124,332	137,547	135,941	127,135	120,215	128,056
Heber Dunes	---	26,505	26,704	32,459	30,249	45,056	48,605	49,123
Hollister Hills	125,800	153,003	143,473	158,785	186,771	177,714	165,104	187,004
Hungry Valley	128,419	352,760	382,225	450,737	536,591	544,322	357,634	237,347
Oceano Dunes	1,093,647	1,243,445	1,204,541	1,364,397	1,428,472	1,809,469	2,055,631	1,991,445
Ocotillo Wells	281,751	365,933	325,056	495,786	609,762	816,450	938,554	1,324,389
Prairie City	77,413	93,720	121,271	140,344	149,446	193,330	188,368	168,941
Total visitation	1,809,518	2,354,053	2,327,602	2,780,055	3,077,232	3,713,476	3,874,111	4,086,305

Source: California State Parks, Off-highway Motor Vehicle Division

Appendix C: Public Lands in California Open to Off-road Vehicles[106]

State Lands

State Vehicle Recreation Areas:

Carnegie: 1500 acres
Hollister Hills: 3200 acres
Hungry Valley: 19,000 acres
Oceano Dunes: Approximately 3,800 acres
Ocotillo Wells: More than 80,000 acres
Prairie City: 836 acres

Federal Lands

National Forests:

Angeles: 364 miles of designated off-highway vehicle routes and more than 10,000 acres of open areas
Cleveland: More than 600 miles of roads and trails; more than 400,000 acres of open areas
Eldorado: 2200 miles of roads
Humboldt-Toiyabe: More than1500 miles of roads and trails; more than 800,000 acres open to cross-country travel (California portion of the forest)
Inyo: More than 3,000 miles of roads and trails; more than1 million acres of open areas
Klamath: More than 5,000 miles of roads and trails; more than 1 million acres of open areas
Lake Tahoe: More than 4,000 miles of roads and trails; more than 900,000 acres of open areas
Lassen: More than 4,000 miles of roads and trails; more than 1 million acres of open areas
Los Padres: More than 1,500 miles of roads and trails
Mendocino: More than 800 miles of roads and trails
Modoc: More than 1 million acres of open areas
Plumas: More than 1 million acres of open areas
San Bernardino: 42 miles of 24- to 50-foot trails; 166 miles for green-sticker/red-sticker use; 903 miles of road open to sport utility vehicles and four-wheel-drive vehicles
Sequoia: 1,267 miles of roads and trails, including trails open to off-road vehicle use within the Giant Sequoia National Monument; 10,000 acres of open areas
Shasta-Trinity: More than 6,000 miles of roads and trails; more than 200,000 acres of open areas
Sierra: More than 2,000 miles of roads and trails; more than 500,000 acres of open areas
Six Rivers: more than3,000 miles of roads and trails
Stanislaus: more than 3,000 miles of roads and trails; more than 500,000 acres of open areas
Tahoe: More than 4,000 miles of trails and roads; more than 900,000 acres of open areas

Bureau of Land Management Lands

Within the Bureau of Land Management's field offices, there are 11 million acres of agency land in California available for open and limited off-road vehicle recreation. The following is not a comprehensive list of all areas managed by the Bureau of Land Management in which off-road vehicles are allowed, but it lists some of the more well-known open off-road vehicle areas managed by the agency in California.[107]

Chappie-Shasta: 200 miles of trail, Shasta County
Cow Mountain: 52,000 acres, Lake and Mendocino Counties
Clear Creek Management Area: 76,000 acres, San Benito and Fresno Counties
Dove Springs: 5,000 acres, Kern County
Dumont Dunes: 8,150 acres, San Bernardino County
El Mirage Dry Lake Off-highway Vehicle Area; 24,000 acres, San Bernardino County
Fort Sage: 22,000 acres, Lassen County
Jawbone Canyon: 7,000 acres, Kern County
Johnson Valley: 140,000 acres, San Bernardino County
Imperial Sand Dunes: 150,000 open acres; Imperial County
Knoxville: 17,700 acres, Lake and Napa Counties
Lark Canyon: 1200 acres; 31 miles of trails, San Diego County
Plaster City: 41,000 acres, Imperial County
Rasor: 22,,500 acres, San Bernardino County
Samoa Dunes; 300 acres, Humboldt County
Spangler Hills; 57,000 acres, Kern County
Stoddard Valley; 50,000 acres, San Bernardino County

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Pristine area in the Trinity Alps, California
Photo by Chris Kassar

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Spring-fed wetland and riparian plant communities respond differently to altered grazing intensity

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Summary

1. Spring-fed wetlands are nested within California's oak savanna–annual grassland, which is considered a non-equilibrium-type system because it shows little community-level response to changes in grazing intensity. This insensitivity to disturbance is thought to result from transient resource limitation and the annual life cycle tending to swamp the effects of defoliation. Because spring-fed wetlands receive relatively consistent and high inputs of water and nutrients, and they are dominated by perennial herbaceous vegetation, we hypothesized that these systems would respond to a grazing intensity gradient in a predictable manner, i.e. equilibrium dynamics would prevail.

2. We experimentally tested plant community responses of spring-fed wetlands to two levels of grazing intensity (light and moderate) and no grazing over 10 years. Wetland vegetation was tracked at two distinct geomorphological parts of the wetland system: spring heads, where emergent water formed marshy zones, and their resultant channelized creeks.

3. We used linear mixed effects models to estimate grazing intensity effects over time. A general result at both springs and creeks was that slope estimates of annual total herbaceous cover over time were negative under moderate grazing but positive under light grazing and grazing removal.

4. Diversity metrics were not affected by grazing treatments at springs. At creeks, Simpson's diversity index increased over time under moderate grazing. When averaged over 10 years, species evenness and Shannon's diversity index were greater under moderate grazing at creeks.

5. Species composition was highly variable from year to year at springs, with no separation of the first ordination axis (detrended correspondence analysis) amongst grazing treatments. In contrast, three relatively stable and distinct equilibria were evident for creeks. These results indicate that springs exhibited non-equilibrium dynamics while their creek counterparts, separated by several metres, behaved in a more equilibrium fashion, in relative terms.

6. Grazing removal significantly increased interannual variability in species composition at both geomorphological types, demonstrating that plant–animal interactions serve as the main control on community composition in these systems by reducing cover and promoting diversity and stability.

7. *Synthesis and applications.* Grazing managers in California's oak savanna–annual grassland have little control of species composition because the annual community is entrained by the weather. Our results show that first-order wetland–riparian communities nested within this annual grassland matrix can be manipulated via grazing intensity adjustments. However, marshy spring-fed wetland areas are less sensitive than channelized creeks to these manipulations and grazing removal will increase variability in species composition for both geomorphological types. Grazing management decisions are usually made at the landscape level based on the matrix vegetation, but nested ecosystems may respond differentially, requiring a more nuanced approach that includes site-specific information.

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Introduction

A major shift in plant community ecology occurred in the last quarter of the 20th century, away from an equilibrium model of community dynamics towards non-equilibrium explanations (DeAngelis & Waterhouse 1987; Ellis & Swift 1988; Oba, Stenseth & Lusigi 2000; Sullivan & Rohde 2002). Non-equilibrium is frequently invoked in rangeland ecology when community changes do not match predictions based on Clementsian successional theory (Westoby, Walker & Noy-Meir 1989; George, Brown & Clawson 1992; Allen-Diaz & Bartolome 1998; Fernandez-Gimenez & Allen-Diaz 1999; Jackson & Bartolome 2002), which posits that communities will respond in a sequential and predictable manner to the environment and disturbances. More importantly, an assumption of these models is that the environment and disturbance additively perturb a system from its equilibrium state, which is the product of competitive plant–plant and plant–animal interactions. The mechanism implicit in these models is the reduction or loss of inferior competitors via Gause's competitive exclusion principle (Grime 1973).

Non-equilibrium explanations of vegetation dynamics tend to be concentrated on arid and semi-arid ecosystems (rangelands) because they possess notoriously erratic weather at time scales that are relevant to their management (Bestelmeyer *et al*. 2003; but see Knapp *et al*. 2002; Seastedt & Knapp 1993). Such models allow for interactive effects between the environment and disturbances, resulting in the possibility of multiple vegetation states that are more or less stable in time (Stringham, Krueger & Thomas 2001). Moreover, non-equilibrium models allow for a decoupling of plant–animal interactions, offering a mechanistic explanation for the often observed lack of grazing effects on vegetation composition (Westoby, Walker & Noy-Meir 1989). Some have argued that non-equilibrium explanations are incorrectly applied to rangeland management and policy, spreading the notion that grazing has no effect on rangeland vegetation (Illius & O'Connor 1999; Cowling 2000; but see Sullivan & Rohde 2002).

Briske, Fuhlendorf & Smeins (2003) sought to integrate the rather polarized equilibrium vs. non-equilibrium debate by relying on Wiens's (1984) description of this framework as a continuum along which all communities can be placed. Subsequent empirical work has shown that not only can ecosystems exhibit more or less equilibrium-type behaviour, but also that within communities and across spatiotemporal scales some areas or components can be better explained with equilibrium and others with non-equilibrium dynamics (Fernandez-Gimenez & Allen-Diaz 1999; Oba *et al*. 2003). For instance, Buttolph & Coppock (2004) showed that riparian areas followed the equilibrium paradigm within an Andean pastoral system that could otherwise be characterized as non-equilibrium. On the other hand, Walker & Wilson (2002) subjected a long-term New Zealand vegetation data set to both formal (mathematical) and informal equilibrium criteria, but were unable to demonstrate formal equilibrium. Indeed, Roxburgh & Wilson (2000) showed that equilibrium was difficult to demonstrate even in a well-manicured university lawn.

Non-equilibrium mechanisms

Conceptual models notwithstanding, relatively high species richness is maintained in many plant communities, especially grasslands. Much of the plant ecology community seeks mechanisms that allow for the co-existence of multiple species (Levine & Rees 2002; Coomes & Grubb 2003; Palmer, Stanton & Young 2003). The fundamental question arising from these efforts is: what phenomena subvert competitive exclusion in plant communities? Rees *et al*. (2001) summarized current understanding of plant diversity maintenance by stating that co-existence arises from two phenomena: a competition–colonization trade-off and niche differentiation. Under the former, succession towards an equilibrium state is the dominant process as species with small, high-density seed and high growth rates colonize gaps (where resources are abundant) but are ultimately outcompeted by species possessing bigger seed or other life-history characteristics that provide greater resource reserves as environmental resources are diminished. The non-equilibrium aspect of this explanation is the creation of gaps via disturbances that derail the march towards an equilibrium community.

Niche differentiation facilitates species' co-existence by spatial and/or temporal separation of resource supply and demand. An example of spatial separation has been demonstrated in California grasslands where perennial grasses, once established, access deeper below-ground resources than annual grasses (Brown & Rice 2000). Here it seems that the competition–colonization trade-off is in effect initially, where the annual grasses proliferate and dominate, but if the better competitor (the perennial grass) can gain a foothold eventually the competition between these grasses with different life histories is reduced by spatial, and to some degree temporal, separation (Rice 1985; Dyer & Rice 1997).

SPRING-FED WETLANDS NESTED WITHIN CALIFORNIA ANNUAL GRASSLANDS

Non-equilibrium is invoked with niche differentiation especially where niche space is partitioned in the temporal realm (Blair 1997; Seastedt & Knapp 1993). A good example of this is manifested in Californian annual grasslands where species composition is notoriously variable from year to year (Pitt & Heady 1978; Bartolome 1989; Evans & Young 1989). Jackson & Bartolome (2002) showed that most of this variability is linked to the vagaries of the Mediterranean climate. Indeed, non-equilibrium vegetation dynamics are most often cited in areas with a highly variable climate (Huntsinger & Bartolome 1992; Sharp & Whittaker 2003; Stringham, Krueger & Shaver 2003). Nested within this non-equilibrium matrix of annual grasslands are spring-fed wetlands (Allen-Diaz & Jackson 2000; Allen-Diaz, Jackson & Phillips 2001), which are dominated by perennial herbaceous species. The nested nature of spring-fed wetlands makes them interesting systems for studying plant community dynamics because resources are abundant and their availability is synchronized to the growing season, compared with the surrounding annual grasslands where water, energy and sometimes mineral nitrogen availability are limiting and out of phase (Jackson *et al*. 1988, 1990; Chiariello 1989; Dahlgren, Singer & Huang 1997; Reynolds *et al*. 1997; Herman, Halverson & Firestone 2003). It is likely that plants of spring-fed wetlands undergo little abiotically driven resource limitation during their growing season as their forward-shifted phenology (spring–summer growth) allows consumption of ample supplies of exogenous water, light and nutrients during summer months (Jackson *et al*. 2006). This convergence of resource availability should favour equilibrium dynamics (Seastedt & Knapp 1993), where responses to disturbance regime alteration are sensitive and predictable. This follows from equilibrium theory, which holds that competitive exclusion is the primary structuring force of communities and that plant–plant and plant–animal interactions should be the most important modulators of competitive balance (Wiens 1984).

We tested hypotheses borne from equilibrium-type community dynamics by measuring vegetation responses to grazing intensity manipulations for over a decade in spring-fed wetlands and their associated creeks (Fig. 1). Our three equilibrium-based hypotheses were as follows.

1. Plant cover and grazing intensity will negatively co-vary because increasing defoliation intensity exposes an increasing amount of bare soil. An alternative and more general hypothesis that would also conform to equilibrium dynamics is that plant cover would be some function of grazing intensity, linear or otherwise. For instance, plant cover might respond to grazing intensity in a unimodal way, constituting evidence for the grazing optimization hypothesis (McNaughton 1979). The important part of our prediction is that plant cover would be related to grazing intensity in any way. If it was not, we would either determine that our grazing treatments did not have the desired effect or that factors other than plant–herbivore interactions were driving plant cover.

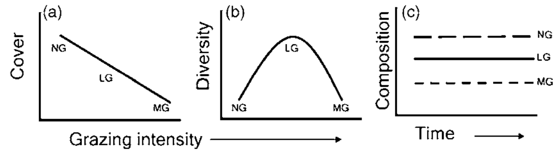

Fig. 1. Equilibrium-based hypotheses that (a) herbaceous cover would decline with increased grazing intensity, (b) diversity would have a unimodal response (i.e. the intermediate disturbance hypothesis) to increased grazing intensity, and (c) composition would be different across grazing intensity treatments over time. Under non-equilibrium dynamics, these parameters would not respond in a predictable manner to changes in grazing intensity because the vegetation state is not a function of plant–animal interactions but rather shaped by stochastic processes such as weather.

2. If a herbaceous plant community is in equilibrium with its environment and the organisms living within it, then plant diversity will show a unimodal response to a grazing intensity gradient, i.e. the intermediate disturbance hypothesis (Connell 1978). Low levels of defoliation result in a few fast-growing tall species, shading out others and resulting in low diversity. On the other hand, grazing-intolerant species would be outcompeted by grazing-tolerant species under high grazing intensities, so we would predict low diversity at either end of the grazing intensity gradient. In a non-equilibrium situation, environmental variability would swamp such diversity shifts, resulting in diversity measurements that are not related to grazing intensity.

3. Plant species composition will track grazing intensity changes because the realization of the plant community will be species whose competitive abilities are maximized under a particular level of defoliation. Hence this variable can be thought of as an integrator of plant cover and diversity responses.

We propose that a failure to reject any or all of these hypotheses would constitute more or less support for equilibrium dynamics prevailing in these systems, with the understanding that ecosystems need not be classified exclusively as either equilibrium or non-equilibrium (Fernandez-Gimenez & Allen-Diaz 1999; Jackson & Bartolome 2002; Briske, Fuhlendorf & Smeins 2003) but acknowledging the utility of such a classification for management (Allen-Diaz & Bartolome 1998; Wiens 1984; Westoby, Walker & Noy-Meir 1989; Behnke & Scoones 1993; Illius & O'Connor 1999)

Materials and methods

STUDY SYSTEM

A significant problem for managing riparian systems is the variability among systems themselves (Rosgen 1985). The diversity among riparian system characteristics has made them difficult to classify (Brown, Lowe & Pase

1984; Nelson & Nelson 1984). For example, first-order riverine systems may maintain vegetation similar to a fourth-order stream, yet their response to grazing may differ based on substrate or slope differences. Spring-fed wetlands are first-order riparian systems where subsurface water emerges in an often diffuse way. However, these waters usually come together to form more channelized creek systems within 1–20 m of the emergent source and maintain distinctive vegetation (forbs and grasses) compared with the up-slope wetlands (sedges and rushes).

In the blue oak woodlands of the Sierra Nevada foothills, spring-fed wetlands form small patches and corridors within a matrix of annual grassland that may or may not contain an oak *Quercus douglasii* Nee and/or foothill pine *Pinus sabiniana* Douglas overstorey. These first-order riparian areas are perennially moist and seasonally flooded; they usually occur at slope-breaks where underlying bedrock or relatively impervious clay layers intercept the soil surface. The resulting emergent water creates favourable conditions for highly productive cattails *Typha angustifolia* L., sedges (*Cyperus* spp., *Carex* spp., *Eleocharis* spp.) and warm-season grasses *Paspalum dilatatum* Poiret and *Echinochloa crus-galli* L. (P. Beauv.). Annual net primary production has been estimated at 400–1000 g m^{-2} dry biomass (Jackson 2002), making these some of the most productive herbaceous ecosystems in California. Cattails occur in saturated soils where water actually pools on relatively flat slopes. Sedges dominate where undulating microtopography creates a mosaic of vegetated tussocks and standing pools of water, which serve to baffle the flow of water on intermediate slopes. Finally, grass-dominated wetlands are characterized by perennial grass species that occur where soils are wet, but not necessarily saturated, most of the year.

STUDY SITE

We conducted this research at the Sierra Foothill Research and Extension Centre (SFREC), located on the western slope of the Sierra Nevada foothills in Yuba County, California, USA (Fig. 2). Owned and managed by the University of California (UC) for more than 40 years, it covers 2300 ha of steep to rolling landscape, 90–600 m a.s.l. The foothill oak savannas of this region have supported livestock grazing for more than 120 years. Grazing was especially heavy during the Gold Rush period of the late 1800s. When SFREC came under UC ownership and management in the 1960s, a more moderate grazing regime was implemented that persists today.

Weather data were downloaded from SFREC's web site (http://danrrec.ucdavis.edu/sierra_foothill/resources_climate_precipitation.html (accessed 22 Feb 2006).). The 30-year average total annual precipitation at SFREC is 72 cm, which falls as rain from autumn through spring. During the study period total rainfall was higher than normal (81 cm rain year^{-1}, CV = 31%, rain year October–September). Maximum and minimum air temperatures in the region range from 32·0 °C (July) to 2·2 °C

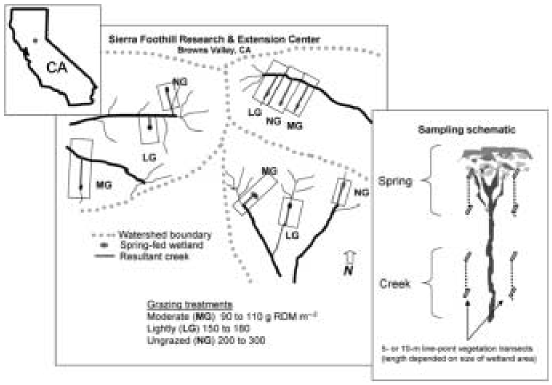

Fig. 2. Study site location, experimental layout and sampling schematic. Grazing treatments were applied to 2–5-ha pastures enclosing spring–creek wetland systems. Intensity of these treatments was applied based on the amount of residual dry matter (RDM) in annual grass uplands surrounding the wetland areas because this is the relevant management scale.

(January). Soils in this area are generally shallow clay loams classified as auburn (loamy, oxidic, thermic, ruptic lithic xerochrepts) and argonaut (fine, mixed, thermic mollic haploxeralfs) series (Herbert & Begg 1969).

GRAZING TREATMENTS AND VEGETATION DATA COLLECTION

Within each of three watersheds, three spring-fed wetlands and their resultant first-order creeks were randomly assigned a grazing treatment, not grazed (NG), lightly grazed (LG) or moderately grazed (MG), applied annually from 1993 through 2002 (Fig. 2). Cattle were placed within 2–5-ha treatment areas surrounding wetland zones that were enclosed by a three-wire electrical fence for variable periods in November, January–March and May, to approximate customary autumn–winter–spring grazing practices and to achieve the desired residual dry matter (RDM) levels in the annual grass-dominated uplands. Because grazing during the growing season has little effect on the present year's forage composition and productivity in California's annual grasslands (Bartolome & McClaran 1992; Jackson & Bartolome 2002), managers use grazing intensities that leave the desired amount of RDM, thereby protecting soil and optimizing future productivity, which is generally related to RDM in a positive and linear way (Bartolome, Stroud & Heady 1980; Heady *et al*. 1992). Experienced SFREC range managers monitored grazing treatment intensity during each treatment period. Cattle were left on a site until visually estimated RDM target levels were attained for each period, representing moderate grazing (MG, *c.* 150 g RDM m^{-2}) or light grazing (LG, *c.* 250 g RDM m^{-2}). To verify grazing treatment levels, we clipped aboveground herbaceous biomass each June (1993–98) from three randomly located 0·0625-m^2 quadrats per fenced treatment plot, dried clippings for 2 days at 65 °C, and weighed and recorded biomass. These data demonstrated that we were effectively applying a grazing intensity gradient to these plots (NG, 375 ± 46; LG, 233 ± 38; MG, 149 ± 16; mean g RDM m^{-2} ± SE, $n = 18$).

Depending on the size of the wetland area, two permanent 10- or 5-m vegetation sampling transects were randomly established on each side of a spring and each side of a creek running parallel to surface water flow at each site (Fig. 2, inset). From 1992 to 2002, transects were sampled using the line-point method (Heady, Gibbens & Powell 1959) to determine plant species cover 3 weeks after late spring cattle removal from grazing treatments (late May/early June). Grassland vegetation sampling methods were reviewed by Sorrells & Glenn (1991) and point-based methods were preferred because they were correlated with biomass measurements of species abundance (Arny & Schmid 1942). The first interception of a lowered, sharpened metal rod with any part of herbaceous vegetation was recorded for 100 systematically sampled points along each transect (i.e. 200 points per experimental unit), irrespective of transect length. The variable transect lengths were imposed because perennial herbs associated with riparian zones occurred in very small areas at some sites. While the transect length and therefore the interval differed, the number of points recorded did not. We do not believe this introduced any bias into the measurements because the interval in both cases (either 1 or 0·5 dm) was larger than individual plant stems and leaves. Bias towards taller vegetation may be introduced with this method, although observations at other California grassland sites showed no differences between data collected in this fashion and where multiple hits were recorded through the herbaceous layer for each point (J.W. Bartolome, unpublished data). The herbaceous layer of these systems are 10–150 cm tall depending on species composition but they do not exhibit distinctive layers of vegetation, so we do not believe our line-point technique introduced significant bias into the estimation of species cover. This method has been employed by range scientists and grassland ecologists for decades (Heady 1958, 1977; Cook & Stubbendieck 1986; Heady *et al*. 1992; Bartolome *et al*. 2004; Kluse & Allen-Diaz 2005).

DATA ANALYSES

Total herbaceous cover was calculated as total vegetation hits (by species) divided by total possible hits (100). In subsequent statistical analyses, this response variable was arcsin transformed as suggested by Crawley (2002). Plant species identification and nomenclature followed *The Jepson Manual* (Hickman 1993). The site × species matrix was subjected to detrended correspondence analysis (DCA; PC-Ord vs. 4·01; McCune & Mefford 1999) from which the first ordination axis (DCA1) was extracted for use as a synthetic species composition index variable. Species richness and the Shannon and Simpson diversity indices (Mueller-Dombois & Ellenberg 1974) were also calculated for each plot × year combination using PC-Ord software.

Total cover, diversity and composition variables were subjected to linear mixed effects (LME) modelling (S-plus V.6) using the restricted maximum likelihood (REML) algorithm in three distinct ways. First, we used an ANCOVA LME approach where the fixed effects of fully parameterized models were slope estimates of the response variable (y) vs. year for each combination of geomorphology (subscripted by j) and grazing treatment (subscripted by k). This approach allowed us to determine how a given vegetation parameter was changing over time (significant slope) and how such a change might be modified by grazing management. Random effects included estimating the variance of intercepts across years for each level of watershed (subscripted by l), each level of grazing nested within watershed, and each level of geomorphology nested within grazing nested within watershed:

$$y_{ijkl} = \beta_0 + \beta_1(year)_{jk} + (b_0)_l + (b_0)_{k,l} + (b_0)_{j,k,l} + \varepsilon_{ijkl}$$
$$i = (1, \dots, 90),\ j = (1, 2),\ k = (1, 2, 3),\ l = (1, 2, 3)$$
$$b_l \sim N(0, \sigma_1^2),\ b_{k,l} \sim N(0, \sigma_2^2),\ b_{j,k,l} \sim N(0, \sigma_3^2),$$
$$\varepsilon_{ijkl} \sim N(0, \sigma^2) \qquad \text{eqn 1}$$

Secondly, we used ANOVA LME models for responses variables whose 1993–2002 values had been normalized to the pre-treatment year (1992) by dividing the value for each experimental unit × year combination by its 1992 value. For these analyses the fixed effects were year (as a categorical factor subscripted by j), grazing treatment (subscripted by k) and interactions among these two factors. Random effects of fully parameterized ANOVA LME models were watershed (subscripted by l) and grazing nested within watershed. Geomorphology was excluded as a model term because there were insufficient degrees of freedom to test more general models; data for springs and creeks were analysed separately. ANOVA LME models had the following structure:

$$y_{ijkl} = \beta_0 + year_j + graz_k + year \times graz_{jk} + (b_0)_l + (b_0)_{k,l} + \varepsilon_{ijkl}$$
$$i = (1, \dots, 90),\ j = (1, \dots, 10),\ k = (1, 2, 3),\ l = (1, 2, 3)$$
$$b_l \sim N(0, \sigma_1^2),\ b_{k,l} \sim N(0, \sigma_2^2),\ \varepsilon_{ijkl} \sim N(0, \sigma^2) \qquad \text{eqn 2}$$

Finally, we examined the amount of variation in species composition from year to year by calculating the coefficient of variation (CV = SD/mean) for DCA1 site scores for each plot over the 10-year study period, creating the variable DCA_{CV}. This variable (y) was subjected to ANOVA LME modelling with grazing treatment (subscripted as j) as the categorical fixed effect and random effects for the saturated model specified as (i) watershed (subscripted as k), (ii) grazing treatment nested within watershed and (iii) geomorphology (subscripted as l) nested within grazing treatment nested within watershed:

$$y_{ijkl} = \beta_0 + graz_j + (b_0)_k + (b_0)_{j,k} + (b_0)_{l,j,k} + \varepsilon_{ijkl}$$
$$i = (1, \dots, 9),\ j = (1, 2, 3),\ k = (1, 2, 3),\ l = (1, 2)$$
$$b_k \sim N(0, \sigma_1^2),\ b_{j,k} \sim N(0, \sigma_2^2),\ b_{l,j,k} \sim N(0, \sigma_3^2),$$
$$\varepsilon_{ijkl} \sim N(0, \sigma^2) \qquad \text{eqn 3}$$

Serial correlation and heteroscedasticity functions were fitted and tested to these models, which along with random and fixed effects were tested by model comparison using likelihood ratios of competing models as described below.

For both the ANCOVA and ANOVA approaches random effect terms were sequentially dropped and nested models compared with those with more parameters by likelihood ratio tests. If models were significantly different ($P < 0{\cdot}05$), the model with the lower Akaike's information criterion (AIC) was selected, otherwise we selected the model with fewer parameters. We fitted all models with both correlated and heteroscedastic error structures but dropped these parameters if they did not significantly improve the model as determined by likelihood ratio tests ($P > 0{\cdot}05$; Pinheiro & Bates 2000). Specifying correlated variance–covariance matrices allows data collected from the same experimental unit over time to be treated as independent observations (Crawley 2002). To account for serial correlation of observations from the same plot over years (i.e. repeated measures), we used an autoregressive function (AR1) that fits a parameter describing residuals whose correlation declines exponentially with time. Once the random effects and error matrices were in place, residual vs. fitted plots were assessed visually for randomness and quantile–quantile plots were examined for departures from normality for each random factor and residuals, as suggested by Pinheiro & Bates (2000).

The significance of fixed effects was determined by dropping terms sequentially and comparing models with likelihood ratio tests. Treatment levels of significant terms were then sequentially combined and subsequent models compared with likelihood ratio tests to 'separate means'. For these comparisons, the full optimization maximum likelihood (ML) algorithm was used because likelihood ratio tests on models with different fixed effects parameters fitted with REML are not interpretable (Pinheiro & Bates 2000).

Results

COVER

For total vegetation cover, the most likely random effects structure with the highest likelihood included three nested categorical terms (model 1 in Table 1). Serial correlation amongst residuals from year to year was tested by including an autoregressive correlation coefficient in model 4, but the likelihood ratio test showed this model offered no improvement over the simpler model 1. To test for heteroscedasticity we fitted separate error variances for each level of grazing (model 5), which improved the model significantly.

With the random effects and variance–covariance matrix settled, we assessed the importance of fixed effects after refitting model 5 using the full optimization ML method (i.e. model 6 in Table 1 comprises the same parameter structure as model 5 but parameter estimates are made with ML rather than REML). We then sequentially dropped the fixed terms geomorphology (model 7) and grazing (model 8) and compared these models with likelihood ratio tests. Dropping the geomorphology term did not significantly alter the likelihood of the model, so the more general model 7 was retained. When the grazing fixed effect was dropped, the resulting model 8 had a significantly different likelihood than model 7, whose AIC was lower, and thus retained.

In model 9 (Table 1) ungrazed (NG) and lightly grazed (LG) treatment levels were combined for comparison with the more complicated model 7, which maintained all three treatment levels. As these models were not different, we concluded that the slope of cover over time was not different between NG and LG plots, and these treatment levels could be combined without significantly altering the predictive ability of the model. However, using this same approach we concluded that NG and LG treatments had slopes that were significantly different from moderately grazed (MG) plots with respect to cover (i.e. models 10 and 11 were significantly less likely than model 7). Model 9's fixed effects coefficients (data not shown) indicated that total vegetation cover was declining under MG and increasing under NG and LG plots.

DIVERSITY

No significant fixed effects were determined for ANCOVA LME models predicting species richness, evenness or Shannon's diversity index. However, using model selection criteria as above, model 6 was more likely than models 7 and 8 (Table 2) indicating, a significant year × grazing × geomorphology interaction was observed for Simpson's diversity index, which emphasizes species dominance (Mueller-Dombois & Ellenberg 1974). Rather than interpret the three-way interaction directly, we separated the data set along geomorphology lines, i.e. springs and creeks. Subsequent model selection showed the grazing term could be dropped from spring models without reducing the explanatory value (model S5; Table 2). Creek models for Simpson's diversity index showed that all treatment levels were significantly different from each other in the following sequence: LG < MG < NG (models C6–C8; Table 2).

Using the ANOVA LME approach, no significant differences amongst grazing treatments or years were found at springs for any of the diversity metrics. At creeks however, evenness and the Shannon diversity index were significantly greater on the MG plots than either NG or LG plots (Table 3).

COMPOSITION

For both springs and creeks our index of species composition was the first DCA ordination axis extracted from species × plot matrices (DCA1). In both cases the eigenvalue associated with these axes were quite high ($\lambda_{springs} = 0{\cdot}79$, $\lambda_{creeks} = 0{\cdot}84$), indicating strong gradients

Table 1. ANCOVA linear mixed effects model selection procedure for the response variable total vegetation cover

Model	Fixed effects	Random effects	Variance–covariance structure	d.f.*	AIC	Log L	Model test	L ratio	*P*	Interpretation of model comparison and subheader conclusion
Selecting random effects structure										
1	Year:Grazing:Geomorph†	1 | Watershed/Grazing/Geomorph‡	Equal var – Zero cov	11	–163·04	92·52				
2	Year:Grazing:Geomorph	1 | Watershed/Grazing	Equal var – Zero cov	10	–130·21	75·10	1 vs. 2	34·83	< 0·0001	Models are sig. diff.– > choose lower AIC (model 1)
3	Year:Grazing:Geomorph	1 | Watershed	Equal var – Zero cov	9	–113·03	65·51	1 vs. 3	54·08	< 0·0001	Models are sig. diff.– > choose lower AIC (model 1) Three levels of random effects represent best model given the data
Testing var-cov structure										
4	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	Equal var – corAR1§	12	–162·23	92·50	1 vs. 4	1·22	0·26	Models not sig. diff.– > choose model 1 because simpler
5	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	varIdent¶ – Zero cov	13	–169·62	97·81	1 vs. 5	10·58	0·005	Models are sig. diff.– > choose lower AIC (model 5) Heteroscedastic variances represent best model
Selecting fixed effects structure										
6**	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	13	–226·44	126·22				
7	Year:Grazing	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	10	–225·14	122·57	6 vs. 7	7·20	0·06	Models not sig. diff.– > choose model 7 b/c simpler
8	Year	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	8	–204·46	110·24	7 vs. 8	24·67	< 0·0001	Models are sig. diff.– > choose lower AIC (model 7) Differences in slope estimates for grazing intensity improve model
Separating means										
9	Year:(NG-LG vs. MG)††	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	9	–226·97	122·49	7 vs. 9	0·16	0·68	Combining NG and LG treatment levels no different than keeping the separate
10	Year:(NG-MG vs. LG)	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	9	–207·42	112·71	7 vs. 10	19·73	< 0·0001	Combining NG and MG significantly reduces deviance explained
11	Year:(LG-MG vs. NG)	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	9	–206·48	112·24	7 vs. 11	20·65	< 0·0001	Combining LG and MG significantly reduces deviance explained

*d.f., number of parameters estimated.
†Use of colon (:) indicates specification of interaction terms but no main effects. Multiplication sign (×) indicates main effects + interaction terms.
‡Vertical bar (|) separates continuous from categorical random effects in S-plus syntax. The forward slash (/) specifies nesting with general to specific moving from left to right. The random effects syntax for model 1 specifies random intercepts for each level of watershed, grazing within watershed, and geomorphology within grazing/watershed. A '1' to the left of the vertical bar indicates no random slopes should be fitted, only intercepts for those terms to the right of the vertical bar.
§corAR1 specifies residual correlation that decays exponentially with distance between time periods (i.e. years).
¶varIdent specifies separate error terms for each level of grazing (i.e. heteroscedasticity).
**Model 6 has a parameter structure identical to model 5 but estimates made with full optimization maximum likelihood rather than restricted maximum likelihood so that comparisons of models with different fixed effects terms could be performed.
††Combined treatment levels NG (not grazed) and LG (light grazed) for comparison with MG (moderately grazed).

Table 2. ANCOVA linear mixed effects model selection procedure for the response variable Simpson's diversity index (syntax explained in Table 1 footnotes)

Model	Fixed effects	Random effects	Variance–covariance	d.f.*	AIC	Log L	Model test	L ratio	*P*	Interpretation of model comparison and subheader conclusion
Selecting random effects structure										
1	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	Equal var – Zero cov	11	−96·37	59·18				
2	Year:Grazing:Geomorph	1 | Watershed/Grazing	Equal var – Zero cov	10	−87·48	53·74	1 vs. 2	10·88	0·001	Models sig. diff.– > choose lower AIC (model 1)
3	Year:Grazing:Geomorph	1 | Watershed	Equal var – Zero cov	9	−24·18	21·09	1 vs. 3	76·19	< 0·0001	Models sig. diff.– > choose lower AIC (model 1) Three levels of random effects (model 1) represent best model given the data
Testing var-cov structure										
4	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	Equal var – corAR1	12	−107·04	65·52	1 vs. 4	12·67	0·0004	Correlation parameter improves model
5	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	14	−109·06	68·53	1 vs. 5	6·02	0·049	Variance estimates for each grazing level improve model Model 5 the best model given the data up to this point
Selecting fixed effects structure										
6*	Year:Grazing:Geomorph	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	14	−160·25	94·13				Fitted with ML (maximum likelihood) instead of REML (restricted ML)
7	Year:Grazing	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	11	−153·48	87·74	6 vs. 7	12·77	0·005	Cannot drop geomorph, choose model 6
8	Year:Geomorph	1 | Watershed/Grazing/Geomorph	varIdent – Zero cov	10	−158·82	89·41	6 vs. 8	9·43	0·05	Cannot drop grazing, choose model 6
Separate model into springs and creeks										
Springs										
Selecting random effects structure										
S1	Year:Grazing	1 | Watershed/Grazing	Equal var – Zero cov	7	−55·68	34·84				
S2	Year:Grazing	1 | Watershed	Equal var – Zero cov	6	−28·35	20·18	S1 vs. S2	29·33	< 0·0001	Models sig. diff.– > choose lower AIC (model S1)
Testing var-cov structure										
S3	Year:Grazing	1 | Watershed/Grazing	varIdent – Zero cov	9	−54·63	36·31	S1 vs. S3	2·95	0·23	Unequal variances do not improve model (choose model S1)
S4	Year:Grazing	1 | Watershed/Grazing	Equal var – corAR1	10	−61·39	40·69	S1 vs. S4	8·76	0·003	Correlation parameter improves model (choose model S4)
Selecting fixed effects structure										
S5	Year	1 | Watershed/Grazing	Equal var – corAR1	8	−64·33	40·17	S4 vs. S5	1·05	0·59	Dropping grazing improves the model
Creeks										
Selecting random effects structure										
C1	Year:Grazing	1 | Watershed/Grazing	Equal var – Zero cov	7	−83·24	48·62				
C2	Year:Grazing	1 | Watershed	Equal var – Zero cov	6	−48·44	30·22	C1 vs. C2	36·80	< 0·0001	Models sig. diff.– > choose lower AIC (model C1)
Testing var-cov structure										
C3	Year:Grazing	1 | Watershed/Grazing	varIdent – Zero cov	9	−79·28	48·64	C1 vs. C3	0·05	0·977	Unequal variances do not improve model (choose model C1)
C4	Year:Grazing	1 | Watershed/Grazing	Equal var – corAR1	8	−84·82	50·41	C1 vs. C4	3·58	0·058	Correlation parameter does not improve model (choose model C1)
Selecting fixed effects structure										
C5	Year	1 | Watershed/Grazing	Equal var – Zero cov	5	−78·69	44·34	C1 vs. C5	8·55	0·014	Cannot drop grazing, choose model C1
Separating means										
C6	Year:(NG-LG vs. MG)	1 | Watershed/Grazing	Equal var – Zero cov	5	−77·63	43·82	C1 vs. C6	9·60	0·008	Combining NG and LG significantly reduces deviance explained
C7	Year:(NG-MG vs. LG)	1 | Watershed/Grazing	Equal var – Zero cov	5	−78·61	44·31	C1 vs. C7	8·62	0·013	Combining NG and MG significantly reduces deviance explained
C8	Year:(LG-MG vs. NG)	1 | Watershed/Grazing	Equal var – Zero cov	5	−78·21	44·10	C1 vs. C8	9·03	0·011	Combining LG and MG significantly reduces deviance explained

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Table 3. ANOVA linear mixed effects model selection for the response variables evenness and Shannon diversity index [as a difference from pre-treatment (1992) values] at creeks (syntax explained in Table 1 footnotes)

Model	Fixed effects	Random effects	Variance–Covariance structure	d.f.	AIC	Log L	Model test	L ratio	*P*	Interpretation of model comparison and subheader conclusion
Evenness index										
Selecting random effects structure										
1	Grazing	1 | Year/Watershed/Grazing	Equal var – Zero cov	7	−49·9043	31·95214				
2	Grazing	1 | Year/Watershed	Equal var – Zero cov	6	−51·9043	31·95214	1 vs. 2	0·00001	0·9982	Models NS diff.– > choose simpler model (model 2)
3	Grazing	1 | Year	Equal var – Zero cov	5	−34·9449	22·47245	2 vs. 3	18·95937	< 0·0001	Models sig. diff.- > choose lowest AIC (model 2)
Testing var-cov structure										
4	Grazing	1 | Year/Watershed	Equal var – corAR1	7	−49·9117	31·95585	2 vs. 4	0·007	0·931	Models NS diff.– > choose simpler model (model 2)
5	Grazing	1 | Year/Watershed	varIdent – Zero cov	9	−51·1665	33·58327	2 vs. 5	3·262262	0·1957	Models NS diff.– > choose simpler model (model 2)
Selecting fixed effects structure										
6	NG-LG vs. MG	1 | Year/Watershed	Equal var – Zero cov	5	−73·0024	41·50121	2 vs. 6	0·42	0·52	NG NS diff. than LG
7	NG-MG vs. LG	1 | Year/Watershed	Equal var – Zero cov	5	−8·41708	9·20854	2 vs. 7	65·00	< 0·0001	NG sig. diff. than MG
8	LG-MG vs. NG	1 | Year/Watershed	Equal var – Zero cov	5	−3·71023	6·85512	2 vs. 8	69·71	< 0·0001	LG sig. diff. than MG
Shannon diversity index										
Selecting random effects structure										
1	Grazing	1 | Year/Watershed/Grazing	Equal var – Zero cov	7	161·7401	−73·87006				
2	Grazing	1 | Year/Watershed	Equal var – Zero cov	6	159·7401	−73·87006	1 vs. 2	0·000	1·00	Models NS diff.– > choose simpler model (model 2)
3	Grazing	1 | Year	Equal var – Zero cov	5	171·7329	−80·86645	2 vs. 3	13·99277	0·00	Models sig. diff.– > choose lowest AIC (model 2)
Testing var-cov structure										
4	Grazing	1 | Year/Watershed	Equal var – corAR1	7	156·9192	−71·4596	2 vs. 4	4·82	0·03	Models sig. diff.– > choose lowest AIC (model 4)
5	Grazing	1 | Year/Watershed	varIdent – Zero cov	9	156·7267	−69·36333	4 vs. 5	4·19	0·12	Models NS diff.– > choose simpler model (model 4)
Selecting fixed effects structure										
6	NG-LG vs. MG	1 | Year/Watershed	Equal var – Zero cov	6	146·0676	−67·03382	4 vs. 6	3·53	0·06	NG NS diff. than LG
7	NG-MG vs. LG	1 | Year/Watershed	Equal var – Zero cov	6	180·3466	−84·17328	4 vs. 7	37·81	< 0·0001	NG sig. diff. than MG
8	LG-MG vs. NG	1 | Year/Watershed	Equal var – Zero cov	6	187·8132	−87·90662	4 vs. 8	45·28	< 0·0001	LG sig. diff. than MG

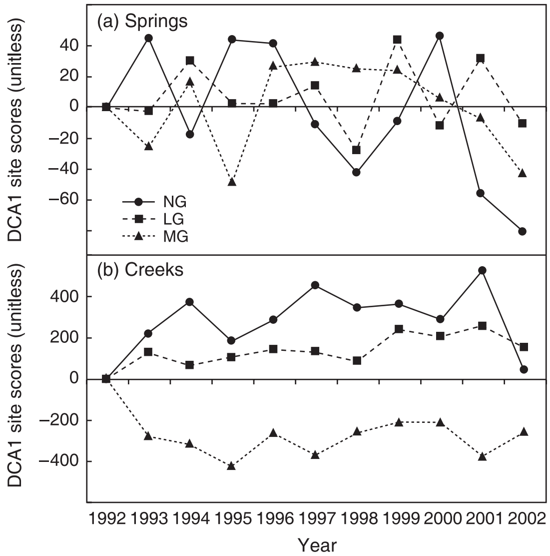

Fig. 3. Means by year of DCA1 site scores normalized to 1992 pre-treatment values for (a) springs and (b) creeks. Raw site scores were subjected to ANCOVA LME with year as the predictor variable, but no significant temporal trends were observed. ANOVA LME showed that normalized site scores were significantly different amongst all three grazing treatments at creeks but not springs, indicating alternative, relatively stable equilibria at the former and non-equilibrium at the latter.

in species space. No trends in DCA1 site scores were determined with the ANCOVA LME approach for either springs or creeks. However, ANOVA LME confirmed the pattern demonstrated in Fig. 3b, that soon after grazing treatment establishment creek vegetation equilibrated at more or less stable compositional states, while spring vegetation did not (Fig. 3a). The creek vegetation was following no directional trajectory, i.e. no significant slopes with the ANCOVA LME, so it was deemed stable. But the vegetation was significantly different amongst the three grazing treatments (models 7–9; Table 4), indicating the existence of distinctive compositional states.

The ANOVA LME for variability in species composition (DCA_{CV}) over years showed that both geomorphology and grazing were significant fixed effects, but no interaction between these factors was observed. Variability in composition was greater at springs than creeks (Fig. 4) but grazing effects were similar in each geomorphological type. Comparison of models comprising all three grazing treatments with those combining two grazing treatments showed that NG could not be dropped without significantly increasing residual deviance. Figure 4 shows that DCA_{CV} in NG plots was greater than LG and MG plots.

Discussion

Equilibrium dynamics were evident in these resource-rich systems, but the strength of the evidence was context dependent (Fig. 5). We did not reject our first hypothesis that herbaceous cover would negatively co-vary with

Table 4. ANOVA linear mixed effects model selection for the response variable DCA1 site scores (normalized to 1992 pre-treatment values) at creeks (syntax explained in Table 1 footnotes)

Model	Fixed effects	Random effects	Variance–Covariacne structure	d.f.	AIC	Log L	Model test	L ratio	P
Selecting random effects structure									
1	Year × Grazing	1 | Watershed/Grazing	Equal var – Zero cov	33	898·98	−416·49			
2	Year × Grazing	1 | Watershed	Equal var – Zero cov	32	899·93	−417·96	1 vs. 2	2·94	0·09
Testing var-cov structure									
3	Year × Grazing	1 | Watershed	Equal var – corAR1§	33	901·09	−417·54	2 vs. 3	0·84	0·36
4	Year × Grazing	1 | Watershed	varIdent – Zero cov	61	908·24	−393·12	2 vs. 4	49·68	0·009
Selecting fixed effects structure									
5*	Year × Grazing	1 | Watershed	Equal var – Zero cov	32	1042·27	−489·13			
6	Grazing	1 | Watershed	Equal var – Zero cov	5	1013·81	−501·14	5 vs. 6	24·81	0·58
Contrasting grazing means									
7	NG-LG vs. MG	1 | Watershed	Equal var – Zero cov	4	1021·087	−506·54	6 vs. 7	10·006	0·0016
8	NG-MG vs. LG	1 | Watershed	Equal var – Zero cov	4	1100·291	−546·15	6 vs. 8	89·2095	< 0·0001
9	LG-MG vs. NG	1 | Watershed	Equal var – Zero cov	4	1070·351	−531·18	6 vs. 9	59·2697	< 0·0001

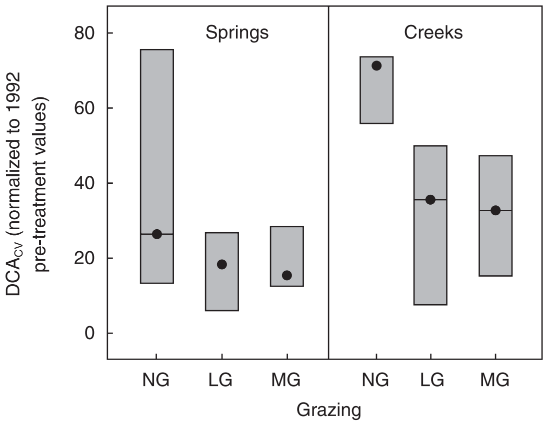

Fig. 4. Boxplots of the coefficient of variation of DCA1 site scores (DCA_{CV}) over the 10-year study period. (dot, median, lower and upper edge of boxes; boxes, interquartile range of the data). The species composition of ungrazed plots was more variable than both grazed treatment levels.

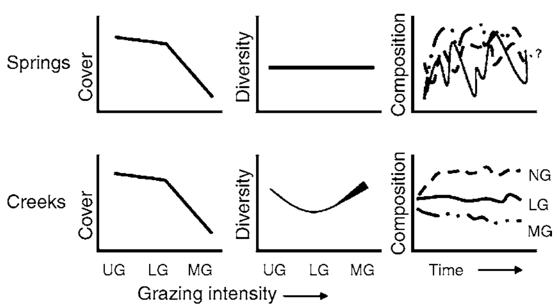

Fig. 5. Patterns we observed at springs and creeks. NG, ungrazed; LG, lightly grazed; MG, moderately grazed. The fatter line for the creek diversity plot indicates that three of four diversity metrics (Shannon's, Simpson's and evenness) were significantly greater under MG.

grazing intensity, lending support to equilibrium theory but also validating the efficacy of our MG treatment. Total covers on LG and NG plots were not significantly different but were both increasing over time. No general diversity patterns were observed. However, our highest grazing intensity level, MG, resulted in greater diversity and evenness at creek systems, which is not surprising given the lower values of plant cover observed in these creeks. Predictions of the intermediate disturbance hypothesis (IDH; Connell 1978; Mackey & Currie 2000), an equilibrium-based response, were not supported, but not because diversity was not related to grazing intensity. Instead, we observed a U-shaped response, where diversity was lowest under our intermediate grazing treatment. Our third hypothesis, that species composition would be controlled by grazing intensity, was strongly supported at creeks but not at all for springs. Composition at springs fluctuated wildly from year to year, while creeks supported alternative, relatively stable vegetation states for each grazing intensity treatment.

Both systems behaved in a more or less equilibrium manner, depending upon the metric used. Perhaps more important than determining whether or not equilibrium dynamics were exhibited, was the observation that in the absence of grazing disturbance these systems tended towards non-equilibrium dynamics, where species composition was much more variable from year to year when grazing was removed and the effects of environmental fluctuations were damped under grazing. This was a general result for both springs and creeks. However, these two separate, but linked, systems also behaved quite differently and would find different niches along Wiens's (1984) equilibrium–non-equilibrium continuum. Very little control of grazing disturbance on plant composition was exhibited at emergent springs, which appeared to be more entrained by year to year variability than creeks (Fig. 3). The creeks very clearly reached alternative equilibrium states that were determined by differences in grazing intensity. This disparity is interesting given these systems are separated by only several metres in all cases. A possible explanation is differences in the geomorphology of these systems. Springs tend to be non-channelized marshy areas with slow subsurface flow, while creeks are by definition channelized with surface flow (Allen-Diaz, Jackson & Fehmi 1998). Along creek channels there is usually a moisture gradient emanating up-slope from the midpoint of the channel that is reflected as a plant composition gradient. Hence a change in water quantity might differentially affect springs and creeks because the entire spring area would be affected by a dry-down, for instance, while the same dry-down conditions at creeks would simply shift the moisture gradient and the corresponding plant community in space. This speculation remains an untested hypothesis.

Many have shown that disturbance promotes diversity (Collins *et al.* 1998; but see Tilman 1983; Pollock, Naiman & Hanley 1998; Mackey & Currie 2000). However, disturbance–diversity relationships have not been consistently demonstrated for wetland systems (Keddy 2000; Moore & Keddy 1989), especially in Mediterranean climates such as California's where interannual weather variability is high. This variability has been noted as key to the development of California's many and varied riparian ecosystems (Gasith & Resh 1999). Disturbance–diversity relationships are often unimodal (Grime 1973; Sousa 1979; Huston 1994) as described by the above-mentioned IDH. A shortcoming of this hypothesis is the conflation on the abscissa of disturbance and productivity (biomass) gradients (Pollock, Naiman & Hanley 1998). As a result, highly disturbed, highly productive ecosystems (such as spring-fed riparian zones) do not fit well into this model. Huston's (1994) dynamic equilibrium model (DEM) allows for the intermediate disturbance level to scale positively with community productivity such that greater disturbance is required to achieve intermediate status along an increasing productivity gradient. Hence our lightly grazed wetlands may have been receiving qualitatively different disturbance treatments than our lightly grazed creeks because of productivity differences, which were not measured but observed, even though grazing intensities on an animal density basis were identical.

Huston's (1994) DEM also allows for temporal or spatial heterogeneity to dampen the effect of disturbance on diversity (Pollock, Naiman & Hanley 1998). Spatial heterogeneity may be greater at creeks compared with wetlands because of the above-mentioned channelization of surface water flow at creek sites. Cross-sectional microsite gradients probably exist from mid-creek to bank edges, adding to the spatial heterogeneity of creeks. Hence moderately grazed creeks maintained greater diversity than either ungrazed or lightly grazed sites. If this is the case, our range of grazing intensity treatments may have encompassed only the ascending part of the unimodal response curve. Although we may not have implemented a full range of disturbances (e.g. we could have implemented a more severe grazing treatment), diversity responses to our disturbance gradient in the long-term study indicated that moderate grazing should promote greatest species co-existence in creek systems but not in springs.

Zedler & Beare (1986) showed that community dominance in estuarine salt marshes of southern California was related to rainfall–drought cycles, where dry periods favoured *Salicornia virginica* and *Spartina virginica* while *Typha domingensis* dominance was contingent upon the length of flooding in high rainfall periods. A similar situation in the Sierra Nevada foothills is plausible, where a disturbance in the form of flushing and scouring of spring-fed wetlands in high-rainfall years creates a more variable environment for alternative plant groups that expand and contract depending on prevailing conditions. Grazing disturbance maintained a more static vegetation state, but release from grazing relinquished more control of the community to the environment, or at least rendered it more sensitive. In a similar vegetation community, the Pampas of South America, Facelli, Leon & Deregibus (1989) found that ungrazed paddocks varied more interannually than grazed paddocks dominated by the bunchgrass *Paspalum dilatatum*, a dominant grass in our ecosystem. Curiously, this is opposite to responses of warm-season grasses of the tallgrass prairie (Collins *et al*. 1998; Knapp *et al*. 1999), but those systems receive high-frequency burning treatments, which tend to decrease plant diversity (Turner, Seastedt & Dyer 1993).

Our findings describe highly dynamic systems, more or less entrained by environmental stochasticity depending on the disturbance regime. They appear to more closely follow environmental phenomena, and therefore exhibit more variability, when grazing disturbance is removed, i.e. non-equilibrial. A more static, equilibrium vegetation state appears to be maintained by grazing disturbance, especially at creeks. We predicted that equilibrium dynamics would prevail at these resource-rich wetlands. We based this prediction on the idea that competitive exclusion is subverted by grazing disturbance, whose removal would instigate a reduction in diversity. A result of the competitive exclusion process is that, via this mechanism, an equilibrium plant community will emerge dominated by competitively superior species. Light to moderate grazing on spring-fed wetlands and their resultant creeks can maintain herbaceous cover and diversity. However, total cover should be frequently monitored in order to assess potentially undesirable trends in the amount of bare ground exposed at higher grazing intensities. Because equilibrium dynamics were evident to some degree, grazing intensities for these systems can be adjusted to manipulate community structure to achieve management goals.

By stratifying geomorphological types, we showed a differential response from proximate systems, demonstrating the need for management decisions based on site-specific criteria. For example, one might adjust grazing intensities specifically for rehabilitation of a marshy spring area differently than for a channelized riparian zone. As a consequence, simultaneous management for multiple types of riparian and matrix vegetation will require persistent monitoring and flexibility. More of the type of information generated from this study, long-term and experimental, is needed for sound adaptive management of highly variable riparian ecosystems of arid and semi-arid regions, but especially those where a Mediterranean climate prevails.

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ORIGINAL PAPER

Does Long-term Grazing by Pack Stock in Subalpine Wet Meadows Result in Lasting Effects on Arthropod Assemblages?

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Abstract Pack stock are often used in mountain environments and are grazed in uplands and wetlands, particularly subalpine wet meadows. Effects of pack stock on wetland invertebrates are unknown. Sequoia National Park, (Sierra Nevada, USA), was an ideal location for the study of lasting stock impacts on fauna, because a) there was an 18-year database of stock usage, b) there were meadows with little grazing that could be contrasted with grazed meadows, c) there is a long winter with no stock use, and d) the start of grazing for each meadow is controlled, so we could sample after greenup but just before stock arrived. We could thus address persistent conditions produced by many years of stock use in isolation from any potential short term impacts. We sampled terrestrial arthropods in paired "grazed" and "ungrazed" meadows across the Park and collected associated vegetation data. We found some negative effects of grazing on vegetation structure, but few lasting negative or positive effects of long-term stock grazing on arthropods in these wetlands. Although it appears that pack stock do not cause lasting damage to this arthropod assemblage, the extent of impact at the height of the grazing season remains unknown.

Keywords Araneae · Baiting · Disturbance · Insecta · Sierra Nevada (USA) · Sweep netting

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Introduction

Pack stock are frequently used on public lands in the Sierra Nevada (USA) and in other mountain environments, and these mules and horses, and occasionally burros and llamas, are often grazed in subalpine wet meadows (McClaran 1989; Spildie et al. 2000; Cole et al. 2004). The timing and locations for grazing are regulated in some areas (McClaran 1989; Moore et al. 2000; Spildie et al. 2000), but impacts to vegetation assemblages nevertheless can occur (Weaver and Dale 1978; Olson-Rutz et al. 1996a; Moore et al. 2000; Cole et al. 2004), and recovery is often not rapid (Olson-Rutz et al. 1996b; Spildie et al. 2000).

Although direct and indirect effects of outdoor recreation on invertebrates in vegetated assemblages have been demonstrated in a variety of ecosystems (e.g., Duffey 1975; Bayfield 1979; Eckrich and Holmquist 2000; Uhrin and Holmquist 2003), we are unaware of any studies on the effects of pack stock grazing on wetland invertebrates, despite the importance of invertebrates to the functional ecology of these habitats (van der Valk 2006; Williams 2006; Batzer and Sharitz 2006). Studies addressing use of grasslands, wetlands, and other vegetated habitats by different large mammals reveal mixed effects on the invertebrate assemblage (González-Megías et al. 2004; Underwood and Fisher 2006). Kruess and Tscharntke (2002) found cattle grazing to affect insects more than the plant assemblage, Bestelmeyer and Wiens (1996) recorded lower ant species richness as a function of cattle and goat grazing, and González-Megías et al. (2004) determined that sheep, goat, and ibex lowered diversity and abundance of arthropods in a Mediterranean mountain environment. Rambo and Faeth (1999) reported that deer, elk, and cattle reduced abundance, but not richness or evenness, of insects in a pine-grassland assemblage. Mysterud et al. (2005) found that sheep grazing in alpine pastures did not affect diversity or abundance of insects, and similarly Heske and Campbell (1991) and Bestelmeyer and Wiens (2001) discovered few differences in ant species richness, abundance, or assemblage structure as a function of livestock grazing. At the other end of the spectrum, Bock et al. (2006) found that grazing of small ranches by horses, cattle, and sheep can increase grasshopper abundances, and Majer and Beeston (1996) found higher ant species richness in more heavily grazed areas. Arthropod diversity can be increased by grazing via indirect effects mediated by shifts in canopy height, structural complexity, and plant diversity (Morris 1990; Olff and Ritchie 1998). Generalization concerning arthropod response to grazing across habitats is difficult due to the complex interactions of many factors (see literature surveys in González-Megías et al. 2004 and Underwood and Fisher 2006). Response of arthropods differs as a function of many variables, including livestock density, differences in grazing behavior, vegetation assemblage, duration of studies, arthropod response variables of interest, and especially duration of disturbance by livestock and the amount of time since last disturbance. Sierra Nevada wetlands were historically not grazed by herbivores larger than mule deer (*Odocoileus hemionus* Rafinesque, see Loomis et al. 1991; Loft et al. 1991; Dull 1999), and larger herbivores with different foraging behavior might be expected to cause shifts in both animal and plant assemblages.

If meadows that have been subject to long-term stock disturbance were to be sampled during a period of stock use, it would be difficult to determine if any apparent impacts were a function of long-term use, current use, or a combination thereof. In this study, we address one question regarding potential impacts of pack stock grazing on arthropod assemblages: Does grazing cause lasting effects that persist over time, or do long winters without stock allow an annual recovery of arthropod assemblages from any impacts that occur during summer usage? We do not address effects on wetland arthropod assemblages at the height of the grazing season. Sequoia National Park was an ideal location for this study, because a) there was a detailed, 18-year, meadow-specific database of stock usage, b) there were many meadows with little or no grazing use that could be contrasted with grazed meadows, c) there is a long winter period with no stock use, and d) the opening date for grazing on each meadow is controlled by the Park, so we could sample after greenup but just before stock arrived. Sampling prior to stock arrival allowed us to address lasting effects of many years of stock use in isolation from potentially confounding effects of current use.

Methods

We compared subalpine wet meadows with and without substantial stock use using a paired design. Generalization from responses of a single group of arthropods can be misleading (Gibson et al. 1992), so we examined effects across all canopy and ground-dwelling arthropod taxa that were collected by sweep nets and baits.

Study Area and Sites

Sequoia National Park is located in the southern Sierra Nevada mountains of California, USA (Fig. 1). The subalpine meadows that are open to pack stock are usually covered by snow for eight months, as are the 3000+ meter passes that allow access to these wetlands. Stock are not allowed into wetlands prior to snowmelt, and the National Park Service determines opening date for each meadow on the basis of the amount of snowfall for a given year, the timing of snowmelt, and the observed condition of the wetland in a given year. Access to selected wetlands is usually allowed about one month after snowmelt, i.e., individual wetlands are typically opened at various times from mid-June to mid-July. Stock use is intermittent from opening date through August, decreases rapidly after August, and ceases completely when the first substantial snow falls, generally in early November. Most stock use thus occurs over a two–three month period. Stock parties that used our study meadows ranged in size from one to 20 animals ($x = 8.2$, SE=0.68) in the five years preceding the study.

Wet meadows are saturated with water much of the year (Williams 2006; Mitsch and Gosselink 2007), and these high diversity oases are common features of the subalpine environment (Körner 2003). The vegetation assemblages in the subalpine wet meadows managed for stock in Sequoia National Park are often dominated by a reed grass, *Calamagrostis muiriana* B.L. Wilson and S. Gray, formerly included in *C. breweri* Thurber, and we therefore focused our study on this assemblage (known as the Shorthair Reedgrass Herbaceous Alliance in the Sequoia/Kings Canyon National Park vegetation classification). Other important taxa include mountain ricegrass *Ptilagrostis kingii* (Bolander) Barkworth, tufted hairgrass *Deschampsia cespitosa* (L.) Beauv., various sedges *Carex* spp., rushes *Juncus* spp., shooting star *Dodecatheon* sp. (c.f. *subalpinum* Eastw.), club-moss Ivesia *Ivesia lycopoidioides* A. Gray, tundra aster *Oreostemma alpigenum* (Torr. and A. Gray) Greene, dwarf bilberry *Vaccinium caespitosum* Michaux, pussy-toes *Antennaria* spp., and western bistort *Polygonum bistortoides* Pursh. Klikoff (1965), Benedict (1983), and Ratliff (1985) provide good overviews of this assemblage. Assemblages characterized by higher levels of soil moisture, such as fens or wet meadows dominated by *Deschampsia cespitosa* and large sedges (e.g., *Carex utriculata* L. Bailey), are also grazed in the Sierra (Stohlgren et al. 1989).

We wished to contrast sites that had a long history of pack stock use with sites that had an equally long history of

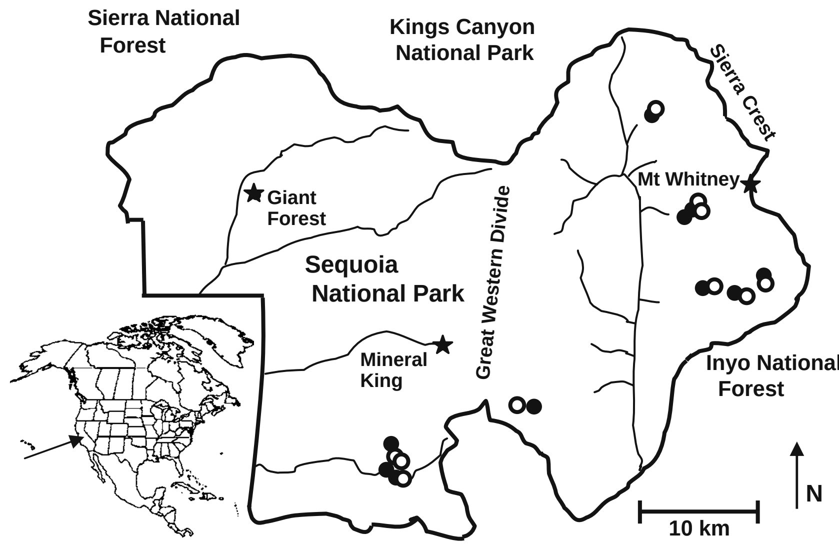

Fig. 1 Sequoia National Park in the Sierra Nevada mountains in California, USA. Black circles represent grazed meadows paired with meadows with minimal stock use (white circles). Sites are separated slightly for clarity

minimal use. Sequoia National Park has detailed records of pack stock use for many individual wetlands that span the last 18 years, as well as older, less formal records. We used these records to select 10 subalpine wet meadows (Table 1) that a) had been exposed to consistent use by pack stock (henceforth "grazed"), b) could each be paired with a subalpine wet meadow with little or no recent stock use ("ungrazed"), and c) were dominated by reed grass. Note that "grazed" in this context refers to all aspects of stock usage, including trampling, rather than cropping alone. Grazed and ungrazed conditions served as the mensurative treatments (Hurlbert 1984) for this study. We were able to locate pairs that were separated by an average of only 0.96 km (SE=0.16) and 59 m (SE=14) of elevation and that were in the same watersheds. We emphasized the tight co-location of paired grazed and ungrazed meadows in part to minimize geophysical and botanical differences. We also wished to minimize potential differences as a function of changing weather conditions, and the close proximity of each pair allowed us to sample both sites in rapid succession, before wind speed, air temperature, etc. could change greatly. Although each pair of wet meadows was tightly co-located, meadow blocks were separated by as much as 40 km, and up to two days of backpacking time was required to reach some sites. We thus sampled a relatively large number of blocks across a broad landscape

Table 1 Site characteristics and usage patterns of grazed meadows over the past 18 years and over the past five years. All sites were wet meadows, regardless of place names. Each site was paired with a nearby ungrazed site (Fig. 1).

			1990–2007			2003–2007		
Site	Elevation (m)	Hectares	Total stock nights	Mean stock nights/year	Mean stock nights/ha/yr	Total stock nights	Mean stock nights/year	Mean stock nights/ha/yr
Hockett Pasture	2595	3.6	2590	144	39.5	408	82	22.4
South Fork Mdw	2587	4.5	3465	193	43.4	746	149	33.5
South Fork Pasture	2600	4.5	1387	77	17.3	195	39	8.8
Penned-Up Mdw	3242	5.3	621	35	6.6	199	40	7.6
Nathan's Mdw	3061	5.9	1883	105	17.9	342	68	11.7
Lower Rock Ck Crossing	2893	25.5	2263	126	4.9	969	194	7.6
Lower Crabtree Mdw	3169	11.7	2042	113	9.6	585	117	10.0
Upper Crabtree Mdw	3192	17.0	2984	166	9.8	689	138	8.1
Tyndall Creek Mdw	3201	11.3	3352	186	16.4	711	142	12.6
Middle Rattlesnake Cyn	2907	5.3	1892	105	20.0	614	123	23.3
Mean	2945	9.5	2248	125	18.5	546	109	14.5
Std. Error	85	2.3	280	16	4.2	80	16	2.8

with good replicate dispersion (Hurlbert 1984). We wanted sites to be as close to mid-season condition as possible in terms of the structure of the vegetation and arthropod assemblages, so we waited to sample until one hour to three days before stock reached the wet meadows in June and July of 2008.

Each ungrazed or grazed site was sampled using a series of subsamples. We used aerial images of the sites to randomly select two 50×50 m subsample locations at each site prior to the field season. After arriving at each site, we established four additional sample locations, each at a randomly determined location within each of the pre-selected 50×50 m subsample locations. We used two of these sample locations for some metrics, and four for others (see below). Various metrics for a given ungrazed or grazed site were therefore means or composites of four or eight total measurements.

Field and Lab Methodology

Fauna We used sweep nets to sample the meadow canopy fauna, and we supplemented these collections with baits targeting ground dwellers, particularly ants. Sweep nets are conical framed nets with a handle (New 1998; Southwood and Henderson 2000) and have a number of advantages for sampling remote areas. These nets are light in weight, easily transportable, do not impact wilderness character, integrate collections over a wide area, collect sparsely distributed species, can be used in habitats that are flooded or saturated with water, and produce samples that require relatively little sorting. Sweep nets have been shown to yield higher numbers of individuals, species, families, and orders, and capture higher levels of diversity than pitfall traps, light traps, or scented traps (Gadagkar et al. 1990). Sweep netting is probably the most widely used method for sampling arthropods in vegetation (Southwood and Henderson 2000), and this technique has been used in other investigations of the effects of grazers on arthropods (Rambo and Faeth 1999; Mysterud et al. 2005).

The response variables for each ungrazed or grazed site were means of two 50-sweep samples, with one 50-sweep sample from each of the 50×50 m subsample locations. We used a collapsible sweep net with a 30.5 cm aperture and mesh size of 0.5×0.75 mm (BioQuip #7112CP). Each of the two 50-sweep samples was in turn a composite of two 25-sweep subsamples from within each 50×50 m area. We sampled a total of 400 square meters at each site. Strengths of this approach include the previously noted integration of a large area and sampling of less common taxa, but conversely small scale invertebrate-habitat relationships (e.g., Crist et al. 1992; With 1994; Wiens et al. 1997) could have been missed. Sweeping was our first activity at each subsample location, because the subsequent work would have been likely to have disturbed fauna. Each sample was transferred to a self-sealing bag, killed with 99% ethyl acetate (Triplehorn and Johnson 2005), and kept as cool as possible until the trailhead was reached and the samples could be transferred to a freezer. All sweep sampling was done by a single worker throughout the project so as to minimize variance.

Baiting (Bestelmeyer et al. 2000; Delabie et al. 2000) targets ants and may also collect other taxa (Alonso 2000; Andersen and Majer 2004). Baiting is commonly used to monitor ant assemblages (Bestelmeyer et al. 2000) and has many of the same advantages as sweep nets for sampling remote areas. Our pilot tests of various bait combinations in 50 subalpine wet meadows over several years showed that honey and tuna baits offered the best combination of field practicality and attractiveness to multiple ant taxa. We placed one honey and one tuna bait within each of the two 50×50 m subsample locations at each site immediately after sweep netting, each at one of the 25-sweep locations. The baits consisted of ~1 cm^2 portions of honey or tuna and were placed on green construction paper cards and weighted by rocks. After 30 min, ants were removed with forceps and placed in a vial containing 70% ethanol. This method worked more reliably than preserving the entire bait or using an aspirator. The data from the honey and tuna baits at each subsample location were combined, and the data from the two subsample locations were used to generate means for response variables at each ungrazed or grazed site.

We sorted sweep samples in the lab, and identified taxa to family, with the exception of mites. Morphospecies counts were made for each sample. We identified ants from the bait samples to species.

Vegetation and Physical Data We estimated percent green, standing brown (senescent), and litter cover as well as percent bare ground at the same two locations within each of the subsample locations that were used for sweep and bait samples. All of these metrics were visual cover estimates from a 10×10 m plot colocated with the area that was sweep netted. We measured canopy height and litter depth at two randomly selected locations within each area that was sweep netted. Cover estimates for each site were therefore means of four estimates, whereas canopy height and litter depth at each site were means of eight measures.

We recorded air temperature (in shade), relative humidity, and wind speed in the center of each 50×50 m subsample location using a Kestrel 3000 digital meter. These metrics were thus means of two measurements at each grazed or ungrazed site. Surface soil compaction was coarsely estimated with a Ben Meadows penetrometer at each of the canopy height/litter depth locations, thus yielding eight measurements per site.

Analysis

We performed 1×2 randomized block ANOVAs and ANCOVAs on a variety of faunal, vegetation, and physical metrics using SYSTAT 12. We analyzed a variety of faunal metrics, including order and family population abundances and family and morphospecies richness. Because large collections have more species than small collections, even if drawn from the same assemblage, we also assessed richness with expected number of species and families after scaling to the number of individuals in the sample with the fewest individuals ($E(S_8)$ and $E(F_8)$, Hurlbert 1971; Simberloff 1972; Magurran 2004). We analyzed family and morphospecies dominance and used probability of interspecific encounter, i.e., the probability that two species drawn from a sample are of different taxa, as a measure of evenness at both the morphospecies and family level (P.I.E., higher values indicate greater evenness, Hurlbert 1971). Margalef's index (D_{Mg}, Clifford and Stephenson 1975; Magurran 2004) was used as a diversity measure for both families and morphospecies. We calculated $E(S_8)$, $E(F_8)$, and P.I.E. using the application Diversity. Some metrics demonstrated departures from normality via Lilliefors tests (Lilliefors 1967) and/or showed heteroscedasticity (F_{max} and Cochran's tests; Cochran 1941; Kirk 1982), but square-root transformations ($(y)^{0.5}+(y+1)^{0.5}$) of proportional data and log transformations (log (y + 1)) of all other data allowed parametric assumptions to be met. Only variables that differ as a function of treatment and that are not likely to be affected by the treatment should be considered for further analysis as covariates (Underwood 1997), and site elevation qualified via these criteria (Table 2). Although elevation differences between grazed-ungrazed pairs were small, most grazed sites were slightly lower than their associated ungrazed sites, and it was therefore important to examine elevation as a covariate. We present ANCOVA (general linear model) results for all response variables except air temperature, which was necessary to exclude because this variable did not meet the assumption of homogeneity of treatment and covariate regression slopes (Sokal and Rohlf 1995; Underwood 1997). Lastly, we constructed rank abundance plots which provide an additional perspective on diversity, richness, and evenness, without collapsing a great deal of information into a single number (Stiling 2001; Magurran 2004; Underwood and Fisher 2006).

Because this study addressed potential anthropogenic impacts, we wanted good power and tight control over Type II error. Although ecologists tend to emphasize Type I error over Type II error, there is often not an ecological basis for this bias, particularly in situations that involve potential environmental degradation. It is increasingly recognized that both types of error deserve equal scrutiny, and it can be advantageous to set alpha as high or even higher than beta in order to increase power and decrease Type II error (Kendall et al. 1992; Mapstone 1995; Dayton 1998; Field et al. 2004). We used as many replicate ungrazed blocks (10 wet meadow pairs) as possible; we were not able to use more sites because of the limited number of subalpine wet meadows that met our criteria for pairing, so power could not be increased by increasing sample size. Before conducting our field work, we used G*Power (Erdfelder et al. 1996; Faul et al. 2007; Mayr et al. 2007), our known sampling design and sample size, and the standard *a priori* estimate for effect size of 0.5, which has been well-established both theoretically and empirically though large meta-analyses (Cohen 1988; Lipsey and Wilson 1993; Bausell and Li 2002) to estimate the *a priori* alpha level that would be required in order to have an equivalent beta error. The result was alpha=beta=0.19, and the associated power (1-beta) was 0.81. Note that this is *not* retrospective power analysis, which is not recommended (Hoenig and Heisy 2001; Nakagawa and Foster 2004). In contrast, the *a priori* beta estimate using these same parameters for a fixed alpha of 0.05 was 0.44 and a power of only 0.56, which would give good protection from Type I error, but poor protection from Type II error and therefore a greater chance of falsely assuming that pack stock have little effect on wet meadow arthropods. We used both alpha=0.19 and the standard alpha=0.05 as significance thresholds in order to provide additional perspective for our results.

Results

Both sets of meadows had >80% green vegetation, ~8 cm canopy height, equal soil compaction, and similar percent bare ground, but there were some differences in vegetation structure (Table 2). Grazed meadows had shallower litter depth as well as lower percent litter and brown vegetation cover (ANOVA). Some differences among meadow pairs (block effects) were apparent for these variables as well as for temperature and humidity. Effects were generally lessened when analyzed via ANCOVA, although both litter depth and cover were still different (Table 2) at alpha=0.19 (see Methods).

We collected and identified 2,683 arthropods in the study, representing 11 orders and 81 families. Diptera had the greatest family richness (29), followed by Hemiptera (12), Hymenoptera (12), and Coleoptera (10). There were 68 families in the ungrazed samples and 63 families in the grazed samples. Rank abundance plots for the two meadow conditions were similar (Fig. 2), and both plots fell between log normal and broken stick configurations. There was slightly more abundance at family ranks 7 through 20 on the grazed plots and slightly more abundance at ranks 20 through 43 on the ungrazed plots.

Table 2 Means, standard errors, and results of 1×2 randomized block ANOVAs (n=20; df=1,9) and ANCOVAs with elevation as a covariate (df=1,9,1) for vegetation and physical metrics. No ANCOVA for air temperature due to heterogeneity of treatment and covariate slopes

	Ungrazed		Grazed		ANOVA		ANCOVA	
	Mean	SE	Mean	SE	Block	Treatment	Block	Treatment
Elevation (m)	3013.9	90.5	2954.1	85.6	<0.01**	<0.01**	NA	
Canopy height (cm)	8.9	0.9	8.5	0.9	0.19	0.73	0.32	0.44
Litter depth (cm)	1.9	0.2	0.8	0.2	0.14*	<0.01**	0.18*	0.08*
Litter cover (%)	4.6	0.8	2.4	1.0	0.09*	0.02**	0.10*	0.12*
Bare ground (%)	7.8	2.9	12.6	3.7	0.91	0.33	0.95	0.55
Brown cover (%)	6.4	1.7	2.3	0.7	0.16*	0.03**	0.21	0.26
Green cover (%)	81.2	4.0	82.7	3.8	0.84	0.82	0.91	0.90
Wind speed (km/hr)	6.3	0.6	5.7	0.6	0.67	0.50	0.58	0.23
Air temperature (°C)	20.2	0.8	20.0	1.0	<0.01**	0.64	NA	
Humidity (%)	32.1	2.8	33.7	1.7	0.11*	0.39	0.20	0.72
Soil compaction (kg/cm^2)	1.5	0.1	1.5	0.2	0.75	0.98	0.83	0.67

*P<0.19 (see Methods); **P<0.05.

Sweep assemblage metrics for ungrazed and grazed meadows were almost identical when assessed via ANOVA or ANCOVA (Table 3). There was also little evidence of block effects at the assemblage level. The overall assemblage was dominated by Diptera and Hemiptera at the order level (Table 4). Three of the four most abundant families (and 7 of the top 10) were dipterans; anthomyiid flies had the highest overall family abundance, followed by cicadellid leafhoppers, ephydrid shore flies, and chloropid grass flies. Ungrazed and grazed plots had the same six most abundant families, although the rank order differed. Only Diptera and Hemiptera were found in all samples; at the family level, chloropids, muscid house flies, and anthomyiids were found in almost all samples (Table 4). Coleoptera was the only group that differed in abundance between ungrazed and grazed meadows when assessed with ANOVA, indicating more beetles on ungrazed sites, although beetles were relatively uncommon in the assemblage. ANCOVAs that included elevation as a covariate similarly did not reveal grazed-ungrazed differences for abundant taxa, but did show larger numbers of Orthoptera, fungus gnats (Sciaridae), and spiders (Araneae) on grazed sites. Approximately one-third of the population variables had significant block effects, the strongest of which were for Orthoptera, Sciaridae, Anthomyiidae, and Araneae (Table 4).

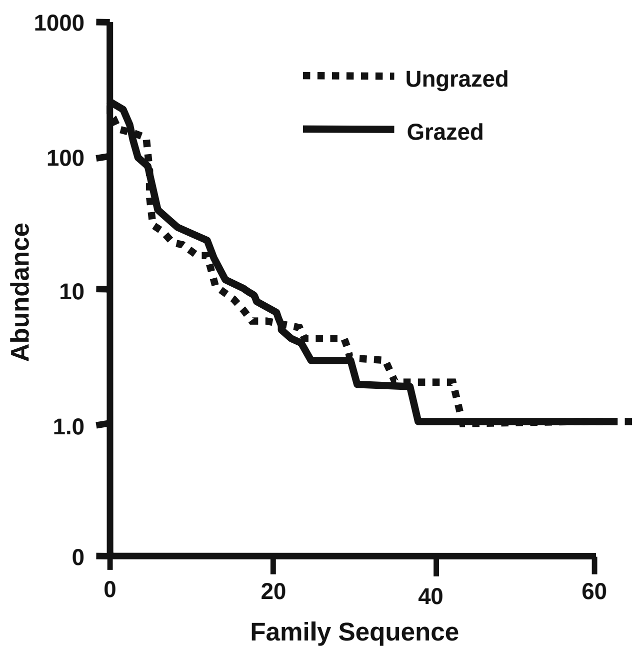

Fig. 2 Rank abundance plot for families comparing ungrazed and grazed wet meadows based on total abundances for study

Relatively few taxa and individuals were collected on the bait cards (Table 4). The ant (Formicidae) catch was dominated by *Myrmica discontinua* Weber, but we also collected small numbers of *Formica lasioides* Emery, *F. neorufibarbis* Emery, *F. aserva* Forel, *F. canadensis* Santschi, and *Camponotus vicinus* Mayr, as well as Acari (mites). Species richness was identical in grazed and ungrazed meadows, but total ant abundance was twice as high on grazed as on ungrazed sites (p=0.14, ANCOVA, Table 4). ANCOVAs also showed significant block effects for ant abundance and species richness. No significant differences were apparent via ANOVA.

Discussion

Some changes in coarse vegetation structure persisted from previous years of stock use to the start of the new grazing season, despite the annual winter respite from stock use. There was significantly less litter depth and cover on our grazed

Table 3 Means, standard errors, and results of 1×2 randomized block ANOVAs (n=20; df=1,9) and ANCOVAs with elevation as a covariate (df=1,9,1) for assemblage-level faunal metrics. All metrics are based on 50-sweep samples

	Ungrazed		Grazed		ANOVA		ANCOVA	
	Mean	SE	Mean	SE	Block	Treatment	Block	Treatment
Total individuals	67.6	15.98	66.6	14.60	0.28	0.90	0.32	0.73
Family richness	13.6	1.91	13.4	1.50	0.61	0.91	0.65	0.61
Species richness	18.3	2.87	18.2	2.20	0.55	0.83	0.63	0.78
Expected no. of families $E(F_8)$	4.8	0.33	4.8	0.15	0.26	0.62	0.25	0.58
Expected no. of species $E(S_8)$	5.3	0.39	5.5	0.24	0.12*	0.45	0.17*	0.77
% Family dominance	40.1	4.72	37.1	2.63	0.23	0.62	0.23	0.59
% Species dominance	35.8	5.48	30.1	3.66	0.26	0.39	0.34	0.85
Hurlbert's PIE (family)	0.8	0.04	0.8	0.02	0.35	0.45	0.32	0.67
Hurlbert's PIE (species)	0.8	0.05	0.8	0.02	0.27	0.36	0.28	0.73
Margalef's family diversity	3.2	0.36	3.0	0.22	0.43	0.75	0.56	0.53
Margalef's species diversity	4.3	0.56	4.3	0.36	0.41	0.86	0.53	0.78
% Predators	11.3	2.04	12.0	2.31	0.12*	0.82	0.15*	0.69

*P<0.19 (see Methods); **P<0.05.

sites, and reductions in litter have also been observed as a result of cattle and sheep grazing (King and Hutchinson 1983; Andresen et al. 1990; Bromham et al. 1999). We observed less standing, senescent (brown) vegetation at the grazed sites, and this effect was probably due to breakage and grazing of vegetation at the end of the previous year. Although bare ground was nominally more extensive on our grazed sites, this difference was not significant, in contrast with findings from past manipulations of pack stock use (Moore et al. 2000; Spildie et al. 2000; Cole et al. 2004). We also found no evidence of lasting impacts on canopy height, in contrast to several other studies of livestock effects (Andresen et al. 1990; Kruess and Tscharntke 2002; Hartley et al. 2003). This result is important, because canopy height is often a positive predictor of insect diversity and abundance (Haysom and Coulson 1998; Kruess and Tscharntke 2002). Experimental clipping (Stohlgren et al. 1989) indicates that wetter vegetation assemblages may be more susceptible to livestock impact than the *Calamagrostis* dominated assemblage.

We found relatively few negative or positive effects of long-term pack stock grazing on the arthropod assemblages in these subalpine wet meadows, but we addressed only persisting multi-year effects rather than the immediate effects that may occur at the height of stock usage. Hatfield and LeBuhn (2007) found sheep grazing to negatively affect bumble bee assemblages in the Sierra Nevada but similarly found these effects to not carry over to a subsequent year. Our one significant faunal contrast via ANOVA showed beetles to have a negative response to grazing, whereas ANCOVA showed positive effects on four taxa, including ants. Studies of livestock effects on arthropods have variously found positive, negative, mixed, or no effect across the entire assemblage (see Introduction); other efforts report differential responses among arthropod taxa. Herbivores (Andresen et al. 1990; Gibson et al. 1992) and leafhoppers in particular, have been found to be more affected by livestock than other taxa (Morris and Lakhani 1979; Morris and Rispin 1987; but see Kruess and Tscharntke 2002). Although invertebrates have been shown to be notoriously sensitive to subtle vegetation differences in many environments (e.g., Wiens et al. 1997; Holmquist 1998; McAbendroth et al. 2005), there were apparently few indirect effects on arthropods driven by litter losses in the grazed meadows. Ants might represent an exception. This group showed a significant positive relationship to grazing, albeit at a higher alpha level and only via ANCOVA after adjusting for elevation. Ants have been shown to have positive responses to livestock grazing in some other habitats (e.g., Majer and Beeston 1996; Bromham et al. 1999; Underwood and Fisher 2006), and these increases can be driven by litter losses (Bromham et al. 1999) similar to those observed in our study.

Were there really few effects on fauna? The almost complete lack of significant negative effects on fauna as tested by ANOVA across 12 assemblage and 21 population metrics, not only at alpha=0.05 but at the high alpha of 0.19 and associated high power of 0.81, provides no indication of an overall negative grazing effect on fauna. The rank abundance plots were also consistent with this conclusion. Analysis by ANCOVA also did not suggest negative effects on faunal assemblage metrics or populations, or positive effects on assemblage metrics, but did suggest positive effects in four of 21 tested faunal populations (orthopterans, fungus gnats, spiders, and ants). These positive effects may be in fact be present, and other

Table 4 Means, standard errors, frequencies, and results of 1×2 randomized block ANOVAs (n=20; df=1,9) and ANCOVAs with elevation as a covariate (df=1,9,1) on abundances of orders and the ten most abundant families in sweep samples (top) and on bait metrics (below). Sweep metrics are based on 50-sweep samples, and all bait metrics are per one aggregate hour of bait deployment using one honey and one tuna bait. Plecoptera and Psocoptera were too rare to test

	Ungrazed			Grazed			ANOVA		ANCOVA	
	Mean	SE	Frequency	Mean	SE	Frequency	Block	Treatment	Block	Treatment
Sweeps										
Orthoptera	0.45	0.29	0.40	1.20	0.76	0.40	0.04**	0.31	0.12*	0.11*
Plecoptera	0.05	0.05	0.10	0.00	0.00	0.00	NA		NA	
Hemiptera	18.70	6.16	1.00	15.55	4.12	1.00	0.11*	0.97	0.24	0.88
Cicadellidae	9.10	3.97	1.00	9.70	3.97	0.70	0.27	0.87	0.75	0.32
Delphacidae	7.25	4.83	0.80	4.40	1.31	0.90	0.68	0.77	0.52	0.20
Thysanoptera	0.20	0.11	0.30	0.10	0.07	0.20	0.81	0.54	0.90	0.83
Psocoptera	0.00	0.00	0.00	0.05	0.05	0.10	NA		NA	
Coleoptera	1.05	0.23	0.80	0.40	0.12	0.60	0.95	0.10*	0.91	0.80
Hymenoptera	2.90	0.66	0.90	3.60	1.34	1.00	0.17*	0.76	0.16*	0.39
Ichneumonidae	0.90	0.34	0.70	1.70	1.00	0.70	0.07*	0.49	0.07*	0.58
Lepidoptera	0.15	0.08	0.30	0.30	0.20	0.30	0.82	0.65	0.52	0.28
Diptera	43.20	14.38	1.00	41.95	10.20	1.00	0.42	0.93	0.37	0.42
Culicidae	1.40	0.97	0.50	1.85	1.37	0.50	0.05*	0.85	0.49	0.54
Sciaridae	1.15	0.43	0.60	1.55	1.09	0.50	0.11*	0.72	<0.01**	<0.01**
Empididae	1.50	1.28	0.50	1.15	0.49	0.50	0.22	0.64	0.21	0.29
Anthomyiidae	7.35	2.94	1.00	11.70	5.42	0.90	<0.01**	0.50	<0.01**	0.57
Muscidae	6.55	2.25	0.90	6.80	1.68	1.00	0.83	0.49	0.54	0.28
Chloropidae	6.10	1.75	0.90	10.20	3.26	1.00	0.91	0.38	0.96	0.53
Ephydridae	12.90	11.52	0.90	4.15	1.94	0.60	0.26	0.94	0.39	0.72
Araneae	0.85	0.30	0.70	1.05	0.73	0.50	0.07*	0.69	0.01**	0.03**
Acari	0.15	0.11	0.20	0.30	0.25	0.20	0.39	0.70	0.54	0.78
Baits										
Myrmica discontinua	1.55	0.93	0.60	3.40	1.81	0.30	0.61	0.75	0.30	0.20
Formicidae	1.95	0.95	0.70	4.05	1.82	0.50	0.44	0.58	0.12*	0.14*
Ant species richness	0.55	0.19		0.55	0.20		0.34	0.90	0.11*	0.81
Acari	0.25	0.13	0.30	0.80	0.64	0.30	0.37	0.55	0.46	0.22

*P<0.19 (see Methods); **P<0.05.

work has shown both orthopterans (Bock et al. 2006) and ants (Underwood and Fisher 2006) to be positively affected by grazing in some habitats. It is also possible that, given the small elevation differences between grazed and ungrazed treatments, the statistical significance of elevation may exceed the associated ecological significance. The presence of significant block effects for about one-third of the faunal population, physical, and vegetation metrics indicates that there were some differences among habitats and that there was sufficient power in the design to detect extensive treatment differences if such differences were present. Although negative effects were generally not observed at the family level, it is possible that some individual species were reduced in abundance or absent on the grazed sites.

Why were there no negative effects on fauna? Negative livestock effects on fauna have been demonstrated in a number of environments (e.g., Bestelmeyer and Wiens 1996; González-Megías et al. 2004), and Kruess and Tscharntke (2002) found insects to be more sensitive than plants to cattle grazing. Although many studies of grazing effects on arthropods have used spatial comparisons, several authors have shown arthropod population densities, biomass, species richness, and/or diversity to increase when stock pressure ceases (Andresen et al 1990 and references therein; Hatfield and LeBuhn 2007). Sequoia National Park, however, has a particularly rigorous stock management program and strives to limit pack stock impacts by controlling opening dates for individual wetlands on the basis of wet meadow condition, assessed via plant assemblage structure and phenological development, as well as estimates of soil moisture determined by the nature of the preceding winter. Although we found some stock impacts on dead vegetation structure, the low levels of stock usage maintained by Sequoia National Park (mean of 18.5 stock nights/ha/yr) were apparently below the threshold for impact to the arthropod assemblage, at least as assessed at the start of the growing season before stock arrived. Cole et al. (2004) note that meadow vegetation can be maintained in good condition with low levels of stock use, but even moderate use often results in impacts. Park Service regulation of meadow opening dates and stock densities, arthropod dispersal capabilities (Hatfield and LeBuhn 2007), concentration of grazing in *Calamagrostis* dominated meadows, short grazing seasons, long winter recovery periods, and our pre-grazing sampling likely combined to limit impacts and/or our detection thereof. There might be different results in Sierra meadows with wetter conditions (e.g., Stohlgren et al. 1989), a longer grazing season, or less regulation. As an example of one impact pathway that was absent in our wetlands, Andresen et al. (1990) found cattle to reduce canopy height with an associated loss of canopy arthropods. In our Sequoia wet meadows, stock were excluded from grazing areas in the spring and a full canopy developed. We sampled before stock arrived and thus before the canopy could potentially be newly degraded. Our results indicate little long-term stock damage to the arthropod assemblage, and this finding is encouraging, but our results do not address potential impacts at mid-season.

If there were no lasting negative effects on fauna, does that mean that any mid-season impacts are inconsequential? No. The many studies recording livestock impacts to epigeal arthropods report results obtained during or immediately after grazing (Kruess and Tscharntke 2002; see Introduction). Our limited mid-season pilot sampling also suggests that pack stock may reduce arthropod diversity and abundance in these subalpine wet meadows. Removal of canopy in the middle of the growing season is more deleterious to the arthropod assemblage than removal during early season (Duffey et al. 1974), and Baines et al. (1998) showed that one mid-season canopy removal had greater negative effect on spider species richness and abundance than two removals in spring and fall. Univoltine species can be affected by mid-season canopy removal more than multivoltine species (Morris 1979), but multivoltine taxa could also have brood size reduced or eliminated during peak stock use periods. Flowering in these subalpine wetlands occurs during stock usage, and removal of flowers can negatively affect butterflies (Feber et al. 1996), and other nectivores (Vickery et al. 2001; Hatfield and LeBuhn 2007), which in turn may reduce pollinator availability to plants. These subalpine wetlands may be “reset” over the long winter and spring, but it is possible that pack stock reduce mid-season productivity and diversity of these wetlands, and such losses could cascade into vertebrate (Vickery et al. 2001) and/or upland assemblages. Although it appears that pack stock do not cause lasting damage to the wet meadow arthropod assemblage, the question as to impacts at the height of the grazing season remains unanswered, and we will not have a full understanding of the role of pack stock in these wetlands until this issue is addressed.

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Chapter 14

Responses of Salmonids to Habitat Changes

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Streams in western North America provide spawning and rearing habitats for several species of salmon and trout that are of substantial economic importance in the region. Timber that grows on lands through which these streams flow is also economically important, and its harvest can substantially change habitat conditions and aquatic production in salmonid streams. Undisturbed forests, the streams that flow through them, and the salmonid communities in these streams have intrinsic scientific, genetic, and cultural values in addition to their economic importance. The complex relations between salmonids and their physical environment, and the changes in these relations brought about by timber harvest, have been investigated extensively (see the bibliography by Macdonald et al. 1988). However, in spite of considerable evidence of profound changes in channel morphology and in light, temperature, and flow regimes associated with timber harvests, much uncertainty exists about the responses of salmonids to these changes.

Responses of salmonid populations to changes in their freshwater environment brought about by forest-management activities are similar in many respects to those caused by other land-management activities such as mining (Nelson et al. 1991, this volume) and livestock grazing (Plates 1991, this volume). For example, fine sediment potentially has the same biological effect in streams whether it results from mining effluent, streambank erosion in grazed pasture lands, or road construction and clear-cut logging (Lloyd et al. 1987). The severity of its effect, however, depends on particle size and concentration, which in turn are related to sediment source. Changes in stream productivity caused by increased nutrient loading can similarly result from mining, livestock grazing, or forest management. Removal of riparian vegetation by any management activity changes the light and temperature regimes of a stream, leading to changes in primary and secondary production, in emergence times of salmonid fry, and in summer and winter survival of juvenile salmonids. There are also similarities between effects of natural catastrophic events, such as volcanic eruptions, and those of timber harvest (Sedell and Dahm 1984; Martin et al. 1986; Bisson et al. 1988). Despite substantial research by federal, state, and provincial agencies, universities, and the timber industry (Salo and Cundy 1987), effects of forest practices on salmonids are still not well known. Less information is available on effects of livestock grazing and mining on salmonids (Rinne 1988; Nelson et al. 1991). We

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have thus chosen to focus our evaluation on the effects of timber harvest. Many of our conclusions, however, should be applicable to habitat changes caused by any management activity.

We consider the effects of logging on salmonids and their freshwater habitats in several ways. We first describe the effects of logging and other aspects of forest management on isolated parts of salmonid life cycles. Then we discuss integrated effects of forest practices on stream-dwelling salmonid populations. We also attempt to elucidate patterns of regional variation in the response of fish populations and to evaluate management practices that best protect fish habitat.

Responses to Specific Components of Habitat Change

In this section we examine the evidence that specific types of environmental change associated with logging practices have influenced short-term survival and growth of salmon and trout. Categories of environmental change commonly attributed to timber harvest activities, and their generalized consequences for salmonids, are shown in Table 14.1. The data supporting the generalizations in Table 14.1 were drawn from a variety of studies; some studies were process-specific and some examined the integrated effects of logging within a drainage. Furthermore, some of the studies were short-term comparisons of logged and unlogged basins whereas others were longer-term comparisons of a particular basin before and after logging. Duration of postlogging assessments has varied from 1 year to over a decade. For some types of habitat change, short-term studies have been sufficient to determine the responses of salmonid populations. For other types of change (e.g., the rate of recruitment of large woody debris), short-term studies have been inadequate to evaluate population response, and multisite comparisons of specific habitat change have been weakened by confounding factors. We considered these limitations as we reviewed each of the possible types of habitat change associated with timber harvest.

TABLE 14.1.—Influences of timber harvest on physical characteristics of stream environments, potential changes in habitat quality, and resultant consequences for salmonid growth and survival.

Forest practice	Potential change in physical stream environment	Potential change in quality of salmonid habitat	Potential consequences for salmonid growth and survival
Timber harvest from streamside areas	Increased incident solar radiation	Increased stream temperature; higher light levels; increased autotrophic production	Reduced growth efficiency; increased susceptibility to disease; increased food production; changes in growth rate and age at smolting
	Decreased supply of large woody debris	Reduced cover; loss of pool habitat; reduced protection from peak flows; reduced storage of gravel and organic matter; loss of hydraulic complexity	Increased vulnerability to predation; lower winter survival; reduced carrying capacity; less spawning gravel; reduced food production; loss of species diversity

TABLE 14.1.—Continued.

Forest practice	Potential change in physical stream environment	Potential change in quality of salmonid habitat	Potential consequences for salmonid growth and survival
	Addition of logging slash (needles, bark, branches)	Short-term increase in dissolved oxygen demand; increased amount of fine particulate organic matter; increased cover	Reduced spawning success; short-term increase in food production; increased survival of juveniles
	Erosion of streambanks	Loss of cover along edge of channel; increased stream width, reduced depth	Increased vulnerability to predation; increased carrying capacity for age-0 fish, but reduced carrying capacity for age-1 and older fish
		Increased fine sediment in spawning gravels and food production areas	Reduced spawning success; reduced food supply
Timber harvest from hillslopes; forest roads	Altered streamflow regime	Short-term increase in streamflows during summer	Short-term increase in survival
		Increased severity of some peak flow events	Embryo mortality caused by bed-load movement
	Accelerated surface erosion and mass wasting	Increased fine sediment in stream gravels	Reduced spawning success; reduced food abundance; loss of winter hiding space
		Increased supply of coarse sediment	Increased or decreased rearing capacity
		Increased frequency of debris torrents; loss of instream cover in the torrent track; improved cover in some debris jams	Blockage to migrations; reduced survival in the torrent track; improved winter habitat in some torrent deposits
	Increased nutrient runoff	Elevated nutrient levels in streams	Increased food production
	Increased number of road crossings	Physical obstructions in stream channel; input of fine sediment from road surfaces	Restriction of upstream movement; reduced feeding efficiency
Scarification and slash burning (preparation of soil for reforestation)	Increased nutrient runoff	Short-term elevation of nutrient levels in streams	Temporary increase in food production
	Inputs of fine inorganic and organic matter	Increased fine sediment in spawning gravels and food production areas; short-term increase in dissolved oxygen demand	Reduced spawning success

Changes in Stream Temperature and Light Regime

Changes in stream temperature and light regime after logging can have both positive and negative consequences for salmonid production. As a result, the effects of changes in temperature and light on salmonids are difficult to predict. Removal of streamside vegetation allows more solar radiation to reach the stream surface, increasing water temperature and light available for photosynthesis (Brown and Krygier 1970). In small streams (first to third order), increased daily temperature fluctuations also result from opening the vegetative canopy (Meehan 1970; Beschta et al. 1987; Bisson et al. 1988). The interpretation of much of the early research on temperature changes induced by logging was that these alterations were predominantly harmful to salmonids (Lantz 1971). Recent evidence has suggested that, under certain conditions, temperature and light increases can be beneficial.

Beschta et al. (1987) summarized studies of stream temperature changes associated with canopy removal over small streams in Pacific coastal drainages. They found a latitudinal gradient in increase of daily maximum temperature in summer that ranged from only a few degrees Celsius in Alaska to more than 10°C in Oregon. In the first 5 years following the 1980 eruption of Mount St. Helens, Washington, which virtually devegetated the watersheds of many Toutle River tributaries, Martin et al. (1986) and Bisson et al. (1988) measured peak midsummer stream temperatures of 29.5°C and diel variations as great as 17.0°C.

Although large changes in thermal and light regimes after timber harvest have been observed during summer, winter temperatures do not appear to be strongly affected by canopy removal, although even minor changes can have important consequences for streams when water temperatures are low. In general, forest canopy removal results in increased winter temperatures in low-elevation coastal drainages (Beschta et al. 1987). In northern latitudes and at higher elevations, a reduction in winter stream temperatures may occur due to loss of insulation from the surrounding forest coupled with increased radiative cooling of the stream. Where winter temperatures are lowered, ice forms more rapidly and the possibility of winter freeze-up increases. Such conditions may reduce habitat quality if anchor ice formation is extensive or if ice jams occur and then release ice flows to scour the streambed (Needham and Jones 1959). In contrast, slight postlogging increases in late-winter water temperature were found in Carnation Creek, a coastal stream on Vancouver Island, British Columbia (Hartman et al. 1987; Holtby 1988b). These temperature increases led to accelerated development of coho salmon embryos in the gravel and earlier emergence of juveniles in the spring. Earlier emergence resulted in a prolonged growing season for the young salmon, but increased the risk of their downstream displacement during late-winter freshets. Many changes in salmonid growth and survival resulted from the stream temperature increases in this watershed (Figure 14.1).

Among the potential benefits of elevated light and temperature during summer are increased primary and secondary production, which may lead to greater availability of food for fish. Solar radiation is an important factor limiting algal growth in forested streams (Hansmann and Phinney 1973; Stockner and Shortreed 1978; Gregory 1980; Gregory et al. 1987). Shifts in periphyton composition in response to elevated temperatures have been observed in laboratory streams

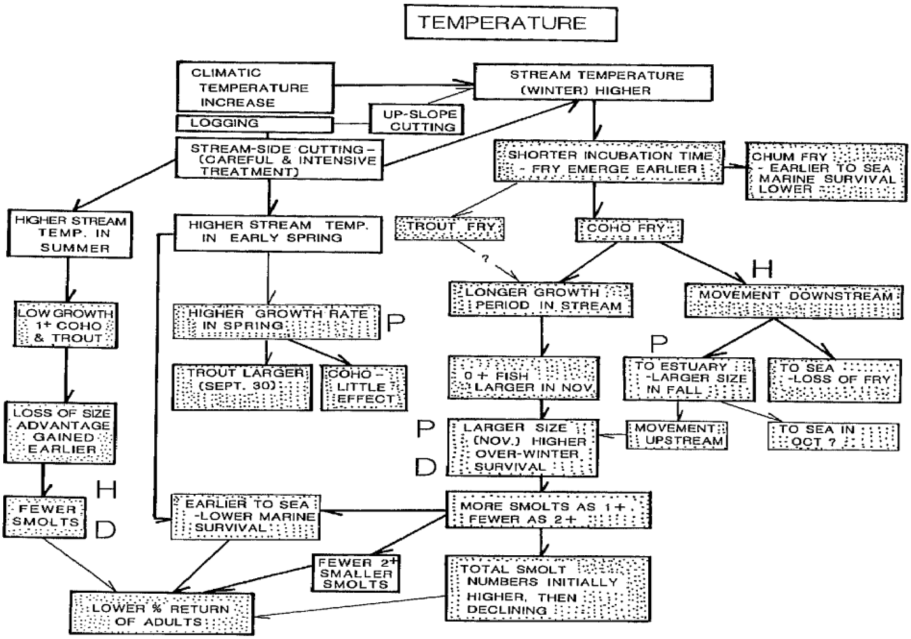

FIGURE 14.1.—Effects of logging on temperature-dependent processes at Carnation Creek, British Columbia, and the effects of temperature changes on fish populations. (From Hartman 1988.) The letter D, Fl, or P indicates the existence of an interaction with woody debris, stream hydrology, or production-related processes. Firmly established conclusions and cause-and-effect relations are indicated by heavy lines. Effects on fish are indicated by stippling. (Phinney and McIntire 1965; Bisson and Davis 1976). Typically, well-lit warm streams are dominated by filamentous green algae instead of the epilithic diatoms that predominate in heavily shaded streams. After logging, increases in filamentous green algae promote the abundance of grazer invertebrates as the stream shifts from allochthonous to autochthonous carbon sources. Streams in clear-cut drainages often produce more baetid mayflies, grazing caddisflies, and orthocladid midges than do streams in forested watersheds (Hawkins et al. 1982). These groups are often more likely to enter the drift and thus become potential food items for salmonids (Gregory et al. 1987). Wilzbach et al. (1986) found that cutthroat trout captured drifting prey more efficiently in open areas than in

adjacent old-growth forest, where light intensities were reduced by canopy shading. In some streams, increased food availability can mitigate those detrimental habitat changes associated with removal of riparian vegetation, including high summer temperatures (Bisson et al. 1988). The importance of autochthonous production to salmonid populations in clear-cut watersheds of the western USA has been suggested by several studies (Murphy and Hall 1981; Hawkins et al. 1983; Bisson and Sedell 1984; Bilby and Sisson 1987). In a whole-river enrichment study on Vancouver Island, British Columbia, stimulation of algal growth by inorganic nutrient addition resulted in greater benefits to salmonids than did stimulation of heterotrophic production through addition of cereal grains (Slaney et al. 1986; Perrin et al. 1987). Bilby and Bisson (1989) also found that summer production of juvenile coho salmon was more directly related to the amount of organic matter produced by autochthonous sources (algal photosynthesis) than to organic matter produced by allochthonous carbon sources (terrestrial litter inputs and fluvial transport), regardless of whether the stream flowed through a clear-cut or an old-growth forest. Potentially negative effects on salmonids of logging-related changes in summer temperature were reviewed by Beschta et al. (1987). These include temperature elevation beyond the range preferred for rearing, inhibition of upstream migration of adults, increased susceptibility to disease, reduced metabolic efficiency with which salmonids convert food intake to growth, and shifts of the competitive advantage of salmonid over nonsalmonid species. Of these five categories of effects, the last is least understood. Reeves et al. (1987) found that stream temperature influenced the outcome of competitive interactions between redside shiners and juvenile steelhead. The shiners were more active and competitively dominant at warm temperatures whereas steelhead were dominant at cool temperatures. The authors hypothesized that a long-term increase in stream temperature could allow redside shiners to displace steelhead from reaches where steelhead formerly enjoyed a competitive advantage. From the perspective of basin-wide temperature, the most important role of tributaries may be to provide cool water downstream. Positive effects of canopy removal in tributaries may be more than offset by reduced cooling influence in main-stem habitats.

Decreased Supply of Large Woody Debris

Among the most important long-term effects of forest management on fish habitat in western North America have been changes in the distribution and abundance of large woody debris in streams. These changes have extended from small headwater streams to the estuaries of major rivers (Sedell and Luchessa 1982; Maser et al. 1988). Overall trends have included reduction in the frequency of pieces of large, stable debris in streams of all sizes; concentration of debris in large but infrequent accumulations; and loss of important sources of new woody material for stream channels (Bisson et al. 1987). As noted elsewhere in this volume (see Reeves et al. and Sedell et al.), these trends have been accelerated by stream channelization and by debris removal for navigation (Sedell and Luchessa 1982), for upstream fish migration (Narver 1971), and for reduction of property damage during floods (Rothacher and Glazebrook 1968). Large woody debris plays an important role in controlling stream channel morphology (Keller and Swanson 1979; Lisle 1986b; Sullivan et al. 1987), in

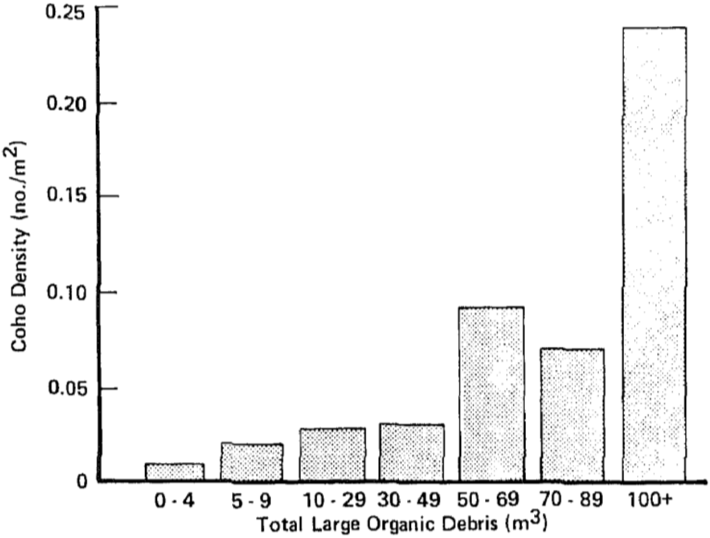

FIGURE 14.2,—Relationship between abundance of woody debris and density of juvenile coho salmon during winter in southeastern Alaska. (From Murphy et al. 1984a.)

regulating the storage and routing of sediment and particulate organic matter (Swanson et al. 1976; Swanson and Lienkaemper 1978; Naiman and Sedell 1979; Bilby 1981; Megahan 1982; Bilby and Ward 1989), and in creating and maintaining fish habitat (Bryant 1983; Lisle 1986a; Murphy et al. 1986; Bisson et al. 1987). The abundance of salmonids is often closely linked to the abundance of woody debris (Figure 14.2), particularly during winter (Bustard and Narver 19756; Tschaplinski and Hartman 1983; Murphy et al. 1986; Hartman and Brown 1987). Large woody debris creates a diversity of hydraulic gradients that increases microhabitat complexity (Forward 1984), which in turn supports the coexistence of multispecies salmonid communities (Figure 14.3). Prior to the development of extensive road networks, large streams and rivers were used to float logs downstream to mills. Widespread reaches of rivers were cleared of snags to permit navigation and to prevent logs from accumulating in large jams during transport (Sedell and Luchessa 1982; Sedell et al. 1991, this volume). Channel clearance accompanied timber harvest in the lower portions of river basins. As logging progressed upstream and the streams became too small to transport logs effectively, splash dams were built to store logs and water until sufficient head had accumulated to *allow* the logs to be sluiced *downstream to a* larger river. Splash dams were numerous in coastal watersheds (Wendler and Deschamps 1955; Sedell and Luchessa 1982). The long-term effect of channel clearance and splash damming was to remove vast quantities of woody debris from medium- and large-sized streams, a condition from which many rivers have apparently not recovered after 50 to 100 years (Sedell et al. 1991). Little is known *about the effects of snag removal* and splash *damming on saimonid populations,* partly because these practices are not frequently used today. It is likely, however,

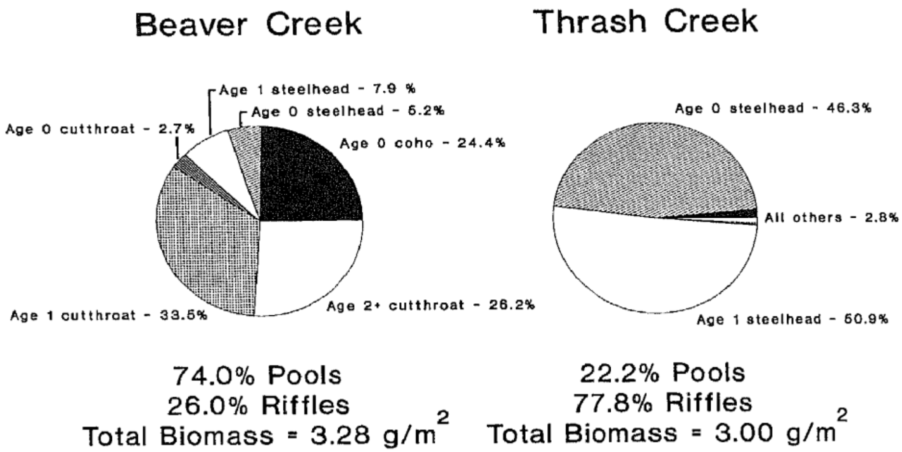

FIGURE 14.3.—Comparison of salmonid communities in two Washington streams with similar fish biomasses: Beaver Creek, a debris-rich stream in the Nisqually River system, and Thrash Creek, a debris-poor stream in the Chehalis River system. (P. A. Bisson, Weyerhaeuser Corporation, unpublished; reproduced in part from Sullivan et al. 1987.)

that species making extensive use of larger stream channels (e.g., chinook and chum salmon) were greatly affected. Beginning in the early 1970s, forest practice rules in some western states required removal of slash (limbs and tops) from streams immediately after timber harvest. In many cases, stream cleaning also removed large pieces of merchantable prelogging debris from the channel. Recent case studies have assessed the immediate effects of debris removal on salmonid populations. Almost without exception, removal of large woody debris has resulted in loss of important habitat features and a decline in salmonid population abundance (Bryant 1980; Toews and Moore 1982a; Lestelle and Cederholm 1984; Dolloff 1986; Elliott 1986). Debris removal caused a decline in channel stability and a corresponding reduction in the quality and quantity of pools and cover. Bisson and Sedell (1984) observed enlarged riffles and a reduction in the number of pools in cleaned stream channels in Washington. The increased frequency of riffles favored underyearling steelhead and cutthroat trout, which preferred riffle habitat, but caused a decrease in the relative proportions of coho salmon and older age classes of steelhead and cutthroat trout, which preferred pools. Removal of nearly all large trees from riparian zones during logging has also caused a long-term reduction in the recruitment of new large woody debris to stream channels, leading to a reduction in the quality of fish habitat. There may be a short-term increase in the debris load caused by entry of slash during harvesting and yarding, but this small unstable debris is often floated downstream within a few years, to be trapped in a few widely spaced debris jams (Bryant 1980). If the debris load of a channel is not replenished by large-scale inputs such as extensive blowdowns or debris avalanches, the second-growth riparian zone becomes the principal source of new woody debris. In young forest stands, inputs of debris large enough to be stable in streams with channel widths greater than about 15 m remain low for at least the first 60 years of riparian forest regrowth (Grette 1985; Long 1987). Many streams in second-growth forests have become progressively debris-impoverished following logging to the edge of the channel. Young riparian stands do not produce sufficient debris of the proper size and quality to replace material lost when channels are cleaned and large preharvest debris gradually decays (Sedell et al. 1984; Andrus et al. 1988). The effect on fish habitat has been a decrease in channel complexity, in number and volume of pools, in quality of cover, and in capacity of streams to store and process organic matter.

Some debris jams formed after logging or after a logging-related debris flow have blocked upstream fish migration. The improper design, construction, or maintenance of culverts on forest roads can also prevent or hinder upstream movements (Toews and Brownlee 1981; Furniss et al. 1991, this volume). To date, the relative importance of natural barriers, logging-related debris jams, and impassable culverts in limiting upstream movements of salmonids has not been assessed in any basin-wide survey. With the exception of dams on large rivers, we have little information on the extent to which fish-passage problems occurring in an entire drainage system have limited salmonid production.

Reduced Dissolved Oxygen Concentration

The introduction of fine logging slash, leaves, and needles into streams as a result of timber harvest can increase biochemical oxygen demand at critical times of low flows and high temperatures. In one small coastal Oregon drainage that was logged to the stream edge, dissolved oxygen concentrations of surface water decreased following timber harvest to below levels acceptable for salmonid survival and growth (Hall and Lantz 1969; Moring 1975a). Accumulation of fine organic matter from logging debris, in combination with increased stream temperature, was responsible for the decrease in dissolved oxygen. However, apart from short-lived effects in small streams in areas that naturally experience high summer insulation, there is no evidence of a major effect of logging on salmonids from low dissolved oxygen concentrations in surface water. Of more importance than the usually transient decreases in dissolved oxygen in surface water are postlogging reductions in intragravel oxygen levels. These reductions may be caused by increased oxygen demand from fine organic matter introduced into the gravel matrix or by reduced interchange of surface and intragravel water. Several field studies have demonstrated reductions in oxygen concentration in redds following logging (e.g., Ringler and Hall 1975). However, increased intragravel sediment, decreased permeability, and decreased velocity of intragravel water often coincide with reduced levels of dissolved oxygen. Hence, the importance of reduced oxygen levels as a primary cause of mortality of embryos and alevins in gravel is difficult to assess. It is rare for oxygen deprivation to directly kill many embryos and alevins in stream gravels after logging, but emergent juveniles that have incubated in oxygen-poor gravels may have reduced viability, and this may be an important factor in the regulation of salmonid populations. In an extensive review of the effects of intragravel oxygen on salmonid embryos and alevins, Chapman (1988) concluded that any reduction in dissolved oxygen below saturation may cause salmonids to be smaller than normal at emergence. Smaller juveniles are at a competitive disadvantage and likely have reduced fitness within the population (Mason 1969).

Altered Streamflow

The pattern of a stream's discharge affects the water depths and velocities that stream-dwelling salmonids will encounter the amount of habitat available at different times of the year, and, to some extent, stream temperatures. Changes in streamflow caused by timber harvest are discussed in detail by Chamberlin et al. (1991, this volume). The effect of timber harvest on streamflow varies with logging method, soil permeability, season, and climate. Vegetation influences streamflow by intercepting precipitation and transpiring water (Bosch and Hewlett 1982). Removal of timber reduces transpiration and interception, and summer low flows and early fall peak flows usually increase as a result (Harr et al. 1975; Harr 1983). Of particular concern in western North America are increased peak flows that result from rainfall on snow following clear-cut logging in the transient snow zone (Harr 1986). Streamflow does not always increase following logging. The presence of a well-developed coniferous forest can result in greater streamflows than occur after timber harvest. In coastal watersheds or at high elevations, fog drip can augment streamflow. This effect, caused by interception of wind-blown fog by tall trees, is most influential in summer (Harr 1982). In addition, increases in streamflow that result from timber harvest are generally short-lived within a timber harvest rotation; streamflow returns toward preharvest conditions as vegetation regrows (Harr 1983). Summer flows 10 years or more after logging may actually be lower than preharvest summer flows (Hicks et al., in press). The effects on salmonids of changes in streamflow have not been documented separately from other effects of logging. To what extent the ameliorating effect of short-term increases in summer flows may be counterbalanced by increased severity of floods in other seasons is not known. In large basins, timber harvest activities are usually dispersed in space and over time. The result is that logging in large basins causes proportionately smaller changes in streamflow than it does in small basins (Duncan 1986).

Increased Fine Sediment

Fine sediment can enter streams during and after timber harvest as a result of road construction, timber harvest, and yarding activities (Everest et al. 1987a). Mass soil movements following road construction and timber harvest produce fine sediment, but the amount of fine sediment washing from the unpaved surfaces of actively used logging roads can equal that produced by landslides (Reid and Dunne 1984). Fine sediment that settles in streams or moves in suspension can reduce salmonid viability. Determination of the effects that deposited fine sediments have on salmonids is complicated by the variability in responses among salmonid species and by the adaptability of salmonids to ambient sediment levels (Everest et al. 1987a). Fine sediment deposited in spawning gravel can reduce interstitial water flow, leading to depressed dissolved oxygen concentrations, and can physically trap emerging fry in the gravel (Koski 1966; Meehan and Swanston 1977; Everest et al. 1987a). Survival of coho salmon in natural and simulated redds is related to the proportion of fine particles in the gravel (Figure 14.4).

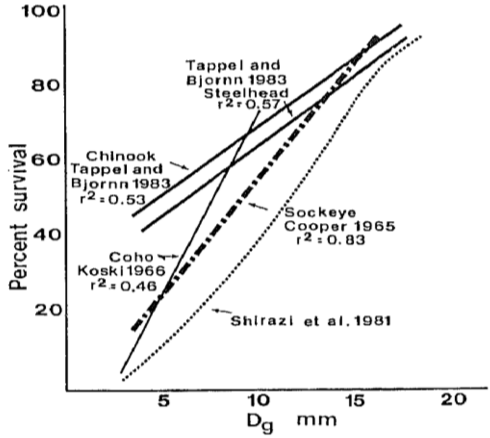

FIGURE 14.4.—Survival to emergence in relation to geometric mean particle size (D_g) for natural coho salmon redds (Koski 1966), and for other salmonids in laboratory gravels (Cooper 1965; Tappel and Bjornn 1983). The composite model of Shirazi et al. (1981) is plotted in the same D_g range (3-17 mm). (Entire figure from Chapman 1988.)

Salmonid survival apparently was affected when timber harvest increased the amount of fine sediment in spawning gravels of some Alaska streams (Smedley 1968; Smedley et al. 1970); in some cases, however, the amount of sediment in gravels returned to prelogging conditions within 5 years (McNeil and Ahnell 1964; Sheridan and McNeil 1968; Sheridan et al. 1984). Studies in coastal drainages of the Olympic Peninsula, Washington, have addressed the effects on spawning success of sediment from landslides and logging road surfaces (Cederholm and Salo 1979; Cederholm et al. 1981; Cederholm et al. 1982; Cederholm and Reid 1987). The concentration of intragravel fine sediment in spawning riffles was positively correlated with mass soil movements and the extent of roading within watersheds. The effects of sedimentation on salmonid spawning success were estimated by monitoring the survival of embryos and alevins in natural redds (Tagart 1976) and in experimental redds containing a mixture of gravel meant to simulate conditions in Clearwater River tributaries. Based on these studies, Cederholm et al. (1981) believed that accelerated erosion caused by landslides and logging roads led to a significant reduction in emergence of coho salmon fry. Coupled with high rates of ocean harvest and resultant low spawner densities, the reduced survival to emergence was thought to have resulted in lower smolt yields from the Clearwater River basin. Studies in British Columbia (Scrivener and Brownlee 1989), Oregon (Hall et al. 1987), and Idaho (Stowell et al. 1983) have also shown declines in survival to emergence or in salmonid abundance associated with sediment increases after logging. Some studies showed temporary increases in sedimentation related to logging, although subsequent effects on salmonids have been difficult to document. Female salmon clean gravel as they construct redds, which may be one reason that effects of fine sediment in gravel are often less than predicted (Everest et al. 1987a). Chapman (1988) reviewed the laboratory evidence for the effect of fine sediment on salmonid reproduction. He found that laboratory studies had generally failed to reproduce conditions in the egg pocket of the redd and therefore did not yield accurate predictions of survival in natural streams. He concluded, however, that increases of fine sediment did reduce survival to emergence.

In addition to directly affecting salmonid survival, fine sediment in deposits or in suspension can reduce primary production and invertebrate abundance and thus can affect the availability of food within a stream (Cordone and Kelley 1961; Lloyd et al. 1987). In northern California, diversity of invertebrates was lower in streams passing through clear-cut areas with no buffers or only narrow buffers than it was in streams in unlogged watersheds. However, the densities of invertebrates were higher in the clear-cut areas or not significantly different from those in unlogged watersheds (Newbold et al. 1980; Erman and Mahoney 1983). The effect on fish production of this change in invertebrate community structure was not investigated. The detrimental effects of large amounts of fine sediment are generally accepted, but precise thresholds of fine sediment concentrations that result in damage to benthic invertebrates are difficult to establish (Chapman and McLeod 1987). Increases in suspended sediment can affect salmonids in several ways. Suspended sediment can alter behavior and feeding efficiency. Fish may avoid high concentrations of suspended sediment (Bisson and Bilby 1982; Sigler et al. 1984); at lower concentrations, fish may cease feeding (Noggle 1978; Sigler et al. 1984) and their social behavior may be disrupted (Berg and Northcote 1985). In nearly all cases, suspended sediment concentrations resulting from timber harvest are not sufficient to cause significant abrasion of the skin or gills of salmonids (Everest et al. 1987a; Bjornn and Reiser 1991, this volume), although temporary spikes of suspended sediment from landslides may approach lethal thresholds.

increased Coarse Sediment

Channel morphology changes when timber harvesting increases the rate at which coarse sediment is delivered to streams. Increased frequencies of landslides and other mass wasting events can cause channels to aggrade where the gradient and other aspects of valley topography permit gravel deposition. Stream reaches that are aggraded with coarse sediments typically become wider, shallower, and more prone to lateral movement and bank erosion (Sullivan et al. 1987). More water passes through deposited gravels, reducing surface flow. Total riffle area increases but pool area decreases, and other types of habitat may be lost (Everest et al. 1987a). Habitat alteration caused by debris torrents has been well documented in the Queen Charlotte Islands, British Columbia. In stream sections affected by debris torrents, average pool depth was reduced by 20-24%, and pool area was reduced by 38-45%. Cover associated with large woody debris was reduced by 57%, and undercut bank cover was reduced by 76%. Riffle area increased by 47-57%, and stream channel width increased by 48-77% (Tripp and Poulin 1986b). The amount of landsliding was directly related to the proportion of the basin area logged (Tripp and Poulin 1986b). The effect of logging was to increase landsliding frequency by 34 times in this geologically unstable terrain (Rood 1984). The frequency of debris torrents increased by about 40 times in logged areas compared to unlogged areas, and increased by 76 times in roaded areas compared to unlogged areas without roads (Rood 1984). A combination of logging and landsliding reduced the salmonid cover attributable to large woody debris by 58%. Reaches with landsliding had reduced width:depth ratios and less surface flow (Tripp and Poulin 1986b). In erosion-prone areas of the Oregon Coast Range, logging roads caused rates of debris avalanche erosion that were 26-350 times greater than in adjacent forested areas. Rates were 2.2-22 times greater in clear-cut areas than in unharvested forest. When these rates were adjusted for the smaller area that was occupied by roads compared to that in clear-cuts, forest roads accounted for 77% of the total accelerated debris avalanche erosion (Swanson et al. 1981). In one stream in this area with little cover from boulders or large woody debris, debris torrents added new structure to the channel, temporarily enhancing the abundance of coho salmon (Everest and Meehan 1981a).

Altered Nutrient Supply

Stream concentrations of plant nutrients and other dissolved ions increase for a few years following logging (Brown et al. 1973; Scrivener 1982, 1988b). When accompanied by increased light, these nutrient additions often result in increased algal growth (Gregory et al. 1987). However, the effects of nutrient increases on invertebrate and salmonid populations have not been thoroughly studied. Bisson et al. (1976) studied the effects of adding nitrate-nitrogen to model stream channels in southwestern Washington. They observed a temporary increase in the biomass of benthic invertebrates and in the production of stocked rainbow trout. After 2 years of continuous nutrient additions, however, no significant differences were found in either invertebrate biomass or trout production between enriched and unenriched streams. Slaney et al. (1986) added both nitrogen and phosphorus to the Keogh River on the east coast of Vancouver Island, British Columbia, after which steelhead grew faster and smolted when younger but larger. Steelhead smolt output also increased to as much as twice the preenrichment level (P. A. Slaney, Fisheries Branch, British Columbia Ministry of the Environment, personal communication), but it is not yet clear if the size of the returning adult run has increased. Studies indicate that nutrient increases (mostly nitrate) are limited to the first decade after logging; that primary production is stimulated in the presence of increased light and nutrient concentrations; that watersheds dominated by volcanic rock are more likely to show enhanced autotrophic production after logging than watersheds dominated by sedimentary or metamorphic rock; that herbivorous invertebrates will most likely benefit from increased algal growth; and that salmonid production may or may not be enhanced during periods of increased nutrient concentration (Gregory et al. 1987).

Responses of Salmonid Populations to Forest Management

Thus far we have reviewed evidence that the growth or survival of salmonids at various life stages changes in response to changes in individual components of the habitat. It is also important to know the combined effect on salmonid populations of all the changes that occur in the course of normal forest management. In this section, we review results of several long- and short-term studies designed to measure response of salmonid populations to timber harvest. We have included

TABLE 14.2.—Physical characteristics and salmonid species at sites where combined effects of timber harvest on salmonid population abundance have been measured.

Location[a]	Latitude (°N)	Watershed area (km^2)		Discharge (m^3/s)	
		Mean	Range	Minimum	Maximum
(1) Southeastern AK	56–57			0.01–0.38	
(2) Queen Charlotte Islands, BC	52–53	8.2	0.5–47.5		
(3) Carnation Creek, Vancouver Island, BC	49	10		0.03	63
(4) Coast and Cascade ranges, WA	46–48	8.3	1–30		
(5) Coast Range, OR	45		0.8–3.0	0.0006–0.0085	1.4–5.7
(6) Coast Range, OR	44–45		5.1–23.7		
(7) Mack Creek, Cascade Range, OR	44	8.3		0.05–0.45	8–11
(8) Cascade Range, OR	44		0.3–17.9		
(9) Cascade Range, OR	44		4.0–8.2	0.02–0.09	
(10) Cascade Range, OR	44		5.4–6.8		
(11) Northern Coast Range, CA	39–42		4.3–25.1	0.002–0.014	0.26–1.42
(12) Northern Coast Range, CA	41		7.3–8.1		

[a]AK = Alaska; BC = British Columbia; CA = California; OR = Oregon; WA = Washington.

[b]1 = chum salmon; 2 = coho salmon; 3 = cutthroat trout (resident or anadromous); 4 = rainbow trout or steelhead; 5 = Dolly Varden.

[c]1 = Aho (1976); 2 = Bisson and Sedell (1984); 3 = Burns (1972); 4 = Chamberlin (1988); 5 = Hall et al. (1987); 6 = Hartman (1982); 7 = Hartman and Scrivener (1990); 8 = Hawkins et al. (1983); 9 = Heifetz et al. (1986); 10 = Johnson et al. (1986); 11 = Koski et al. (1984); 12 = Moring (1975a); 13 = Moring (1975b); 14 = Moring and Lantz (1975); 15 = Murphy and Hall (1981); 16 = Murphy et al. (1981); 17 = Murphy et al. (1986); 18 = Poulin (in press); 19 = Tripp and Poulin (1986a); 20 = Tripp and Poulin (1986b); 21 = Tripp and Poulin (in press).

[d]Species not identified by site.

studies in which quantitative estimates have been made of the abundance of salmonids and their habitats. In conjunction with our evaluation, we recommend management practices that protect fish populations and their habitats from harmful effects. The studies that meet our criteria, their locations, and descriptions of the watersheds involved are listed in Table 14.2. We excluded studies limited to specific parts of salmonid life history, such as studies of embryo survival in gravel infiltrated by fine sediment.

Role of Streamside Management Zones

One of the major developments in forest management in relation to fisheries over the past 25 years has been increased protection of streamside vegetation

TABLE 14.2.—Extended.

Location[a]	Channel gradient (%)	Surface water temperature (°C)		Species[b]	Reference[c]
		Winter	Summer		
(1)	0.1–3.0	1.2–4.8	13–17	2,3,4,5	9,10,11,17
(2)	0.9–10.0 (mean, 4.2)			2,4,5	18,19,20,21
(3)	0.2 (lower section)	1	18	1,2,3,4	4,6,7
(4)	1–8			2,3,4	2
(5)	1.4–2.5	5–12.2	14.4–29.5	2,3,4	5,12,13,14
(6)	0.3–2.0		22	2,3,4[d]	8
(7)	10		14.4–17.0 (at high elevation)	3	1
(8)	1–13			3	15
(9)	1–10		<21	3,4	16
(10)	1–10		15.5–18.5	3,4[d]	8
(11)	3–5		13.9–25.3	2,3,4	3
(12)	8.0–18.0		22	2,3,4[d]	8

during timber harvest. The gradual development of streamside management concepts has come about partly because of the studies we review and partly because more has been learned about the ecological structure and function of riparian zones (Swanson et al. I 982a; Gregory et al. in press). The importance of streamside management as a tool to protect fishery values has been demonstrated in several studies that have compared fish habitat and salmonid populations in streams that were and were not given riparian protection during timber harvests. The evidence shows that streamside management zones minimize damage to habitat and effectively maintain the integrity of fish populations. This evidence is generally consistent over a wide span of time and space.

Alsea Watershed.—In the Alsea Watershed Study on the Oregon coast, fish populations and stream habitats were much less altered in a watershed that was patch-cut with intact streamside zones than in one that was completely clear-cut to both streambanks (Hall and Lantz 1969; Moring and Lantz 1975; Hall et al. 1987). This long-term evaluation of the influence of timber harvest on streams and their salmonid populations was established in 1958. It extended for 15 years and compared an unlogged control watershed with one that was completely clear-cut and another that was patch-cut with buffer zones left along the main channel.

Changes in physical habitat were extreme in the clear-cut basin. Only minor changes occurred in the patch-cut basin. Among the changes in the clear-cut stream was a large increase in summer stream temperature (Brown and Krygier 1970). A substantial reduction in dissolved oxygen occurred in surface water during the summer of logging, owing to accumulation of fine debris in the stream channel (Hall and Lantz 1969). During the first winter after logging, suspended sediment in the clear-cut stream increased about fivefold above the prelogging concentrations (Beschta 1978). Suspended sediment levels returned nearly to normal by the end of the 7-year postlogging phase of the study. There were increases in concentration and changes in vertical distribution of fine sediment in spawning gravels in the clear-cut watershed (Moring 1975a; Ringler and Hall 1988). As a consequence, gravel permeability in the clear-cut stream was substantially reduced below that in the patch-cut stream (Moring 1982). There were two episodes of short-term increase in suspended sediment in the patch-cut watershed caused by debris avalanches associated with roads, but these increases persisted for only 1 year. The avalanches occurred high in the watershed and formed settling basins that trapped most of the material from the hillslope. Thus they did not influence channel structure in the lower reaches inhabited by anadromous salmonids.

The two dominant salmonid species, coho salmon and cutthroat trout, responded differently to these habitat changes. Response of the coho salmon populations was difficult to assess. Smolt production in all three watersheds decreased after logging, as measured by the size of the normal smolt migration from February through May. However, owing to a progressive decline in numbers of smolts produced in the control watershed over the period of the study, and to substantial year-to-year variation in smolt abundance, the significance of the reductions was uncertain. The only change in abundance or behavior of smolt-sized coho salmon that could be attributed to timber harvest was a substantially earlier migration from the clear-cut watershed for the first 4 years after logging. A high proportion (average, 41%) of the total numbers of smolt-sized salmon migrated from November through January in those 4 years. The fate of these early downstream migrants could not be determined, but they were presumed to have suffered above-average mortality.

Another source of salmon production in these watersheds was affected by logging. Numbers of *migrant fry* that *left* headwater *streams soon after emergence* from the gravel declined to less than half the prelogging value in the clear-cut watershed (Hall et al. 1987) but changed little in the other two basins (Figure 14.5). The most likely cause of this reduced migration was the impaired gravel permeability, which could have substantially lowered survival from egg deposition to emergence (Hall et al. 1987). Cutthroat trout resident in the clear-cut basin during late summer decreased to about one-third of their prelogging abundance immediately following logging (Figure 14.6). Their numbers remained near that low level for the entire postlogging study period (Moring and Lantz 1975). Abundance of cutthroat trout did not change in the patch-cut or control streams.

By its completion in 1973, the Alsea Watershed Study had established the value of riparian zone protection during logging as a means of maintaining productive

SALMONID RESPONSES TO HABITAT CHANGE 499

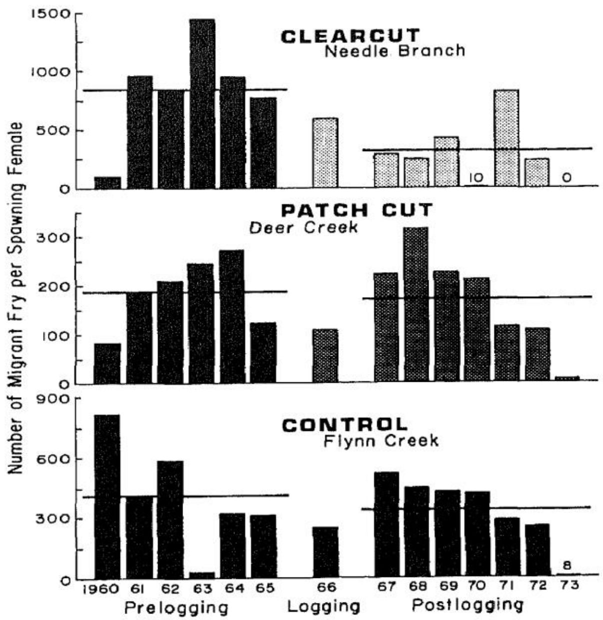

FIGURE 14.5.—Numbers of migrant coho salmon fry standardized by the number of spawning females that produced them in each stream of the Alsea Watershed Study, Oregon Coast Range. Horizontal lines are means of pre- and postlogging periods. (From Hall et al. 1987.)

fish habitat. Management practices and forest practice rules gradually began to incorporate greater protection for streamsides zones.

Carnation Creek.—The value of protecting the streamside zone was also demonstrated in the most comprehensive study of its kind to date, at Carnation Creek, British Columbia (see Poulin and Scrivener 1988 for an annotated bibliography). The watershed, on the west coast of Vancouver Island, has a cool, maritime climate. The study began in 1970 with a 5-year prelogging phase; it continued through a 6-year logging phase and a 5-year postlogging evaluation. Some monitoring continues, although the major research effort has been completed.

The study design involved three streamside harvest treatments, each applied to a different part of the same main-stem channel. In the lowest part of the basin, an unlogged buffer strip, varying in width from 1 to 70 m, was left along the stream. The next section upstream received a "careful treatment," in which all streamside trees were felled but were not yarded into or across the stream. Farthest

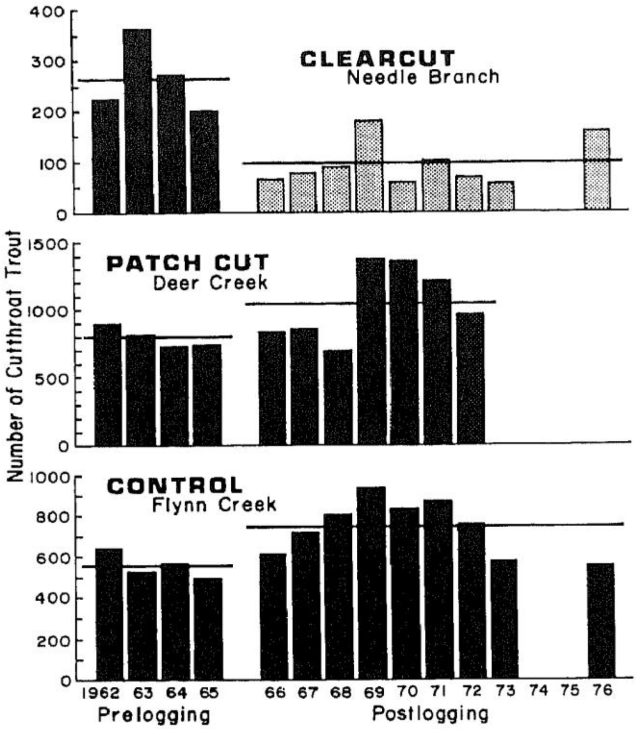

FIGURE 14.6.—Late-summer population estimates of juvenile cutthroat trout in the three streams of the Alsea Watershed Study, Oregon Coast Range. Horizontal lines are means of pre- and postlogging periods. (From Hall et al. 1987.)

up stream, in an "intensive-treatment" section, timber was felled into the stream and yarded from it (Hartman and Scrivener 1990). It was difficult to assess the effects of these treatments on salmonid populations because timber was harvested progressively and a variety of treatments was applied to the same channel. In several instances, downstream transport of material derived from a more severe upstream treatment influenced physical processes, habitat quality, and fish populations in the lower sections where treatment had been more conservative. Nonetheless, much useful information and several management recommendations have come from the study (e.g., Toews and Brownlee 1981; Hartman 1988). Several physical changes in the stream were related to the intensity of streamside treatment. Owing to removal of streamside vegetation from a substantial length of the main channel and of its tributaries, stream temperature increased in all months, the mean increase ranging from 0.7°C in December to 3.2°C in August. In contrast to other Pacific coast streams, in which autotrophic production increased notably when the canopy was opened, periphyton abundance in Carnation Creek was low before logging and did not increase after logging

(Shortreed and Stockner 1983). The failure of primary production to respond to increased light levels was attributed to a low level of phosphorus. The volume and stability of large woody debris decreased in the two sections without a buffer strip, but did not change consistently in the section with a buffer strip. Streambank erosion and stream widening were significantly greater upstream from the buffered section than within it; stream width increased nearly 2 m in the careful-treatment section between 1979 and 1985 but only about 0.1 m in the protected reach (Hartman and Scrivener 1990). Bank erosion and scour and redeposition of stream substrates were the primary causes of an increase in fine sediment in spawning gravels. The major source of fine sediment appeared to be the two sections without a buffer strip, especially an area affected by a debris torrent (Hartman and Scrivener 1990). The fraction of the bed composed of small gravel and sand increased from about 29% before logging to about 38% in 1985-1986 (5 years after the end of logging), an increase of about 30% above baseline (Scrivener and Brownlee 1989). Levels of suspended sediment were low and variable, and did not appear to increase significantly after logging.

Increased fine sediment in spawning gravel was accompanied by a reduction in estimated survival of chum salmon to emergence, from 22% before logging to 12% after (Scrivener and Brownlee 1989). Holtby and Scrivener (1989) predicted that this increased mortality, combined with an earlier migration of emergent fry to the estuary in response to increased stream temperature, would adversely affect future numbers of adult chum salmon returning to spawn. Adult returns were estimated to have been reduced by 25% as a result of logging, and year-to-year variation in abundance increased.

The number of steelhead smolts leaving Carnation Creek declined dramatically following the start of logging (Figure 14.7A), possibly because of reduced winter habitat for age-1 fish. Comparison of steelhead smolt production and adult numbers in Carnation Creek with numbers in adjacent rivers suggests that the changes were not part of a coast-wide trend, but rather the result of logging-related changes (Hartman and Scrivener 1990). Cutthroat trout smolts did not change in abundance following logging (Figure 14.7A). The size of smolts of both cutthroat trout and steelhead fluctuated substantially over the period of the study, but showed no clear change as a result of logging (Figure 14.7B). Coho salmon in Carnation Creek, like chum salmon, experienced a reduction in estimated survival to emergence as a result of increased fine sediment in spawning gravel caused by logging (Scrivener and Brownlee 1989). Average survival of coho salmon was 29% before logging but 16% after logging. The extended consequences of increased early mortality were difficult to interpret, however, because subsequent survival to spawning was influenced by other logging-related changes in the watershed as well as by climatic, biological, and fishery dynamics during the oceanic phase of the life cycle. Higher stream temperatures after logging caused juvenile coho salmon to emerge earlier and to grow faster during their first summer, resulting in larger sizes of both age-1 and age-2 smolts (Figure 14.8B). Winter survival from age 0 to age 1 in the main channel should have been reduced by decreases in the amount and stability of large woody debris (Tschaplinski and Hartman 1983). Because age-0 fish were larger in autumn, however, and because they made extensive use of a well-developed floodplain rearing area during winter, overwinter survival actually increased and a larger proportion of smolts went to

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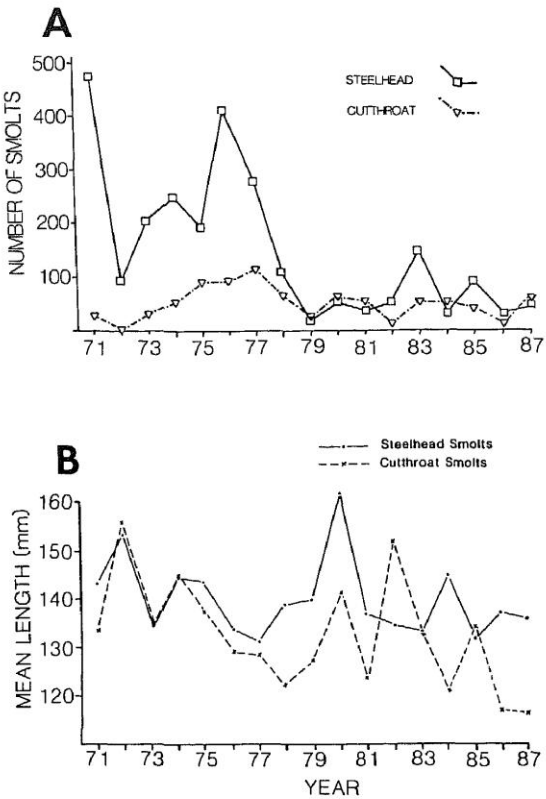

FIGURE 14.7.—Numbers (A) and size (B) of steelhead and cutthroat trout smolts leaving Carnation Creek from 1971 to 1987. Logging occurred between 1975 and 1981. (From Hartman and Scrivener 1990.)

sea at age 1 than was the case prior to logging. Consequently, total numbers of emigrants increased (Figure 14.8A). Nevertheless, the survival of smolts at sea probably declined because their spring migration took place an average of 2 weeks earlier than it did before logging (Holtby 1988b). The net result of logging was estimated to be a 6% reduction in returns of adult coho salmon and greater year-to-year variation in the number of returning spawners (Holtby and Scrivener 1989).

Changes in abundance of fish populations in Carnation Creek have been difficult to attribute to specific streamside treatments because there was only a single fish trap at the outlet of the watershed, and most of the assessment was based on overall changes in populations of the entire basin. In addition, there was no unlogged control watershed. These limitations were largely overcome by means of

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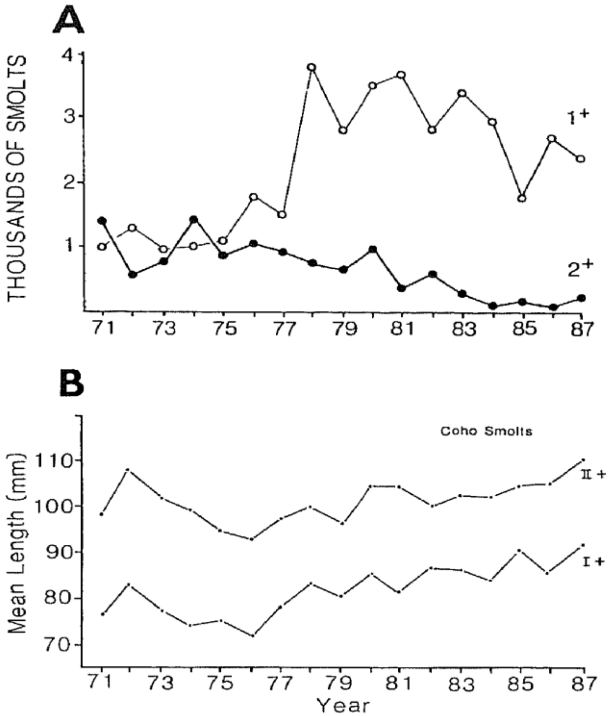

FIGURE 14.8.—Numbers (A) and size (B) of age-1 and age-2 coho salmon smolts migrating from Carnation Creek from 1971 to 1987. Logging occurred between 1975 and 1981. (From Hartman and Scrivener 1990.)

an elaborate series of linked regression models that accounted for the effects of logging, climate, and fishing (Holtby 1988b; Holtby and Scrivener 1989). This integrated approach to understanding the effects of logging on a salmonid population identified five distinct changes in the life history of coho salmon that were related to year-round temperature increases. These changes were (I) accelerated embryo development and earlier alevin emergence; (2) larger size at the end of summer, resulting from increased length of the growing season; (3) greater overwinter survival due to larger size; (4) more and larger age-1 coho salmon smolts and fewer age-2 smolts; and (5) earlier seaward migration of smolts in the spring. This analysis, which considered all life-history phases, illustrates the detail needed to understand the complex effects on salmonids of logging-related changes in streams. These efforts stand out as the most comprehensive attempt to place logging impacts within the context of the entire life cycle of an anadromous salmonid population.

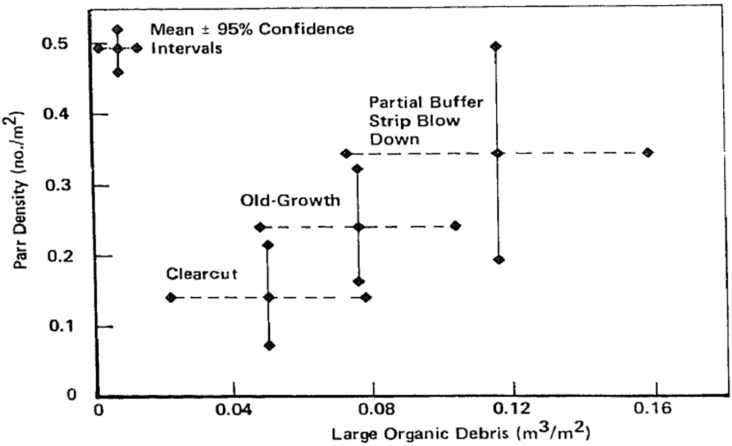

FIGURE 14.9.—Relationship between abundance of woody debris and density of coho salmon parr during winter in southeastern Alaska. Buffer-strip sites had experienced partial windthrow and had more debris than did old-growth sites. (From Murphy et al. 1984a.)

Southeastern Alaska.—Streamside management zones were shown to be effective in maintaining habitat and salmonid populations in a postlogging study of 18 streams in southeastern Alaska. Responses of coho salmon, Dolly Varden, and cutthroat trout populations to two treatments—clear-cutting to the stream edge (1-12 years previously) and clear-cutting with a buffer zone (3-10 years previously)—were assessed for 1 year (Murphy et al. 1986). Some buffer strips had experienced windthrow or had been selectively logged. In summer, coho salmon fry were more abundant in clear-cut and buffered reaches with partially open canopies than in old-growth reaches, apparently because increased primary production led to increased food availability. In winter, parr densities were positively correlated with the amounts of large woody debris present in streams (Figure 14.9). Juvenile salmonids depended on pools with cover created by large woody debris and on the protection provided by undercut banks (Heifetz et al. 1986). As a result, parr were less abundant in clear-cut reaches, where most of the large debris had been removed, than they were in old-growth and buffered reaches, where debris remained. Thus the increased abundance of coho salmon during their first summer was believed to have been more than offset by declines associated with loss of winter habitat (Koski et al. 1984). The net result was lower abundances of parr in, and lower predicted smolt outputs from, clear-cut reaches compared with sites with buffer strips or in old-growth forest (Murphy et al. 1986). Steelhead fry and parr responded like coho salmon (Johnson et al. 1986).

Northern California.—Logging in the north Coast Range of California was compatible with natural production of anadromous fish when adequate attention was given to stream protection (Burns 1972). Four streams were studied for one summer before, during, and after logging (3 years total). The two streams in watersheds that were carefully logged with protection of streamside vegetation showed minimal changes in habitat characteristics and in salmonid populations.

Cascade Mountains.—In some locations, lack of a streamside management zone has not appeared to be detrimental to production of salmonids in the years immediately following logging. Frequently, short-term production has even been enhanced by removal of all streamside vegetation. This was particularly true for small clear-cuts along high-gradient stream reaches where sediment and temperature changes were minimal (Gregory et al. 1987). A trend towards short-term enhancement of cutthroat trout populations in streams in the western Cascades of Oregon and Washington was shown by initial comparisons of logged and unlogged streams (Aho 1976; Hall et al. 1978; Murphy and Hall 1981; Bisson and Sedell 1984). Comparisons of 24 stream reaches in the Oregon Cascades showed that biomass of cutthroat trout was highest in clear-cut, intermediate in old-growth, and lowest in second-growth forests (Murphy and Hall 1981). The different responses of trout to logging in the Cascades and in the Oregon Coast Range (Alsea Watershed Study) led to further attempts to determine the underlying mechanisms. Paired logged and unlogged sites were compared in the Oregon Cascades and in the Oregon and California Coast Ranges (Murphy et al. 1981; Hawkins et al. 1983). The results of these studies, which included both high- and low-gradient streams, were generally consistent with the earlier work in the Cascades. Trout populations were higher in most of the recently logged sites than in the shaded forest sites.

Increased food abundance seemed to be the primary basis for increased growth and biomass of the trout populations in these streams. Opening of the canopy allowed greater primary production, which led to greater abundances of invertebrates in both the benthos and the drift (Hawkins et al. 1982; Wilzbach et al. 1986). In addition, trout were able to forage more effectively in the open sites; efficiency of prey capture was directly related to the amount of sunlight reaching the stream surface (Wilzbach et al. 1986). The conclusion from most of these studies was that greater food availability compensated for any negative effects of increased sedimentation and other habitat changes, at least in the first 10-15 years after logging. There were some indications that the return of heavy shade to the streams after 15 years was associated with a decrease in trout abundance to levels below those found in old-growth forests. This could be a particular problem in low-gradient sites, where sediment accumulation is greatest (Murphy and Hall 1981; Murphy et al. 1981). Timber harvesting along streams where streamside management zones are not provided, particularly when accompanied by removal of large woody debris from the channel and banks, often leads to substantial modification of channel structure (Sullivan et al. 1987). Such alterations may lead to changes in the relative abundance and age structure of salmonid species in these streams. Shifts in species and age composition were noted when populations of cutthroat trout, steelhead, and coho salmon were compared at 25 logged and unlogged sites in southwestern Washington (Bisson and Sedell 1984). The 12 logged sites had been clear-cut from 1 to 11 years prior to the survey. No protection had been provided in the streamside zones of the logged watersheds, and woody debris had been removed from the channels. Habitat changes associated with timber harvest and debris removal included decreases in pool area and volume and increases in riffle length. Where paired comparisons could be made in the same drainage basin, the biomass of all salmonids combined averaged 1.5 times greater in logged than in unlogged sites (Bisson and Sedell 1984). In most logged streams, the proportion of age-0 steelhead and age-0 cutthroat trout increased, but the proportion of age-0 coho salmon and age-1 and age-2 cutthroat trout decreased.

Bilby and Bisson (1987) studied the factors controlling the summer residency and production of age-0 hatchery-raised coho salmon in the upper Deschutes River, Washington, a relatively cool area at 1,200 m elevation. These fish were stocked at high density (up to 6/m^2) in paired old-growth and clear-cut watersheds. Production in the clear-cut watershed ranged from 1.2 to 2.6 times greater than production in the old-growth watershed over 2 years. Proportionately more fish remained in the old-growth site, which possessed better physical rearing habitat than did the clear-cut site. However, summer mortality in the old-growth site was significantly greater than in the clear-cut site, and growth rates were lower in the old-growth site. The number of coho salmon remaining in the two streams at the end of summer was not significantly different, but fish in the clear-cut site weighed about 20% more than fish in the old-growth site. Bilby and Bisson (1987) concluded that summer production was most strongly influenced by trophic conditions (the clear-cut site having more available food), whereas volitional residency was most strongly influenced by habitat quality (the old-growth site having better rearing habitats).

Benefits of streamside management zones.—The benefits of opening the canopy when timber is harvested to the stream edge may be offset by reduced diversity of fish species or increased annual variation in production. In the long term (over the length of a timber harvest rotation), most evidence indicates that careful streamside treatment is a better strategy than clear-cutting to the stream edge. Originally, the primary purpose of streamside management zones was control of stream temperatures. Recent timber harvest policies and forest practice rules have acknowledged the additional role of such zones in protecting channel stability and in providing future sources of large woody debris. We now recognize many more functions of the riparian zone (Gregory et al., in press). Future management efforts need to be directed toward prescription of the appropriate sizes and densities of trees in streamside management zones so that these zones accomplish all desired functions (Steinblums et al. 1984; Bisson et al. 1987).

In summary, intact streamside management zones have the following benefits:

- maintenance of stable streambanks, with overhanging cover and undercut banks.
- protection for stable large wood in the channel.
- provision for a continuing source of large wood for the future.
- maintenance of stable streambed and stream channel, minimizing increased sedimentation.
- prevention of substantial modification of stream temperature.

Effects of Roads and Hillslope Timber Harvest

Valuable as it may be, protection of streamside zones alone is not necessarily sufficient to insure maintenance of productive stream ecosystems in watersheds where timber is harvested. Particularly in steep terrain, debris avalanches and related mass movements of soil, timber, and debris from hillslopes may adversely affect salmonid habitats. Increased frequency of debris torrents resulting from road building and logging has been implicated as a major cause of degradation of fish habitat in some areas with steep slopes, unstable soils, and predisposing climate.

One series of studies has documented the effect of landslides on salmonids and their habitats in small streams on the Queen Charlotte Islands, British Columbia. Salmonid habitats were surveyed in 44 streams with known logging histories (Tripp and Poulin 1986b). Stream reaches studied included those in logged basins with debris torrents, those in logged basins without debris torrents, and those in unlogged basins without torrents. Methods of stream rehabilitation, effects of different cutting patterns, and alternative harvest methods 'to reduce mass soil movement were studied.

As part of the same study, salmonid abundance was investigated in 29 streams, and smolt migration was assessed in seven of these (Tripp and Poulin 1986b). Densities of juvenile coho salmon in late summer were almost as high in streams affected by debris torrents as they were in logged streams without torrents. The amount of cover present in summer was similar in both groups of streams but the type of cover differed greatly between groups. The cover in streams unaffected by debris torrents was predominantly rootwads, small and large woody debris, undercut banks, and deep water. In streams affected by debris torrents, cover was formed by rocks, boulders, and associated turbulence at the water surface.

Winter survival following logging was substantially lower in streams with debris torrents than in those without torrents, and was related to the type of cover in the two groups of streams. Winter cover was less abundant and less diverse, and pieces of large woody debris were less numerous, in streams affected by debris torrents than in unaffected streams. The amount of pool habitat regarded as good for overwintering juvenile salmonids, especially coho salmon, was reduced by an average of 79% in affected streams compared to unaffected ones, and the amount of cover suitable for overwintering was reduced by 75%. Correspondingly in these logged areas, overwinter survival of coho salmon averaged 1.8% in stream reaches affected by debris torrents compared to 24.5% in unaffected streams. Smolt yield averaged 0.02 fish per lineal meter in affected streams but 0.24 fish/m in unaffected streams, a 12-fold difference. Smolt yield from streams in unlogged areas unaffected by torrents was not measured, but was estimated to be 1.1-1.4 fish/m based on results from a stream in which habitat had been improved with channel structures (Tripp and Poulin, in press). Among western North American streams that flow through unlogged forests and have less than 10 km of their lengths accessible to anadromous salmonids, the average smolt yield is 1.4/m (Marshall and Britton 1980; Holtby and Hartman 1982).

Channel morphology in Queen Charlotte Island streams was profoundly influenced by logging-associated debris torrents. We can speculate on the species-specific effects of changes in stream reaches affected by debris torrents. Reduced average pool depth and area, reduced cover, and increased riffle area and channel width (Tripp and Poulin 1986b) would decrease habitat suitability for pool-dwelling species such as coho salmon. Comparison of hydraulic conditions in two Queen Charlotte Island streams, one in a logged and the other in an unlogged area, led Hogan and Church (1989) to conclude that there indeed was a decrease in rearing area for coho salmon at higher discharges in the logged area. Habitat suitability for species such as juvenile steelhead, which can occupy shallow-water habitats (Everest and Chapman 1972; Hicks 1990), might increase following logging. However, surface water flow decreased after mass wasting, owing to percolation through gravel in riffles and side channels (Tripp and Poulin 1986b), reducing the quality of these channel units as fish habitat. Riffle habitat quality was similarly reduced in Carnation Creek following logging (G. Hartman, Canada Department of Fisheries and Oceans, personal communication). The dramatically increased occurrence of mass wasting in logged basins, and especially in logged and roaded basins, indicates that channel widening, reduced area and quality of pools, and reduced surface flow might be a widespread result of logging in similar geoclimatic regions. Timber harvest and road construction on unstable slopes, combined with several years of above-average rainfall, damaged spawning areas in the South Fork Salmon River, Idaho (Platts and Megahan 1975). Fine gravel and sand (<4.7 mm) from weathered granite entered stream channels as a result of accelerated surface erosion and landslides, increasing river bedload by 3.5 times and practically destroying the spawning potential of the main river. A moratorium was placed *on logging and road construction on National Forest lands in the watershed* in 1965, vegetation was planted, and roads were stabilized, after which surface and landslide erosion declined dramatically. The proportion of fine sediments in spawning areas decreased progressively in four monitored areas from a range of 45 to more than 80% in 1966 to 12-26% in 1974, and the particle-size distribution returned to near optimum for spawning of chinook salmon (Platts and Megahan 1975). The response of fish runs to improved spawning conditions was difficult to evaluate owing to problems of fish passage at numerous downstream hydroelectric dams in the Snake and Columbia river basins. In this section, we have identified logging-induced changes in the rate of erosional processes as potentially damaging to salmonid populations. These harmful changes can be ameliorated by reduced road building and by careful design, placement, and construction of the forest roads that are built (Furniss et al. 1991). Lands especially susceptible to erosion should be excluded from logging. Exclusion of headwall areas from timber harvest, because of their high risk of landslides, has been used experimentally in the Oregon Coast Range (Swanson and Roach 1987).

Geoclimatic Trends in Salmonid Response

Geology, geomorphology, and climate control hillslope angles, soil depth, and resistance of bedrock to weathering. However, at the time scales considered in most studies of logging effects on salmonid populations (1-15 years), the geomorphic surface of a basin controls erosion rates (Gregory et al., in press).

Geomorphology and Geology

Hillslope angles, soil depth, and resistance of bedrock to weathering determine the *susceptibility of* hilislopes *to* failure, the geomorphic structure and stability of stream channels, and the hydrologic and nutrient regimes. In the Queen Charlotte

Islands, steep slopes, high rainfall, and erosive soils were responsible for the increased mass wasting that followed logging and damaged salmonid populations (Gimbarzevsky 1988; Poulin, in press).

The geomorphic configuration of a stream and its valley can influence salmonid population dynamics (Sullivan et al. 1987). Many of the juvenile coho salmon in Carnation Creek overwintered in the floodplain of the lower stream, which contained a network of sloughs, beaver ponds, and abandoned meander channels (Tschaplinski and Hartman 1983; Brown 1987). Winter survival and growth of coho salmon in these slack-water areas far exceeded those of coho salmon overwintering in the main stem (T. G. Brown 1985).

Geology has also been used to explain local differences in stream channel morphology and salmonid populations (Sullivan et al. 1987). In the Oregon Coast Range, cutthroat trout and steelhead predominated in watersheds underlain by basalt, whereas coho salmon predominated in streams cut through sandstone (Hicks 1990). Streams in sandstone valleys had lower gradients and a greater proportion of their lengths consisted of pools compared with streams of the same size in basalt areas, which were dominated by riffles. Boulders can create stable stream structure and diverse habitat in the absence of large woody debris, and boulder-rich streams can continue to support good populations of salmonids if debris is lost (Osborn 1981; Hicks 1990). Complex geologic factors such as resistance of bedrock, amount of fracturing, and distance from source control the size distribution of substrate elements (e.g., Hack 1957; Dietrich and Dunne 1978), which partly determines channel geometry and habitat quality (Hicks 1990). Differences in watershed geology influence nutrient availability, and these differences have helped explain some of the responses of fish populations to logging. In southeastern Alaska, periphyton, benthos, and coho salmon fry were abundant in both old-growth and clear-cut portions of watersheds rich in limestone. Little periphyton and few coho salmon were found in logged reaches in watersheds dominated by igneous rock (Murphy et al. 1986). Other studies also have related nutrient concentrations and biological productivity to watershed geology (Thut and Haydn 1971; Swanston et al. 1977; Gregory et al. 1987). Nitrogen is often the primary limiting nutrient in watersheds dominated by volcanic parent material, whereas streams draining glacial deposits or granitic bedrock are more often phosphorus-limited.

Geography and Climate

Studies of logging effects show some influences of latitude and climate. Mean annual precipitation, surface water temperature, and snowfall vary with latitude, altitude, and distance from the coast (Geraghty et al. 1973). As maximum surface water temperatures increase from north to south, thermal loading as a result of clear-cutting increases (Brown 1969, 1970; Beschta 1984). This effect is superimposed on higher ambient surface water temperatures in more southern and inland areas (Beschta et al. 1987). We speculate that seasonally related effects of logging on salmonid production are more severe in the southern portions of the species' ranges than in the north. Summer streamflows are lower in the south than in the north. Higher water temperatures and lower surface flows associated with logging, combined with higher ambient summer temperatures, may reduce food and space resources, which will intensify competition (Allen 1969) and limit survival of salmonids in summer.

Large woody debris mediates winter survival of salmonids throughout western North America, but the mechanism appears to differ between the northern and southern portions of their ranges. Protection from the high velocities of winter peak flows is the main role of woody debris in areas with winter floods (Chamberlin et al. 1991). In the north, where winter is a period of low flow, the maintenance of pool depth by scour around woody debris is important in preventing stream freezing.

Our attempt to evaluate trends in response of populations from west to east, from the Cascades to the Rocky Mountains and eastward, was hampered by a lack of data. There have been several coordinated studies, including those of the Slim–Tumuch watershed in British Columbia, the Tri-Creeks system in Alberta, and the Nashwaak basin in New Brunswick (Macdonald et al. 1988), and several isolated investigations (e.g., Platts and Megahan 1975; Welch et al. 1977; Grant et al. 1986; Lanka et al. 1987), but little has been published on the response of salmonid populations to timber harvest in these more easterly areas. Climate also influences the occurrence and rate of mass wasting, which is the predominant means by which large amounts of soil, rock, and organic material are delivered to streams (Everest et al. 1987a). Antecedent weather conditions and storm intensity were more important than geology in determining the susceptibility of different bedrock types to erosion in the Queen Charlotte Islands (Poulin, in press). Bedrock type, however, influenced the slope of valley walls, which in turn influenced the extent of mass wasting (Gimbarzevsky 1983, 1988). The effects of mass wasting on winter fish survival in logged streams affected by debris torrents were thought to be so severe that too few adults might return to sustain a population of coho salmon in the future (Tripp and Poulin 1986b). Geomorphologic and geologic variables have been used successfully to explain salmonid abundance in logging-related studies (Ziemer 1973; Swanston et al. 1977; Heller et al. 1983; Lanka et al. 1987). However, no two studies have identified the same set of variables as being important. This shows that we have some understanding of the link between geology, geomorphology, and fish abundance that can be useful locally to predict the effects of timber harvest, but that we still have no models that apply over a wide area.

Limits of Present Knowledge

Uncertainties remain about the effects of forest practices on salmonid populations, demonstrating the limits of our present knowledge. Nevertheless, there appear to be some unifying links among studies of the integrated salmonid response to logging. An increase in solar radiation reaching a stream after removal of streamside vegetation usually enhances production of algae and invertebrates. The net effect is often greater salmonid production within the first 20 years after timber harvest. At least some of the benefit of increased summer production can be lost by reduced winter survival. Reduction in winter survival can result from loss of habitat structure caused by reduced amounts and stability of large woody debris and by increased amounts of coarse sediment. Differing habitat requirements of salmonid species such as coho salmon and steelhead (Hartman 1965) result in varied responses to logging-related changes in channel morphology (Reeves and Everest, in press). Reductions in habitat complexity and pool depth truncate the diversity of species and age-classes, favoring single species and age-classes. Increases in riffle area increase the suitability of habitat for steelhead, if streamflows are sufficient to maintain adequate water depth.

Incongruous Time Scales

In our opinion, one explanation for the incomplete state of knowledge lies in the very different time scales characteristic of management practices, natural processes, and evaluative studies. The period of a single rotation in timber harvest in western North America is 45-100 years. The natural processes that influence stream productivity and that may mitigate changes caused by timber harvest operate over a variety of time scales ranging from months to centuries. Yet almost all of the studies evaluating the effects of forest management on stream habitat and salmonid populations have encompassed less than 5 years. Nearly all have concentrated on changes immediately following timber harvest. Of the few studies at the population level carried out to date, only the Carnation Creek and the Alsea Watershed studies extended for 15 years or more, and even that length of time is inadequate for full evaluation of logging effects. Some population studies (e.g., Murphy and Hall 1981) dealt with streams whose watersheds had been logged as many as 30-40 years earlier, but none of the studies was capable of monitoring the effects of environmental disturbance over the length of a rotation. The need for long-term evaluation is further illustrated by observations from the Carnation Creek study. Had the study terminated in the first few years after logging, the eventual impact on the coho salmon population would not have been detected (Hartman et al. 1987). Studies of salmonid populations in Great Britain (Egglishaw and Shackley 1985; Elliott 1985) also show the need for long-term research. These studies monitored anadromous salmonid populations in watersheds that had not experienced recent disturbance, yet Elliott (1985) noted that even 18 years of data on population abundance, streamfiow, and water quality had failed to provide a clear indication of the environmental factors that limit production. Short-term studies cause a biased perspective of habitat change. Some change that is most apparent persists for only a few years. Undue concern may be raised over a condition that is transient in the time scale of forest succession. Increased levels of suspended and deposited sediment could be considered an example of this sort (Everest et al. 1987a). In contrast, some effects that are beneficial or neutral in the short term can be followed by negative effects over the longer term. The short-term increase in productivity that accompanies higher light levels after canopy removal from some streams is an example. As the canopy closes during forest succession, the overall production of salmonids may drop below that of unlogged watersheds and remain there for many years (Gregory et al. 1987). Speed of canopy closure following removal of streamside vegetation depends on the local environment. Return of insolation at the stream surface to preharvest or lower levels as streamside vegetation regrows may be rapid in some locations such as the Oregon Coast Range, but the canopy may remain open for many years in an environment where plant growth is slow (Summers 1983).

A decrease in the supply of large woody debris is another change in habitat that may not occur for many years. Recent studies in Oregon have indicated that many existing stable pieces of large woody debris entered streams years ago. Heimann (1988) found that in watersheds logged without a buffer strip, preharvest debris continued to be the predominant wood in streams 140 years after timber harvest, but the total amounts of debris were only about 30% of preharvest levels. Unless adequate provision is made to supply streams with large pieces of wood into the future, the quality of fish habitat may decline severely. Decomposition times of some tree species are so long (e.g., >100 years for western redcedar: Swanson ct al. 1976) that there may be a lag time of many years before the shortfall is apparent. Two problems confound the clear separation of logging effects from those caused by other environmental changes and by fishery management activities. The first problem is that most studies of logging effects have used present-day habitats and fish populations as the baseline for comparison with postlogging conditions. As a consequence, logging-related reductions of salmonid populations that may have occurred over the past 100 years or more have gone undetected. In many areas, present-day levels of wild salmonid stocks are only small fractions of historical abundances. There has been little study of long-term (100 years or more) change in habitat of the sort undertaken by Sedell and Luchessa (1982) and Sedell and Froggatt (1984). Among the causes of habitat loss have been channel scour during log drives, removal of debris to improve navigation, logging of riparian conifers, and salvage of large wood from stream channels. The second problem is that the effects of logging are difficult to separate from the effects of other activities, such as power generation, irrigation, grazing practices, and fishery management (Pella and Myren 1974). Overfishing and loss of productive spawning and rearing habitat (e.g., Chapman 1986) have occurred concurrently with logging. In particular, excessive ocean harvests, especially of coho and chinook salmon, have often resulted in inadequate returns of wild fish to spawn. The low level of spawning has complicated assessment of the natural productive potential of streams (Cederholm et al. 1981). These conditions combined have made it extremely difficult to separate logging effects on fish populations from the effects of other natural and cultural influences within the time scales of evaluative studies.

Climatic Variation

A further difficulty in tracking logging-related habitat disturbances over time is introduced by long-term fluctuations in climate or the effect of a single event of very large magnitude. Large floods occurring periodically have disproportionate influences on channel morphology. Unusually severe storms occurred in the northwestern USA in 1949, 1955, 1964, and 1972. These intense storms caused high discharges, bed-load movements, landslides and associated debris torrents, and erosion of streambanks and floodplain terraces, all of which resulted in substantial, widespread changes in stream habitats. To some extent, timber harvest has intensified the effects of such floods on stream habitats (Anderson et al. 1976), and evidence of the resulting channel changes may persist for decades or more (Swanson et al. 1987). Frissell and Hirai (1989) described a complex interaction between land-management practices and long-term changes in the timing of storms that appears to have contributed to reductions in spawning success of chinook salmon in coastal streams of southern Oregon. Less apparent physically, but perhaps no less important biologically, are severe droughts such as those that occurred in the 1930s. Another unusual climatic event that can exert a strong influence on population abundance is an oceanic El Nino (Bottom et al. 1986; Johnson 1988).

In the evaluation of logging-related habitat changes and their effects on fish populations, disturbance regimes imposed by management should be compared to natural disturbances in both frequency and amplitude. Salmonid populations in western North America have evolved in a landscape where they experienced severe environmental stresses, including forest fires, floods, mass soil and ice movements, debris torrents, glaciation, and volcanism. Through evolutionary processes, salmonids have developed life history strategies enabling them to withstand considerable environmental variation and unpredictability. Salmon and trout accomodate exploitation and other stresses, at least in the short-term, through compensatory responses at one or more life stages. However, some effects of timber harvest may be more severe than similar natural perturbations, resulting in mortality beyond the range of compensation. For example, the abundance of large woody debris has been more substantially reduced by logging and associated salvage and stream-cleaning activities than by wildfire or volcanism (Sedell and Dahm 1984).

A major consequence of all these sources of variability in the natural environment is the large natural variation—both temporal and spatial—in the abundance of salmonids (Burns 1971; Lichatowich and Cramer 1979; Hall and Knight 1981; Cederholm et al. 1981; Platts and McHenry 1988). This variation is often large enough to obscure changes caused by timber harvest or other land management activities (Pella and Myren 1974; Platts and Nelson 1988).

Spatial Scales

Incongruence of spatial scale, as well as of temporal scale, has hindered interpretation of many studies. Bisson (in press) compared typical variations in western Washington salmonid populations in space and time on the basis of both standing stocks and smolt yields. He concluded that the smaller the scale, the greater the variability among sites. That is, differences in population abundance and smolt production tended to be greatest among limited stream reaches, whereas differences in smolt yield for whole basins tended to be proportionately smaller. Because most studies of timber harvest have focused on stream reaches or small watersheds, they have taken place at a spatial scale often characterized by relatively great variability, thus making logging-related changes more difficult to separate from natural variation.

Focus on individual stream reaches has also limited the accuracy of many studies. The study designs used have underestimated sampling error and have not addressed the comparative importance of different parts of a basin (Hankin 1984, 1986). Reach analysis does not effectively assess cumulative effects that may accrue from a multitude of small disturbances within a large watershed, or downstream changes that result from a single type of disturbance repeated over time. The criticism applies most strongly to short-term postharvest evaluations, but the limitation also occurs in watershed-level studies.

There may be problems of scale in extrapolating work in relatively small basins to larger watersheds. For instance, debris torrents that remove channel structure from steep, lower-order drainages can add structure to higher-order, lower-gradient reaches downstream (Benda 19856). In addition, larger main-stem reaches have been shown to rear proportionately more fish on a basin-wide scale than the combined lower-order tributaries, despite high fish densities in headwater streams (Dambacher, in press; J. R. Sedell, U.S. Forest Service, unpublished). Though main-stem production in some systems is now low, its historic role may have been greater before channel complexity was lost (Sedell and Froggatt 1984). There are several reasons why studies have been concentrated in small basins. First, isolation of the effects of timber harvest from other activities requires working within watershed boundaries where logging is the only management influence. The mosaic of patterns of land use and other influences found in large basins therefore usually limits the size of study basins to small watersheds. Logging of large basins also occurs over too long a period to allow evaluation of its effects, because only a small proportion of the watershed area is normally harvested at any one time. Secondly, small streams have often been selected for study because they could be sampled easily, rather than because of their significance to total basin production. This is particularly true for the siting of facilities for counting salmonid smolts and adults. Maintaining effective fish traps during high streamflows is a major constraint on the size of stream that can be effectively monitored. Recent development of small, portable, floating smolt traps (McLemore et al. 1989) should provide the ability to monitor much larger systems than has been possible in the past.

Suggestions for Future Studies

We suggest three principal approaches for evaluation of the influences of logging on salmonids: (1) substitution of space for time by extensive evaluation of many watersheds in a short period, (2) long-term studies before and after logging, and (3) innovations in design that could improve data collection efficiency or strengthen the inferences drawn from data. In earlier analyses, Hall et al. (1978) and Hall and Knight (1981) suggested that studies of the effects of logging could be grouped in a two-way classification. The basis for grouping is (1) whether the studies are before-and-after comparisons or posttreatment evaluations of timber harvest, and (2) whether detailed studies are carried out on one or few streams (intensive) or shorter surveys are conducted on many streams (extensive). This classification results in four categories, the advantages and disadvantages of which are listed in Table 14.3. There is no single optimum study design, but Hall and Knight (1981) favored the extensive posttreatment survey, especially when it is combined with carefully designed process-oriented studies. Given the substantial work done since that time, and a recent further evaluation of study design (Grant et al. 1986), we have reviewed those conclusions. Extensive posttreatment analysis, especially when coupled with pairing of treatment and control streams or watersheds, is an attractive and relatively inexpensive method of analysis. In certain circumstances, it appears to have been effective (e.g., Murphy and Hall 1981; Bisson and Sedell 1984; Murphy et al.

TABLE 14.3.—Summary of advantages and disadvantages of the four major approaches to watershed stream analysis. (From Hall and Knight 1981.)

Advantages	Disadvantages
(A) Intensive before-after (5–7 years before treatment, 5–7 years after)	
Possible to assess year-to-year variation and place size of effects in context of that variation	No replication; results must be viewed as a case study
	Results not necessarily applicable elsewhere (areas of different soils, geology, fish species, etc.)
Can assess short-term rate of recovery (about 5 years)	Results influenced by changes in weather
No assumptions required about initial conditions	Final results and management recommendations require exceptionally long time to complete—15 years or more after initial planning stage
Possible to monitor effects over whole watersheds (provided substantial investment in facilities such as flow and sediment sampling weirs, fish traps)	Difficult to maintain intensity of investigation and continuity of investigators over such a long period
	Considerable coordination required; must rely on outside agencies or firms to complete logging as scheduled
Long time frame provides format for extensive process studies	
(B) Extensive before–after (1–2 years before treatment, 1–2 years after)	
Provides broader geographical perspective than (A)	Little opportunity to observe year-to-year variation
	Able to assess only immediate results, which may not be representative of longer time sequence
Larger number of streams lessens danger of bias by extreme case	Treatment influenced by unusual weather
Increased generality of results allows some extrapolation to other areas	Must rely on coordinated efforts (see A above)
Relatively short time to achieve results (3–4 years from planning stage)	
(C) Intensive posttreatment (one watershed, paired sites; several years after treatment)	
Shorter time for results than (A)	Provides no separate control stream; requires assumption that upstream control reach was similar to treated area prior to treatment
Moderate ability to assess year-to-year variation	
Provides opportunity for moderate level of effort on process studies	"Control" most logically must be located upstream of treatment; strong downstream trend in any variable would confound analysis
	Provides no spatial perspective; results of limited application elsewhere
(D) Extensive posttreatment (e.g., 10–30 watersheds); all observations made in 1–2 years (time after treatment variable)	
Wide spatial perspective allows extrapolation to other areas	No data available on pretreatment conditions, forcing assumption that control and treatment sites were the same before treatment
Long temporal perspective is possible; recovery can be assessed for as many years as past logging has occurred	Control predominantly upstream
	Total cost and sampling effort concentrated in short period; requires extensive planning
Provides ability to assess interaction of physical setting and treatment effects (e.g., effects of sediment input at different stream gradients)	Not as effective as (A) in assessing whole watershed effects
Requires least time of the four designs to complete (as little as 2 years)	Harvest methods used in early logging may not be comparable to those used later
Probably most economical of the four approaches per unit of information	

1986). In a test of this study design in some streams in eastern Canada, Grant et al. (1986) concluded that the basic structure of the design was valid, although they detected no clear changes in salmonid biomass in response to timber harvest. However, comparison of paired sites close together along the same stream channel is less appropriate for assessing the effects of whole watershed disturbances than for assessing the effects of localized disturbances. Further consideration has led us to conclude that stream-reach comparisons, a feature of most extensive survey designs, should be broadened to a larger watershed perspective. Recent development of basin-wide sampling designs for both habitat characteristics and fish populations (Hankin and Reeves 1988) should allow more effective assessment of habitat changes in entire basins.

Some studies, particularly earlier ones, did not examine certain key habitat variables that we now know to be important. Recent advances in the classification and quantitative description of stream habitats (Bisson et al. 1982; Frissell et al. 1986; Grant 1986; Sullivan et al. 1987), along with recognition of the important role of large woody debris (Sedell et al. 1984, 1988), have improved our ability to describe deleterious changes in habitat quality. Nonetheless, our present understanding seems inadequate to predict changes in fish populations by analysis of habitat changes alone. Understanding often seems inhibited by a tendency to focus on a limited subset of habitat variables, such that insufficient consideration is given to potential limiting factors (Everest and Sedell 1984; Chapman and McLeod 1987; Everest et al. 1987a; Bisson, in press).

Research effort has not been applied equally to all species. There has been a strong bias towards studies of coho salmon, cutthroat trout, and steelhead (Table 14.2). Chinook salmon in particular have been much less studied relative to their distribution and abundance, probably because most studies have focused on small watersheds. There has been some emphasis on pink and chum salmon in Alaska and British Columbia, but little of the work has been focused on responses at the population level.

We suggest that more emphasis is needed on long-term analysis of logging impacts. This recommendation is based on several related observations. The successful and insightful analyses that have come from the Carnation Creek study could not have resulted from a shorter-term study. The evaluation of long-term ecological studies by Strayer et al. (1986) highlighted the advantages and contributions of such studies and provided a good perspective on the value of extended monitoring. Increased support from the U.S. National Science Foundation for research on long-term ecological reserves reflects a recognition by the scientific community of the value of such work. Finally, analysis of long-term trends in habitat condition (Sedell and Luchessa 1982; Sedell and Froggatt 1984) has convinced us that gaining an accurate perspective on change in habitat may require a very long time.

One innovative study design was suggested by an historical analysis of habitat condition prior to large-scale timber harvest and salvage of downed timber in the

Breitenbush River in western Oregon (J. R. Sedell, U.S. Forest Service, unpublished). The original data forms from a survey of the river system conducted in 1937 by the U.S. Bureau of Fisheries were recovered. The survey was repeated in 1986 and showed a loss of more than 70% of pool habitat over the past 50 years. The river had changed to a single channel without debris jams and did not match its earlier description as "a mass of channels criss-crossed with downed logs and debris." The early surveys did not include quantitative assessment of fish populations, however. In a related attempt to evaluate the cumulative effects of timber harvest on channel structure, Grant (1988) used aerial photographs spanning the years from 1959 to 1979 to discern increases in stream channel width in the same reach of the Breitenbush River. Lyons and Beschta (1983) also used analysis of aerial photographs to measure changes in channel width in the Willamette River drainage in the Oregon Cascade Range over the same time period. They found major increases in channel width associated with timber harvest and road building. If it were possible to couple such analyses with models relating habitat quality to population abundance (see Fausch et al. 1988), a much clearer picture of the long-term effects of logging might emerge. We suspect that there are many such early surveys containing valuable information that have since been misplaced, forgotten, or otherwise ignored. If properly interpreted, these archived surveys could be used to provide a valuable baseline against which present-day conditions could be compared.

One conclusion that cannot be avoided from the studies relating salmonid populations to timber harvest activities is that fish habitats have been simplified over time and that this process of simplification continues. In this respect, agriculture, mining, grazing, and urbanization have the same net negative effects on habitat diversity and population abundances of major fish groups. The phenomenon is world-wide and is well documented for agricultural lands (Karr et al. 1983) and urbanized areas (Leidy 1984; Leidy and Fiedler 1985). Habitat degradation by simplification is one of the most important limits on the diversity of fish communities (Karr and Schlosser 1978). Highly diverse fish communities have been documented in association with stable channels, instream cover from boulders, living streamside trees, and downed trees in channels (Karr and Schlosser 1978; Welcomme 1985). Our understanding of the changes in stream habitat and salmonid populations brought about by timber harvest has improved, but our knowledge remains limited. Certain changes in stream habitats caused by logging—increased temperature and fine sediment, for example—appear to be less detrimental and more transient than originally perceived. Others—such as loss of large woody debris and changes in channel morphology and stability—were not forseen as problems earlier but have contributed to significant habitat degradation in the long term. We are concerned that logging impacts will become more severe as timber harvest moves to progressively less stable landforms and as the area of managed forest increases. As timber is harvested from the accessible and gentle terrain, logging occurs on steeper and less stable slopes, as it has in the Queen Charlotte Islands, for example. The results there and from other areas of steep terrain in western North America argue for intensified efforts to develop hazard-assessment techniques that will predict the potential disturbances to fish habitat from mass erosion and debris torrents. Some progress in this direction has been made (Benda 1985b; Swanson et al. 1987).

There is still much to learn before we can predict with confidence the effects of a particular logging operation on salmon and trout populations or prescribe management activities that will provide optimum habitat protection. However, a much wider array of analytical tools is now available. Several useful management guidelines and procedures for translating research findings into management practices have recently been produced (e.g., Toews and Brownlee 1981; Bryant 1983; E. R. Brown 1985; B.C. Ministry of Forests et al. 1988; Hartman 1988; Bilby and Wasserman 1989).

Finally, we emphasize that our review of 30 or more years of research on forestry and fisheries interactions has yielded several general principles that should be considered when logging operations are planned and implemented. Some of the most important of these follow.

- Protection of streamside zones by leaving streamside vegetation intact will help maintain the integrity of channels and preserve important terrestrial–aquatic interactions.
- Productivity of streams for salmonid populations tends to be enhanced under conditions of moderate temperatures, low to moderate sediment levels, high light levels, adequate nutrients, an abundance of cover, and a diversity of habitat and substrate types.
- Productive floodplain and side-channel habitats should be protected.
- Streams should be protected against frequent and extreme episodes of bed-load movement or sediment deposition through careful streamside management and through proper planning and engineering of roads and timber harvest systems.
- Management of streamside zones should include provisions for long-term recruitment of large woody debris into stream channels and for protection of existing stable large woody debris.
- The geology, geomorphology, and climate of a watershed mediate the response of fish populations to timber harvest. Site-specific management recommendations must consider regional landforms and climatic variation.

No single research approach will provide all the answers to the many complex questions that remain. We advocate a combination of short-term extensive studies over broad geographic regions and long-term intensive studies that emphasize critical terrestrial–aquatic interactions at the watershed scale. We further suggest that future long-term studies involve both biologists and foresters at the planning stage so that various logging treatments can be experimentally evaluated in a sound scientific manner. This process will also help to reduce the perceived conflict between forest and fishery management that has all too often hindered progress. We note that two of the most successful research efforts, the Alsea Watershed Study in Oregon and the Carnation Creek study in British Columbia, used such an approach. Coordinated experiments that are thoughtfully designed can speed solution of many of the remaining problems, resulting in our improved ability to intelligently manage all the resources of forested watersheds.

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The Effect of Deposited Fine Sediment on Summer Survival and Growth of Rainbow Trout in Riffles of a Small Stream

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[Article]

The Effect of Deposited Fine Sediment on Summer Survival and Growth of Rainbow Trout in Riffles of a Small Stream

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Abstract.—Elevated fine-sediment inputs to streams can alter a variety of conditions and processes, including the amount of fine sediment stored in riffles. We sought to measure the influence of deposited fine sediment on the survival and growth of juvenile rainbow trout *Oncorhynchus mykiss* (106–130 mm fork length) using a field experiment that included 18 enclosures in riffles of a small northwestern California stream. The experiment included six replicates of three levels of deposited fine sediment (low, background, and high) that embedded riffle cobbles at 0, 50, and 100%, respectively. Only 1 of 12 fish survived in high-sediment enclosures, while survival of fish in low- and background-sediment treatments equaled or exceeded 50%. Low- and background-sediment treatments could be distinguished from each other by a difference in fish growth: fish in the low-sediment treatment gained mass, on average, while all surviving fish in the background-sediment treatment lost mass. In addition to providing relatively high survival and growth benefits for juvenile rainbow trout, low-sediment experimental units were colonized at significantly higher rates by other vertebrates, particularly coastal giant salamanders *Dicamptodon tenebrosus*. The amount of stored fine sediment in small streams may substantially influence the total amount of habitat available to vertebrates at the watershed scale.

Human activities such as road building, agriculture, and timber harvest can elevate the input of fine sediment into stream channels. This sediment may be periodically transported as suspended load or bed load, or stored in the channel. All three of these outcomes may have important consequences for stream biota. For example, elevated suspended load increases turbidity, which reduces capture success for drift feeding fish (Sweka and Hartman 2001), while higher bed load transport may affect the density of benthic invertebrates (Culp et al. 1986) and the reproductive success of some fishes (Montgomery et al. 1996). During periods of low to moderate streamflow when the capacity for sediment transport is low, stream biota may be most affected by fine sediment stored in the channel.

Stored fine sediment can affect fish in several ways. Channel aggradation can reduce surface streamflow by increasing flow through the streambed, thereby reducing fish survival (May and Lee 2004) and growth (Harvey et al. 2006). Deposited sediment can also reduce the availability of benthic invertebrates vulnerable to fish predators due to species-specific substratum preferences and increase agonistic interactions among fish due to reduced visual isolation. Suttle et al. (2004) linked both of these mechanisms to a negative relationship between the growth of age-0 steelhead (anadromous rainbow trout *Oncorhynchus mykiss)* and the embeddedness of the substratum in artificial channels placed in pools. The composition of the substratum can also influence predation risk for fish, with consequences for both growth and survival. Fischer (2000) measured 30% higher respiration rates for burbot *Lota lota* on pebble substratum compared with cobble substratum, even in the absence of predators. Fish may forego feeding opportunities if feeding requires substantial exposure to predators. Coarse substratum is more likely to provide microhabitats offering profitable feeding opportunities with relatively low predation risk.

Substrate-dependent predation risk can also directly influence survival (e.g., White and Harvey 2001). The potential value of coarse substratum in reducing predation risk by providing cover is indicated by its increased use by fish when predators are present (Vehanen and Hamari 2004). Relatively coarse, unembedded substratum may be particularly important in shallow habitats in small streams because predation risk can be depth-dependent (Harvey and Stewart 1991), and low streamflows may provide little surface turbulence to conceal fish. In this study, our objective was to measure the effect of deposited fine sediment on the survival and growth of juvenile rainbow trout in shallow habitat in a small stream during summer, using a field experiment that incorporated the potential effects of deposited fine sediment on both energetics and predation risk.

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Study Site

Jacoby Creek drains a 42-km^2 watershed into Humboldt Bay in the Coast Range of northwestern California. The study area lies within the Arcata City Forest in the upper drainage (watershed area = 15 km^2) at an elevation of approximately 250 m. The active stream channel in the reach averages about 4 m wide and alternates between shallow riffles and moderately deep pools (maximum depth ~ 1.5 m) over gravel and cobble substrate. Except for one 50-m-long bedrock-confined section, gradient in the reach rarely exceeds 1–2%. The region experiences dry summers and wet winters. Because of the moderating effect of coastal fog, air temperature at our study site rarely exceeds 25°C in the summer or drops below 4°C in the winter, while water temperature ranges from 9°C to 16°C in summer and from 4°C to 12°C in winter. Mean annual rainfall exceeds 120 cm, more than 90% of which falls between November and March. Streamflow in the study reach averages less than 0.05 m^3/s in the summer (mean = 0.022 m^3/s; range = 0.017–0.029 m^3/s during this study) and approximately 0.75 m^3/s in the winter with peak flows greater than 100 m^3/s. The combination of a steep, highly erosive watershed, intensive land management (principally logging), and high rainfall delivers substantial sediment to Jacoby Creek. Consequently, extensive bed load movement and channel changes result from high flows in most winters. While the stream remains clear (<5 nephelometric turbidity units [NTU]) in the summer, turbidity averages over 30 NTU in the winter, with peaks in excess of 1000 NTU. Coastal redwood *Sequoia sempervirens* and Douglas-fir *Pseudotsuga menziesii* constitute the majority of the forest cover in the watershed, although red alder *Alnus rubra* and bigleaf maple *Acer macrophyllum* dominate much of the riparian zone. The study reach lies upstream of barriers to fish passage. The fish assemblage in the study reach consists solely of a resident population of rainbow trout. Coastal giant salamander *Dicamptodon tenebrosus*, tailed frogs *Ascaphus truei*, and red-legged frogs *Rana aurora* complete the aquatic vertebrate assemblage. Piscivorous vertebrates observed in the study area include river otter *Lutra canadensis*, raccoon *Procyon lotor*, great blue heron *Ardea herodias*, green heron *Butorides virescens*, belted kingfisher *Ceryle alcyon*, American dipper *Cinclus mexicanus,* and garter snakes *Thamnophis* spp.

Methods

In early July 2002, we selected 18 riffles in a 1.6-km study reach as experimental units. The units spanned the channel, varied in length from 2.25 to 3.75 m, and enclosed areas of 7–7.75 m^2. We randomly assigned six replicates of three fine-sediment treatments (high, background, or low) to the 18 units. Prior to any substrate manipulation, we determined the background percentage of fine sediment in the bed material to a depth of 10 cm in each riffle unit by sieving through a 13-mm-square mesh a 0.025-m^3 sample of the native substrate that had been collected with a shovel. This step revealed that background fine sediment averaged 40% by volume with low variability. In unmanipulated riffles, surface cobbles were embedded in finer material to a depth of about 50% of their height, on average, with some cobbles in each unmanipulated experimental unit 0–25% embedded. We set the high-sediment treatment at twice the percentage of fine sediment in unmanipulated riffles. We added sand and gravel from nearby exposed bars to increase the fine sediment concentration in high-sediment treatments. This almost completely embedded all cobbles within the enclosures, although each high-sediment enclosure retained a small number of cobbles embedded about 50%. For the low-sediment treatment, we removed all the native substrate within units to a depth of 10 cm with shovels, sieved out the fine sediment using 13-mm-square mesh, and then returned the larger fraction to the unit. This procedure yielded a completely unembedded surface layer of cobbles and coarse gravel. To increase the similarity of disturbance to the substratum among treatments, we brushed the surface of the substrate in the background-sediment treatment with a coarse-bristle push broom and removed and replaced cobbles. All sediment manipulations were completed on 2–3 July 2002.

After waiting 27–28 d to allow recolonization of disturbed substratum by periphyton and benthic invertebrates, we enclosed each unit at the upstream and downstream boundaries with plastic fencing (6-mm mesh) which was buried at least 10 cm below the stream bottom in fine gravel, extended at least 30 cm into the streambank, and was supported by metal stakes. We also made two passes with a backpack electroshocker and removed captured vertebrates from each experimental unit. On 31 July and 1 August 2002, we completed an additional two or three electrofishing passes to ensure removal of all rainbow trout older than age 0 and stocked enclosures with fish collected from outside the experimental units by electrofishing. We stocked each unit with two juvenile rainbow trout that differed in length by a minimum of 10 mm fork length (FL; larger fish: mean FL = 124 mm, range = 116–130; smaller fish: mean FL = 111 mm, range = 105–116) and in mass by at least 3 g (larger fish: mean mass = 21.3 g, range = 17.8–24.4 g; smaller fish: mean mass = 15.1 g, range = 12.0–18.2 g). We selected fish of different sizes to avoid persistent agonistic interactions between fish within enclosures. Before releasing fish into the experimental units, we measured their FL to the nearest millimeter and their wet weight to the nearest 0.01 g with an electronic balance, and inserted a passive integrated transponder (PIT) tag to allow identification of individuals. Enclosure fences were cleaned once or twice each week over the 6-week experiment. We did not include age-0 fish in experimental manipulations because they were too small to be contained by enclosure fences at the beginning of the experiment. The fences also allowed passage of small larval coastal giant salamanders.

To compare physical conditions among treatments, we measured depth, water velocity, and substrate at 15 points within each experimental unit after installing the enclosure fences. Measurements were made at five equally spaced points along three cross-stream transects that approximately quartered each experimental unit. We measured average water column velocity with a Marsh–McBirney flowmeter equipped with a top-set rod. We classified the substratum by identifying the size-class covering the most area in a 25-cm-diameter circle centered on each point, using a simplification of the Udden–Wentworth particle size scale: cobble (64–256 mm), coarse gravel (16–64 mm), fine gravel (2–16 mm), and sand (0.06–2 mm).

To monitor water temperature during the experiment, we continuously deployed temperature loggers in 4 of the 18 experimental units and shuffled five additional temperature loggers among the remaining units to provide data for developing equations relating temperatures in the temporarily gauged units to temperatures in the continuously gauged units. The loggers recorded stream temperature at 0.5-h intervals. Strong relationships (all $R^2 > 0.93$) in water temperature between units provided estimates of temperature within units that lacked temperature loggers.

At the conclusion of the experiment on 11–12 September 2002, we first collected two Surber samples (0.09 m^2, 363-μm mesh) of benthic invertebrates at two randomly selected locations within each unit, then we used multiple-pass electrofishing to collect surviving experimental fish and other vertebrates in the units. In an attempt to estimate the quantity of benthos immediately available to fish, we dislodged invertebrates from substrate surfaces during Surber sampling but did not extensively disturb particles below this level. For each stocked fish collected, we recorded FL, wet mass, and PIT tag number, and collected stomach contents using gastric lavage. The length and mass of all other vertebrates were also recorded. We preserved both diet and benthic samples in a 70% solution of ethanol.

We processed benthic and diet samples in the laboratory, identifying invertebrates to genus (if possible) using a stereomicroscope with an ocular micrometer (maximum 70X magnification). We recorded body lengths to the nearest 0.1 mm for up to 50 individuals of every taxon in each sample. If a sample held more than 50 individuals for any one taxon, we measured a sample of 50 and counted the rest. We converted individual prey lengths to estimates of dry mass using taxon-specific relationships provided by K. W. Cummins and M. A. Wilzbach (USGS California Cooperative Fishery Research Unit). Gut fullness was expressed as the ratio of invertebrate dry mass (mg) to the wet mass (g) of fish. We also computed the percentage of overlap (Schoener 1970) in taxonomic composition between the benthic samples and fish diets within experimental units.

Analyses followed a one-way analysis of variance (ANOVA) design with three treatments, but with two exceptions: First, the limited number of outcomes for fish survival in each enclosure (zero, one, or two fish) suggested a categorical approach to data analysis; therefore, we used Fisher's Exact Test to analyze the 3 × 3 table of treatment × survival outcome. Second, the lack of replicate observations on fish for one treatment dictated the use of *t*-tests to compare fish growth and gut fullness in the other two treatments. We used Ryan's *Q* to make pairwise comparisons among treatments in ANOVAs with significant overall results (Day and Quinn 1989).

Results

Physical conditions other than the composition of the substratum were similar among treatments. The means and variances of depth and water velocity within enclosures did not differ significantly among treatments (Figure 1; *P*-values: 0.35–0.81 for one-way ANOVAs with 2, 15 df). Mean water temperature ranged only 12.0–12.4°C during the experiment among all enclosures. Predictably, different substratum size-classes predominated in the three treatments: cobble (69%) in the low-sediment treatment, coarse gravel (64%) in the background-sediment treatment, and fine gravel (54%) in the high-sediment treatment.

Rainbow trout survival and growth differed among treatments. Survival was poor overall, but lower in the high-sediment treatment than in the other two treatments (Fisher's Exact Test: $P < 0.01$). We recovered only 1 of 12 fish stocked in high- sediment enclosures, while survival reached 6 of 12 fish in the background-sediment enclosures and 7 of 12 fish in the low-sediment enclosures (Figure 2). We did not observe

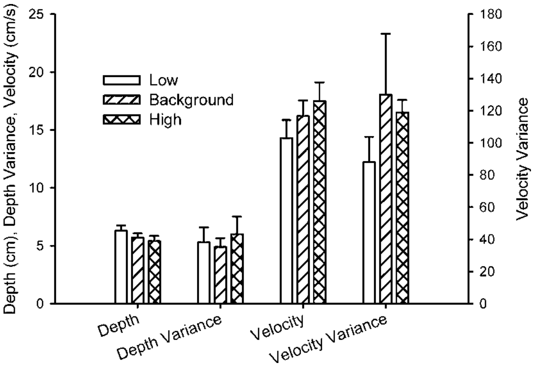

FIGURE 1.—Depth and velocity in enclosed riffles in Jacoby Creek, California, that differed in the amount of deposited fine sediment and cobble embeddedness. (Cobble embeddedness in the three treatments: low, 0%; background, 50%; high, 100%.) Vertical bars indicate 1 SE; $n = 6$ replicates per treatment.

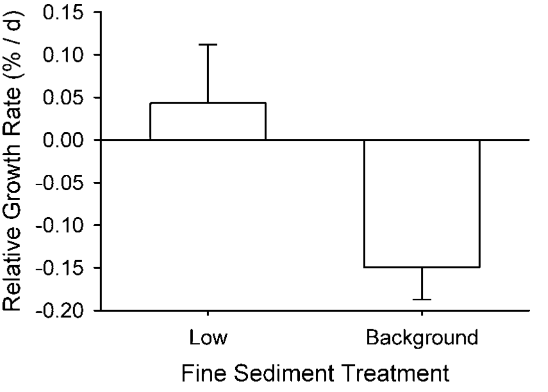

FIGURE 3.—Growth of rainbow trout in enclosed riffles of Jacoby Creek, California, that differed in the amount of deposited fine sediment and cobble embeddedness. Vertical bars indicate 1 SE; $n = 5$ per treatment, because no fish were recovered from one of the six enclosures in each of the background-sediment and low-sediment treatments.

any dead or injured individuals within enclosures, and all enclosure fences remained intact throughout the experiment. While low- and background-sediment treatments yielded similar survival rates, they could be distinguished by differences in growth (Figure 3; $P < 0.05$ for t-test with 8 df). All fish in background-sediment enclosures lost mass while, on average, those in low-sediment treatments gained mass.

Aside from the fish we stocked, the total mass of vertebrates captured at the end of the experiment varied among treatments (Figure 4; $P < 0.01$ for the one-way ANOVA with 2, 15 df). The mean final combined mass of coastal giant salamander larvae, age-0 rainbow trout, and tailed frog adults in the low-sediment treatment exceeded the comparable mean mass in high-sediment treatments by about seven times; Ryan's Q (with $P < 0.05$) distinguished the low-sediment treatment from the other two. Coastal giant salamanders contributed 83% of the total mass of other vertebrates, ranging from an average of 67% in the high-sediment treatment to 89% in the low-sediment treatment. We probably failed to remove all coastal giant salamanders from the enclosures at the start of the experiment. Thus, immigration both before and after installation of the enclosure fences probably influenced the number of coastal giant salamanders collected at the end of the experiment, but in both cases the animals were responding to the sediment conditions produced by experimental manipulation.

The pattern in benthic invertebrate biomass among

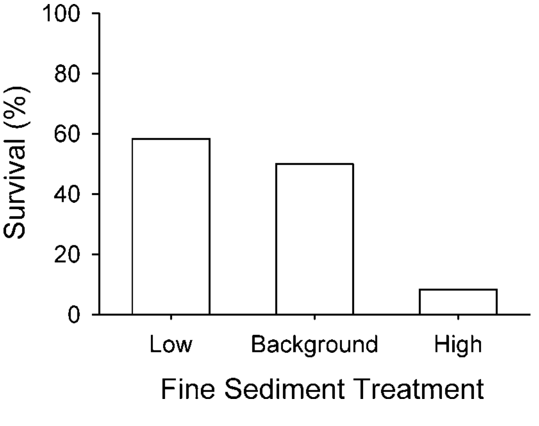

FIGURE 2.—Survival of rainbow trout in enclosed riffles in Jacoby Creek, California, that differed in the amount of deposited fine sediment and cobble embeddedness.

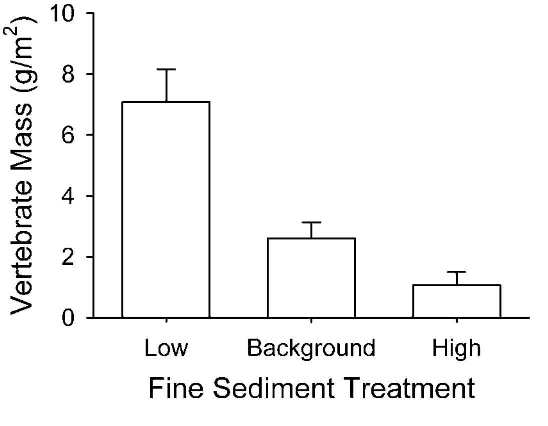

FIGURE 4.—Biomass of vertebrates other than stocked rainbow trout in Jacoby Creek riffle enclosures with three levels of deposited fine sediment, at the end of a 6-week experiment. Vertical bars indicate 1 SE.

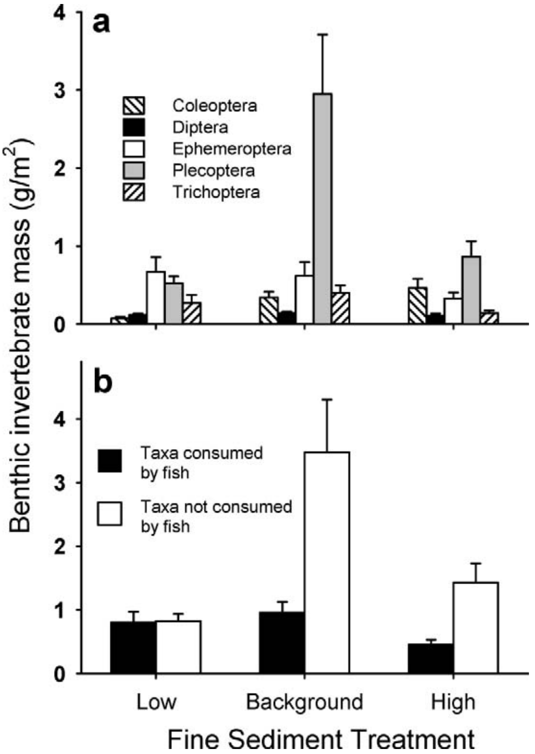

FIGURE 5.—Estimated dry mass of benthic invertebrates on the surface of the substratum of riffle enclosures with different levels of deposited fine sediment and cobble embeddedness in Jacoby Creek, California, categorized by **(a)** taxonomic order and **(b)** taxa found in concurrent rainbow trout diet samples versus other taxa. Vertical bars indicate 1 SE.

sediment treatments differed from the pattern for vertebrates (Figure 5a). The background-sediment treatment yielded the highest biomass of benthic invertebrates, due mainly to the relative abundance of perlid and perlodid stoneflies. The other two treatments yielded samples with similar total biomass, although they differed in taxonomic composition. Because these samples were collected by disturbing only the surface of the substratum, samples from the low-sediment treatment consisted almost entirely of invertebrates collected from unembedded cobbles, while samples from the other treatments included animals collected from both cobble and finer bed material.

Diet samples from the 12 fish collected at the end of the experiment varied in total mass and in taxonomic composition. Mean dry mass of stomach contents/wet mass of fish was about six times higher in the low-sediment treatment than in the background-sediment treatment, but results for the low-sediment treatment were highly variable (*t*-test with 11 df and unequal variance, $P = 0.09$). The taxonomic composition of the stomach samples did not overlap strongly with benthic samples in background-sediment or low-sediment treatments (<20% for both). Considering only those taxa found in the diet samples, low- and background-sediment treatments contained similar biomasses of benthic invertebrates, while the biomass of these taxa was lower in the high-sediment treatment (Figure 5b). Perlid and perlodid stoneflies, which were relatively abundant in the benthic samples from the background-sediment treatment, were not found in the diet samples.

Discussion

One key result from this study was that abundant fine sediment in shallow riffles significantly reduced survival for rainbow trout that are more than 100 mm FL, apparently through greater predation risk as a result of reduced habitat complexity. We hypothesize that stocked fish we did not recover at the end of the experiment were removed by predators, because enclosure fences remained intact, with 10 cm or more of fence buried in fine gravel. Also, prior uses of the same fencing materials and techniques have provided no evidence that fish can escape from the enclosures; one previous experiment of similar length in Jacoby Creek yielded more than 90% recovery of tagged fish within fenced enclosures (Harvey et al. 2006). Other studies have documented low survival for similar-sized fish in shallow stream habitats that lack cover. Harvey and Stewart (1991) observed low survival of cyprinids (80–110 mm total length) in enclosures in a Tennessee stream with habitat similar to the high-sediment treatment in this study. Although this study focused on riffles and the Harvey and Stewart (1991) study on pools, the difference in water velocity probably had little influence on the detectability of fish because high-sediment enclosures in this study had little or no disturbance of the water's surface.

Survival results for the background-sediment treatment in this study (50% over 6 weeks) differ somewhat from those of previous studies that have focused on salmonids in shallow stream habitat, but differences in fish density among studies may account for some of the variation. For example, Rosenfeld and Boss (2001) recovered all cutthroat trout *O. clarkii* (120–160 mm total length) enclosed in riffles during a month-long summer experiment in a British Columbia stream similar in size to Jacoby Creek. However, this study used a higher starting density, and the densities of surviving fish in this experiment and in Rosenfeld and Boss (2001) were very similar (0.15–0.17 fish/m^2). In a 9-week study in Jacoby Creek (Harvey et al. 2005), survival of rainbow trout (mean FL = 118 mm) averaged 70% in 22 enclosures with mean depth less than 10 cm, but that study used natural fish density, which was lower than the starting densities in the present experiment. In addition to the possible influence of variation in fish density, survival estimates from experiments like these are likely to vary because of low resolution due to small numbers of fish, limited replication, and variability in survival due to spatio-temporal variation in predator density (Boss and Richardson 2002).

This study's background-sediment results for fish growth parallel results from Rosenfeld and Boss (2001) which indicated that gravel and sand riffles in small streams do not provide energetically favorable habitat for salmonids older than age 0. The negative growth rates of rainbow trout (105–130 mm FL) in the background-sediment enclosures in Jacoby Creek closely paralleled growth rates of slightly larger cutthroat trout at the same final density in Husdon Creek, British Columbia, in enclosures offering similar physical conditions (Rosenfeld and Boss 2001).

In contrast to background riffle conditions, the low-sediment riffles in this experiment provided opportunities for weight gain by rainbow trout over 100 mm at a density (about 0.16 fish/m^2) similar to that found in Jacoby Creek pools. Bolliet et al. (2005) hypothesized that the availability of microhabitat providing relatively efficient drift feeding contributed to their finding of lower growth by young of the year brown trout *Salmo trutta* in embedded versus unembedded sections of an artificial stream. While our physical measurements suggested similarity among treatments in mean depth and velocity, low-sediment riffles may have offered a few microhabitats among unembedded cobbles that provided both feeding opportunities and cover exceeding those of any microhabitats available in background- and high-sediment enclosures. This possibility may link to the similarity in survival for fish in the low- and background-sediment treatments in this study: survival in the background-sediment treatments may have come at the cost of growth as fish spent time in refuges that did not provide feeding opportunities. Fish in either the low-sediment or background-sediment enclosures were rarely detected by streamside observers.

One related mechanism that might have contributed to mass gain by fish in low-sediment enclosures is the influence of substrate-mediated stress on metabolic rates (Fischer 2000). While Fischer (2000) measured this effect in a benthic species (burbot), similar effects might be expected in any species for which substrate strongly influences the risk of predation and predator activity is not easily assessed. Millidine et al. (2006) recently documented a relationship between metabolic rate and habitat features for a salmonid: a 30% reduction in the metabolic rate of juvenile Atlantic salmon *S. salar* provided with overhead hiding cover.

Previous research by Suttle et al. (2004) has linked salmonid growth rates to the effects of deposited sediment on food availability. That study demonstrated lower growth for young of the year steelhead and lower densities of benthic invertebrates classified as vulnerable to fish predation as deposited sediment increased in experimental channels placed in stream pools. In contrast to that experimental setting, the enclosures in this study spanned the wetted stream channel in areas of high water velocity. Therefore, we assume immigration strongly influenced prey densities in both the drift and the benthos. Considering the influence of immigration and the potential for movements by benthic invertebrates between lower levels of the substratum and the level we sampled, we do not consider the results of our benthic collections a clear indication of greater availability or production of benthic invertebrates in the background-sediment treatment compared with the low-sediment treatment. More information on the effects of elevated fine-sediment transport and storage regimes on overall food availability for fish at the whole-stream scale would clearly be valuable. Much of the work on links between stream invertebrates and fine sediment has focused on response variables such as species richness, which are not readily translated to food availability for fish.

Suttle et al. (2004) also documented higher rates of aggressive interactions for young of the year steelhead when embeddedness increases into the range of 60–100%. This mechanism probably did not strongly influence our results, because we used large fish at relatively high densities in treatments that provided few high-quality microhabitats: interactions between individuals would be expected in all three treatments. In fact, the experiment provides evidence of density-dependent growth in the low-sediment treatment that might be attributable in part to aggressive interactions, because all three fish in low-sediment enclosures with one remaining fish gained weight while those in the two low-sediment enclosures with two remaining fish lost mass.

This study adds to existing experimental evidence of the detrimental effects of deposited fine sediment on juvenile salmonid growth in streams (Suttle et al. 2004; Bolliet et al. 2005) and also suggests that deposited fine sediment can lower survival by enhancing predation risk. The low-sediment treatments in this experiment appeared to offer relatively large juvenile trout energetically favorable microhabitats that also provided relatively low risk of predation. Evaluation of the potential effects of deposited fine sediment on salmonid population dynamics must also incorporate mechanisms that operate on other life stages and in other seasons. A complete analysis of how elevated input of fine sediment affects population dynamics must also include the effects of suspended sediment and bed load transport, such as the effect of suspended sediment on reactive distance to drifting prey and on predation risk, and the effect of bed load transport on incubation success and production of benthic invertebrates. Combined with observations linking unembedded substrates in streams to higher densities of various other vertebrates, such as salamanders observed in this and other studies (Parker 1991; Welsh and Ollivier 1998) and benthic fish (Jowett and Boustead 2001; White and Harvey 2001), these results indicate that elevated fine-sediment inputs will often deserve consideration by resource managers. In some watersheds, reducing the levels of stored fine sediment in small stream channels may substantially increase the watershed-scale total of productive habitat available to both fish and other vertebrates.

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Cattle grazing has varying impacts on stream-channel erosion in oak woodlands

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Abstract

We conducted a 5-year study on the impact of grazing on stream-channel bare ground and erosion, and a 3-year study of cattle-trail erosion on intermittent stream channels draining grazed oak-woodland watersheds. While the concentration of cattle along stream banks during the dry season resulted in a significant increase in bare ground, we were unable to detect stream-bank erosion resulting from any of the grazing treatments applied. However, we did find that cattle trails are an important mode of sediment transport into stream channels. While cattle trails are common on grazed rangeland, excessive trailing often indicates that stock watering points are too far apart.

Keywords: cattle, grazing, erosion, sediment, cattle trails, water quality, stream, Sierra Nevada foothills, oak woodland

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RESEARCH ARTICLE

Cattle grazing has varying impacts on stream-channel erosion in oak woodlands

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There are 16 million acres of grazed rangeland in California's foothills. Much of the state's water flows through these areas and can be polluted with sediment from livestock grazing. Additional troughs and watering ponds can improve cattle distribution and protect water quality.

We conducted a 5-year study on the impact of grazing on stream-channel bare ground and erosion, and a 3-year study of cattle-trail erosion on intermittent stream channels draining grazed oak-woodland watersheds. While the concentration of cattle along stream banks during the dry season resulted in a significant increase in bare ground, we were unable to detect stream-bank erosion resulting from any of the grazing treatments applied. However, we did find that cattle trails are an important mode of sediment transport into stream channels. While cattle trails are common on grazed rangeland, excessive trailing often indicates that stock watering points are too far apart.

Most of California's surface water flows through the state's 16 million acres (6.4 million hectares) of annual rangelands in foothills. These foothill rangelands are drained by intermittent streams, which begin to flow following adequate rainfall during the October-to-May rainy season (Lewis et al. 2000). In dry years many intermittent streams in these rangelands do not flow. Sediment is a nonpoint source pollutant of concern in these surface waters.

Erosion and resulting sedimentation is a natural process that is often accelerated by human activities such as mining, construction, roads, timber harvest, crop production and livestock grazing. Excessive sedimentation clouds water, which reduces the amount of sunlight reaching aquatic plants; covers fish spawning areas and food supplies; and clogs the gills of fish. In addition, other pollutants such as phosphorus, pathogens and heavy metals are often attached to the soil particles and end up in downstream water bodies. Consequently, landowners and managers are planning and implementing erosion-control practices as part of regulatory requirements to reduce sediment loading in coastal streams and rivers (see page 134, 149). Because grazing is the dominant use of foothill rangelands it is frequently implicated as a source of sediment. However, Lewis et al. (2001) found that roads were a larger sediment source on ranches than grazing.

Improper livestock use can increase stream-bank erosion and sedimentation by changing, reducing or eliminating the vegetation that borders streams (Kaufman and Krueger 1984). Several studies have implicated livestock-induced stream-bank erosion, which leads to channel downcutting or widening (Kaufman and Krueger 1984; Hall and Bryant 1995; Sierra Nevada Ecosystem Project 1996).

In 1994, we began to monitor sediment delivery at the bottom of a 342-acre grazed oak-woodland watershed in Madera County. While not continuously grazed, this watershed was grazed several times each year, leaving 600 to 800 pounds of residual dry matter per acre in the fall, which meets UC guidelines (Bartolome et al. 2002).

We found that little sediment was suspended in water samples collected near the bottom of the watershed. On further investigation we found that most sediment moved along the bottom of the channel as bedload during storm runoff, increasing with flow and settling in low-gradient reaches of the stream channel (George et al. 2002). At this site, grazing management generally maintained adequate residual dry matter (Bartolome et al. 2002) on hill slopes; in addition, earlier studies by the U.S. Department of Agriculture's Natural Resources Conservation Service (USDA/NRCS) had documented little overland flow and resulting surface erosion. Therefore, we suspected that the source of most of the bedload sediment was the stream channel and the adja-

Livestock grazing impacts, such as stream-channel erosion and impaired water quality, can often be addressed with management practices that alter livestock distribution on the landscape.

cent variable-source area that merges with the stream channel. We initiated two studies to determine the influence of grazing cattle on erosion in or near stream channels. The stream-channel study documented changes in stream-channel groundcover and stream-bank erosion due to different seasons and intensities of grazing. The cattle-trail study compared sediment movement on cattle trails and adjacent vegetated surfaces on the slopes of the variable-source area that merges with the stream channel. The purpose was to determine if cattle grazing and trampling contributed to the bedload sediment associated with intermittent stream channels in the central Sierra Nevada foothills.

San Joaquin Experimental Range

These studies were conducted on the San Joaquin Experimental Range (SJER) in Madera County in oak woodlands, which are dominated by coarse-loamy granitic soils of the Ahwahnee series (fig. 1). Numerous intermittent stream channels, most of which are tributaries to Cottonwood Creek, dissect the station. Cottonwood Creek is a fourth-order stream that drains into the San Joaquin River just below Friant Dam. During this study, stream flow began in early January following 11 to 14 inches (270 to 360 centimeters) of rainfall from October to December. While granite rocks, oak trees and other woody vegetation provide some stability, the majority of the stream banks are vegetated by shallow-rooted annual grasses and forbs. The stream bottoms are predominantly sand with some granitic cobble, rock and boulders. The channels are 2 to 10 feet wide, 1 to 3.3 feet deep and bedrock-controlled in many reaches. Three intermittent tributaries to Cottonwood Creek were selected for study at SJER. The study reaches (stream segments) are low-gradient with less than 2% slope and are Rosgen Class B5 (Rosgen 1996). Stream channels 1, 2 and 3 are 1 to 2 miles apart and at an elevation of 900 to 1,348 feet.

Glossary

Animal unit: Often defined as a cow and her calf, or as a total of 1,000 pounds of body weight; **animal unit month:** amount of feed an animal unit will consume in 1 month.

Bankfull depth: Depth where channel is completely filled and water begins to spill onto adjacent floodplain.

Bedload: Heavier particles such as sand, gravel and rock that are too heavy to be suspended in the water column but that roll along the channel bottom during high flow events.

Channel bank: The often-steep component of a stream channel extending from the channel bottom to the top of the bank.

Nonpoint source pollution: Diffuse discharges of waste throughout the natural environment such as from mining, urban runoff, agriculture and logging.

Residual dry matter: Litter (old plant material) left standing or on the ground at the beginning of a new growing season (Bartolome et al. 2002).

Rosgen Stream Classification: A method of classifying stream channels based on channel gradient and confinement (Rosgen 1996).

Stock density: Number of animals per unit area at any point in time.

Stocking rate: The number of specific kinds and classes of animals grazing a unit of land for a specified time period.

Stream-channel down-cutting: Increase in channel depth due to erosion of channel bed.

Stream-channel widening: Increase in channel width due to erosion of stream banks.

Stream reach: Segment of a stream.

Undercut stream bank: A bank that has had its base cut away by the action of stream flow along natural overhangs in the stream.

Variable-source area: Runoff-generating saturated soils that vary in their extent with storm intensity and length.

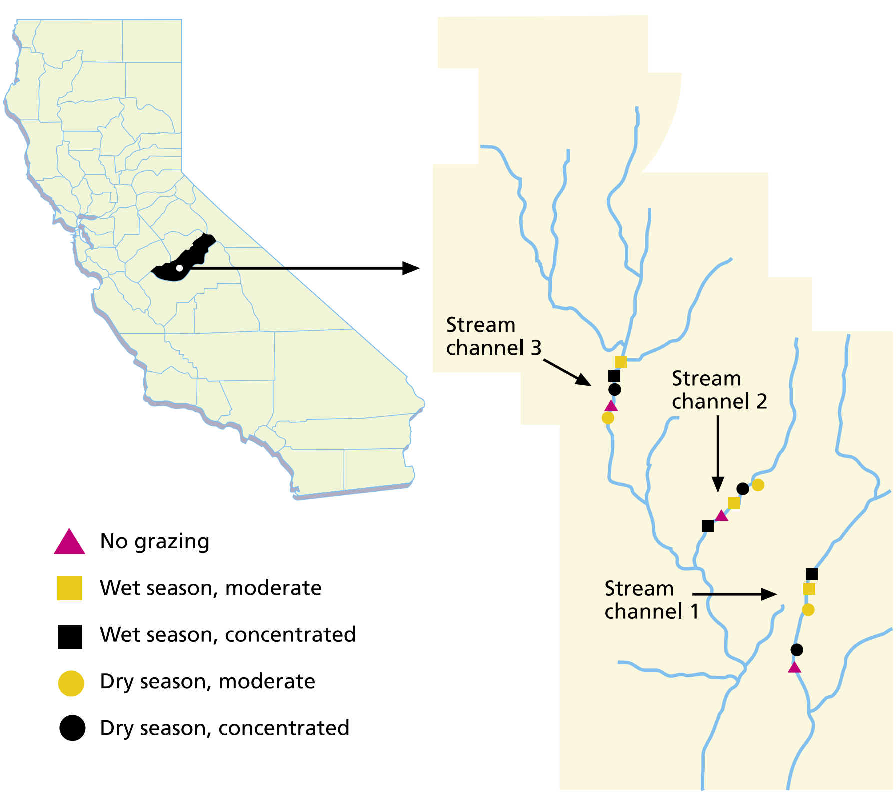


Fig. 1. Location of Madera County study site, and treatments along stream channels at the U.S. Forest Service San Joaquin Experimental Range.

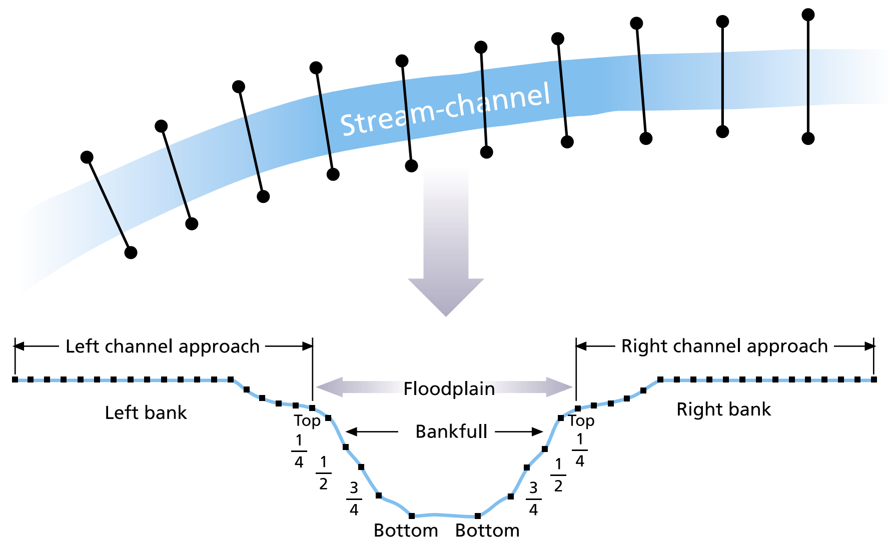


Fig. 2. Diagram of 10 stream-channel cross-section transects.

Grazing impacts on stream channels

Maintaining adequate vegetative groundcover is the first defense against soil erosion. Thus, decreasing groundcover or increasing bare ground is an indication of increasing risk of erosion. Increasing stream-channel width and depth are indicators of stream-channel erosion. The objective of our study on stream channels was to determine changes in bare ground along the channel banks and changes in channel width and depth in response to two seasons (wet and dry) and three grazing intensities (no grazing, moderate and concentrated).

Grazing treatments. Five grazing treatments were applied to five randomly selected 1-acre pastures established for a 5-year study along each of three intermittent streams (fig. 1). The treatments were:

- No grazing.
- Wet season, moderate grazing (stubble height = 2 to 3 inches).
- Wet season, concentrated grazing (stubble height < 2 inches).
- Dry season, moderate grazing (stubble height = 2 to 3 inches).
- Dry season, concentrated grazing (stubble height < 2 inches).

The concentrated grazing treatments were designed to apply the extremely heavy use often associated with a feed or watering site. Each grazing treatment was applied to the same pastures during 4 consecutive years.

Dry-season grazing treatments were applied between July 1 and Oct. 1, a period of little or no rainfall. Wet-season treatments were applied while the soil was moist and maintained until the end of the growing season. Typically, the wet season begins in late October or early November and ends by May 1. This period includes the slow winter growth period and all of the rapid spring growth period of the growing season (George et al. 2001). The moderate and concentrated grazing treatments were stocked at about 1.7 acre per animal unit month. Cooked molasses protein supplements and mineral blocks were placed along the stream banks in the pastures treated with the concentrated grazing treatments and additional animals were added to achieve the target stubble height associated with feeding and watering sites. For the concentrated grazing treatments, instantaneous stock densities equivalent to 100 cows per acre were occasionally achieved but not maintained within the corridor delineated by the cross-section transects.

Measuring stream-channel erosion. Stream-channel measurements were recorded during the first week of June at the beginning of the dry season starting with a baseline year in June 1994. Channel cross-sections were measured using methods outlined by Bauer and Burton (1993). For each stream reach, 10 permanent cross-section transects, 20 to 30 feet long, were placed perpendicular to the stream channel at a distance of 1 to 1.5 times the channel width apart (fig. 2). The transects were marked with permanent stakes and referenced to a permanent benchmark. Stream-channel elevation was determined every 6 inches (15 centimeters) along the transect using a stretched tape, laser level and stadia rod. For each transect, we measured width at bankfull, distance from the left permanent stake to both right and left bank at bankfull height, depth every 6 inches, and maximum depth. Cross-sectional area, channel average depth and width-to-depth ratio were calculated. Pasture averages for each morphological parameter were calculated from the 10 transects in each pasture. Cross-sectional area of the channel was determined using bankfull elevations following the methods of Rosgen (1996). Elevation and position readings of the permanent end-stakes were checked with benchmark elevations each year.

Measuring groundcover. Groundcover was determined using the line-point transect method (Bonham 1989) along the same transects used to survey elevations for stream-channel morphology (Bauer and Burton 1993). Bare ground was calculated by subtracting the percentage of groundcover from 100.

To separate cattle impacts on the channel bank from the adjacent floodplain and uplands, the line-point transect was divided into two sections on each side of the channel (fig. 2). The channel bank is the often-steep component of a stream channel extending from the channel bottom to the top of the bank. This section includes the point often defined as "bankfull" (Rosgen 1996). Bare ground was determined at five points along the left and right channel banks: top of bank, bottom of bank, and one-quarter, one-half and three-quarters of the distance from the top of the bank to the channel bottom (fig. 2). The stream-channel approach extended 9 feet (270 centimeters) from the top of the bank and included varying portions of the floodplain (Rosgen 1996) and uplands depending on channel morphology along the channel reach. Bare ground was determined from the top of the bank to 3 feet at 6-inch intervals,

Fall comparison of stream channel in concentrated grazing treatment, *upper,* and no grazing, *lower*.

Comparison of sediment trap in a vegetated area, *upper,* and sediment trap in an adjacent trail, *lower*.

and at 1-foot intervals from 3 to 9 feet along the stream cross-section transect.

Bare ground, channel effects

There were significant ($P < 0.001$) increases in bare ground on the channel bank and approach due to dry-season concentrated grazing when compared to the ungrazed control (fig. 3). Bare ground for the other grazing treatments was not significantly different from the ungrazed control or the dry-season concentrated treatment. These results indicate that practices causing cattle to congregate near stream channels during the dry season can significantly decrease groundcover, which protects the soil surface from erosion. Because these streams are dry during the summer they are generally not attractive to cattle. Therefore, the intensity of grazing and trampling resulting from the concentrated grazing treatment is unlikely to occur under proper stocking rates and grazing practices.

No significant ($P > 0.05$) stream-bank erosion was detected when each stream-channel measurement (width, distance to right and left bank, maximum depth, mean depth, cross-sectional area or width-to-depth ratio) for each of the five grazing treatments was averaged across all years. Stream-channel depth changed significantly ($P < 0.05$) from year to year, reflecting the seasonal and annual movement of bedload along the stream-channel bottom. The greatest between-year change was from 1996 to 1997 — an above-average rainfall year — resulting in higher-than-normal flow events that scoured bedload sediment from the channel, increasing stream-channel depth.

In their review, Trimble and Mendel (1995) found conflicting reports on the relationship between grazing along stream channels and sediment loss from stream banks. Several of these studies reported that increased channel width was the result of sloughing of undercut banks. The stream-channel banks in this study were not undercut, and could not achieve this form under any grazing scheme due to the sandy soil type and dominance of shallow-rooted annual vegetation.

We observed grazing and trampling along the stream-channel bank by cattle in the treated pastures, yet detected no change in channel width at bankfull. Fine-textured and wet stream-bank soils have been shown to be a factor in vulnerability to erosion (Clary and Webster 1990; Wolman 1959; Hooke 1979; Marlow and Pogacnik 1985; Marlow et al. 1987). The stream-bank soils in our study may be less likely to erode because they are well-drained coarse sands that have a low water-holding capacity. Trimble and Mendel (1995) suggested that watersheds subjected to high-intensity, long-duration storms generating high stream discharges were more vulnerable to stream-bank erosion than watersheds that receive relatively

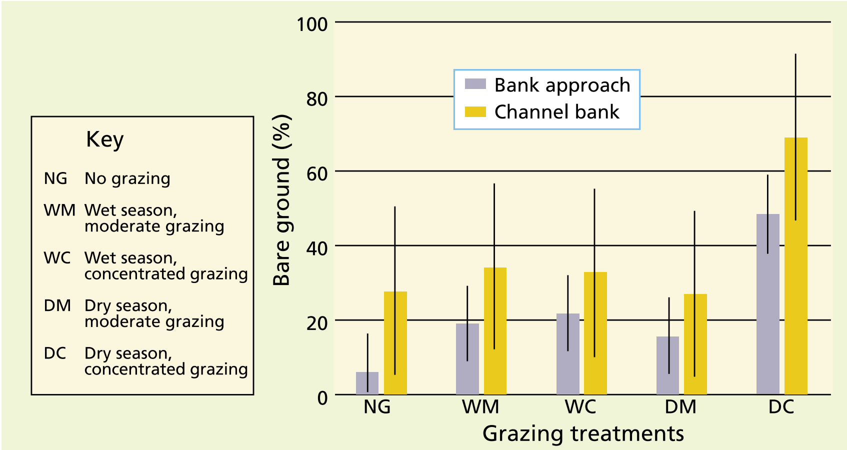


Fig. 3. Comparison of effects of fall treatment on bare ground (%) for bank approach and channel bank. Bars indicate 95% confidence level.

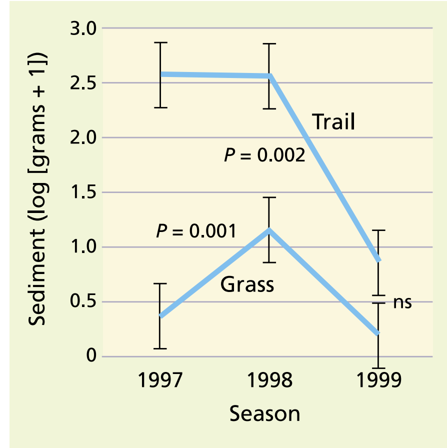


Fig. 4. Comparison of sediment from trails and adjacent grass-covered surfaces for three water years.

equitable flow from snowmelt. During our study one or more high stream discharges occurred each year, lasting for only a few hours during and following a storm. However, such single storm events were not intense or sustained enough to lead to significant erosion. Lack of high-intensity rainfall and runoff early in the rainy season may reduce stream-bank erosion. While intense grazing and trampling can leave unvegetated loose soil at the beginning of the rainy season, low-intensity rainfall, characteristic of the early rainy season, results in germination and seedling establishment that stabilizes grazed and trampled soil surfaces before periods of more intense rainfall begin.

Channel deepening was detected in the control treatments, indicating a loss of bedload sediment from the control reaches. Treatment randomization within each stream (block) resulted in the controls being placed at the lowest or next-to-lowest pasture in the sequence of five pastures along each stream. One might expect channel depth in the controls to become shallower if they were influenced by the delivery of sediment from upstream grazed treatments, but not for the channel to deepen. While there was no significant change in channel width, there was a trend toward channel narrowing that may have resulted in increased stream power, which could have eroded bedload sediment in the control pastures.

Another interpretation of the stream-channel study is that the cross-section method frequently recommended for detecting stream-channel erosion is not sensitive enough to detect the scale of erosion induced by the activity of grazing cattle. Furthermore, erosion resulting from episodic high-flow events may mask grazing-induced erosion. More refined methods of detecting erosion and longer term studies of grazed and ungrazed stream reaches may be the only means of separating grazing-induced erosion from other sources (such as natural and rodent-caused erosion, and roads).

In summary, the risk of grazing-induced stream-bank erosion in the granitic soils of the southern Sierra Nevada foothills is low because: (1) stream banks are not vertical, lowering the risk of bank undercutting and eventual sloughing; (2) the soils lack the fine texture and high water-holding capacity often associated with stream-bank erosion; (3) these southern Sierra Nevada foothills are not subject to long-duration, high-intensity storms; and (4) bare soil surfaces at the beginning of the rainy season are stabilized by newly established seedlings before periods of intense rainfall begin.

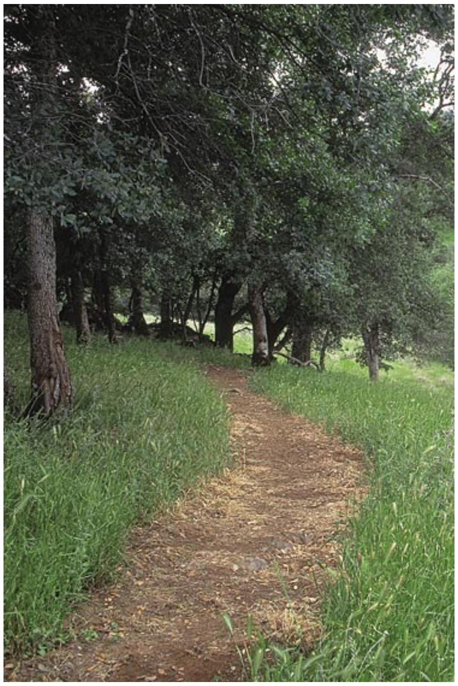
Sediment and cattle trails

During the 1997, 1998 and 1999 water years, sediment traps (Wells and Wohlgemuth 1987) were placed in pairs at several locations along stream channel 2 (fig. 1) that drains the research watershed at the San Joaquin Experimental Range. Traps were sheet-metal boxes open on one side to catch sediment. One of each pair of traps was placed in a cattle trail near the point where the trail crosses the stream channel. The second sediment trap of each pair was placed in well-vegetated areas of similar slope and slope-length adjacent to the trap in the trail. Throughout the rainy season the sediment traps were emptied as needed during and following storms. Sediment samples were dried and weighed. ANOVA was used to separate treatment and year differences.

Sediment transport was significantly greater in cattle trails than in vegetated areas in the rainfall years ending in 1997 and 1998 (fig. 4). There was no significant difference in 1999. In 1997 and 1998 there was sufficient rainfall to generate measurable runoff, and the intermittent streams began flowing in January of those two rainfall years. Rainfall in 1999 was low, resulting in little runoff and sediment movement in cattle trails. While cattle-trail crossings affect a very small total of the channel length within the watershed, the results of this study suggest that trails can be an important, management-caused conduit of sediment from the variable-source area to the stream channel. Following a major runoff event, we observed diversion of intermittent stream-channel flow into a cattle trail forming a new channel.

Cattle prefer to travel along established trails. In steep terrain, trails provide routes from water sources to preferred foraging sites. Trailing increases as the distance between foraging sites and water sources increases. In steep terrain, one trail may cross

Suzanne Paisley



Cattle trails can aid the flow of sediment into streams. By reducing the distance between foraging and watering sites, the formation of cattle trails can be reduced.

several stream channels. Several trails are common in the steep terrain of foothill rangelands, resulting in many trail crossings of each stream channel. Regular trampling by livestock keeps these trails devoid of vegetation throughout the year and reduces the infiltration rate, resulting in increased surface runoff along trails, especially along the downhill approach to stream crossings. During the dry season cattle trampling loosens surface soil, providing a ready source of sediment during the rainy season. The trails become a conduit for surface runoff and a source of sediment.

Reducing stream-channel impacts

To limit cattle impacts on stream channels, especially on public lands, mitigations such as reduced stocking rates, grazing lease termination, and fencing of streams and riparian areas have frequently been implemented. These practices can devastate the economic viability of range livestock enterprises, reducing their competitive ability and adversely affecting the economies of rural communities. Furthermore, livestock exclusion limits our ability to use grazing to manage wildlife habitat, fire fuel loads and weed infestations. Livestock grazing impacts, such as stream-channel erosion and impaired water quality, can often be addressed with management practices that alter livestock distribution on the landscape. While lease termination and fencing are certain methods of reducing livestock impacts on stream channels and water quality, less restrictive management changes such as strategic placement of water sources and supplemental feeding sites away from critical areas have been shown to reduce livestock grazing impacts (Frost et al. 1989).

Trailing is reduced when the distance between foraging sites and water sources is reduced. Adequate stock-water development can lead to improved control of cattle distribution. Strategic placement of fencing may also reduce trailing. Sediment and other livestock-generated, nonpoint source pollution sources could be substantially reduced with increased water sources on foothill ranches. These rangeland improvements should receive high priority in the allocation of agency conservation and pollution-control funding.

Researchers at UC Davis, USDA and other Western universities are currently studying the effectiveness of traditional distribution practices for attracting livestock away from environmentally critical areas or into areas to be grazed for weed control. The results of these studies will be used to fine-tune our knowledge of how livestock use Western landscapes and to improve our ability to predict the best placement of stock water and supplement sites.

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Ecological Costs of Livestock Grazing in Western North America

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Abstract: *Livestock grazing is the most widespread land management practice in western North America. Seventy percent of the western United States is grazed, including wilderness areas, wildlife refuges, national forests, and even some national parks. The ecological costs of this nearly ubiquitous form of land use can be dramatic. Examples of such costs include loss of biodiversity; lowering of population densities for a wide variety of taxa; disruption of ecosystem functions, including nutrient cycling and succession; change in community organization; and change in the physical characteristics of both terrestrial and aquatic habitats. Because livestock congregate in riparian ecosystems, which are among the biologically richest habitats in arid and semiarid regions, the ecological costs of grazing are magnified in these sites. Range science has traditionally been laden with economic assumptions favoring resource use. Conservation biologists are encouraged to contribute to the ongoing social and scientific dialogue on grazing issues.*

Costos ecológicos del pastoreo de ganado en el oeste de Estados Unidos

Resumen: *El pastoreo de ganado es la práctica de manejo de la tierra más ampliamente utilizada en el oeste de Norte América. El setenta por ciento del oeste de Estados Unidos se utiliza para pastoreo, incluyendo áreas silvestres, refugios de vida silvestre, bosques nacionales e inclusive algunos parques nacionales. El costo ecológico de esta forma ubicua de uso de la tierra puede ser dramático. Ejemplos de este costo incluyen pérdida de la biodiversidad; decrecimiento de las densidades de población para una amplia variedad de taxones; alteraciones en las funciones del ecosistem, incluyendo ciclos de nutrientes y sucesiones; cambios en la organización de la comunidad y cambios en las características físicas de hábitas terrestres y acuáticos. Dado que el ganado se congrega en ecosistemas ribereños, los cuales están entre los hábitas biológicamente más ricos dentro de las regiones áridas y semi-áridas, los costos ecológicos del pastoreo se magnifican en estos sitios. Tradicionalmente, la ciencia de pastizales, ha estado cargada de suposiciones económicas que favorecen el uso del recurso. Se alienta a los biólogos conservacionistas a contribuir al diálogo social y científico en los problemas del pastoreo.*

Introduction

Aldo Leopold (1953) once said that to be an ecologist is to live "alone in a world of wounds." The spectacular groundswell of interest in conservation biology is heartening evidence that we no longer work alone. But what about a world of wounds? The wounding of natural processes accelerates, but some wounds are more conspicuous than others. Recognizing a clearcut forest is easy, but it often takes a trained eye to comprehend damage to rangelands. The destruction caused by livestock grazing is so pervasive and has existed for so long that it frequently goes unnoticed. Livestock grazing has re-

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ceived far less attention from conservation biologists than its widespread influence would suggest is appropriate. When I recently surveyed the first six volumes of this journal, for example, I found almost three times as many articles on deforestation as on grazing-related topics.

Livestock grazing is the most widespread influence on native ecosystems of western North America (Wagner 1978; Crumpacker 1984). Grazing by livestock, primarily cattle, is nearly ubiquitous throughout this region. Approximately 70% of the 11 western states of the United States (Montana, Wyoming, Colorado, New Mexico, and westward) is grazed by livestock (Council for Agricultural Science and Technology 1974; Longhurst et al. 1984; Crumpacker 1984), including a broad diversity of ecosystem types and virtually all types of land management designations. Grazing occurs in creosotebush deserts, blackbrush deserts, slickrock mesas, sagebrush flats, pinyon-juniper woodlands, chaparral, ponderosa pine forests, and alpine meadows above timberline.

Grazing occurs on the majority of federal lands in the West, including most of the domains of the U.S. Bureau of Land Management (BLM) and the U.S. Forest Service, as well as in many national wildlife refuges, federal wilderness areas, and even some national parks. In 16 western states, approximately 165 million acres of BLM land and 103 million acres of Forest Service land are grazed by 7 million head of livestock, primarily cattle (U.S. General Accounting Office 1988*a*). Of the BLM lands in these states, 94% is grazed. Of federal wilderness areas, 35% have active livestock grazing allotments (Reed et al. 1989; this figure is from a nationwide survey—the percentage for the West is probably higher). Urbanized areas, some dense coniferous forests, and a few rock-and-ice peaks are about all that is free from the influence of livestock. Given the ubiquity of livestock, it behooves us to understand the consequences of its presence on the Western landscape.

Understanding the influence of domestic livestock upon native ecosystems is a problematic process. Ascertaining the potential natural vegetation of most Western ecosystems is difficult because ungrazed land is extremely rare. Ecologists have gained insight into the effects of grazing primarily in three ways: (1) Historic records provide perspective on the dramatic changes that have transpired since the introduction of livestock to the West (see Cooper 1960). As Hastings (1959) pointed out, however, one must be cautious in interpreting historical records, due to the subjectivity of different observers. Historic photographs have also been used in an attempt to recreate an ecological baseline (see Hastings & Turner 1965); Bahre (1991) reviewed the necessary cautions in interpreting historic photographs. (2) Areas excluded from grazing through serendipity, such as isolated mesa tops, provide startling contrast to adjacent areas that have been continuously grazed (see Rummell 1951). (3) Areas that intentionally exclude livestock (exclosures) provide a before-grazing and after-grazing comparison. Exclosures can be monitored as they recover from the effects of grazing and can be compared with adjacent grazed sites. Almost all exclosures share two characteristics: (1) their areas are usually quite small (Bock et al. 1993*a*), often less than 50 ha; and (2) they have been grazed prior to exclosure. In other words, very few studies of truly ungrazed landscapes exist. Most recreational impact studies concur that the original impact upon a pristine site is the most severe (Cole 1981; Cole & Marion 1986); thus, exclosure studies probably underestimate the true extent of grazing effects because they cannot monitor the most drastic damage, which occurred long ago. In addition, virtually all exclosure studies examine areas too small to encompass landscape-level diversity. In summary, we lack a clear ecological benchmark for determining the effects of grazing.

Attempts to discern grazing effects are also hampered by the difficulty in distinguishing between different range management practices. Management variables include grazing intensity ("stocking rate"), livestock species, seasonality of grazing, and degree of active management, such as movement of livestock between pastures. Unfortunately, the management history of many sites is unknown. Many studies do not describe grazing intensity (see, for example, Glinski 1977; Reynolds & Trost 1980; Crouch 1982). Furthermore, standardized terminology is lacking for different grazing intensities. Relative terms, such as "heavy," "moderate," and "light" grazing, may be undefined (see Jeffries & Klopatek 1987) or qualitatively defined in very different ways. Among the criteria used are presence of livestock, presence of trails, range condition (see Jones 1981), and amount of herbage remaining after a grazing season (see Welch et al. 1991). Studies that have quantified grazing intensity have do so inconsistently. For example, two studies (Mosconi & Hutto 1982; Baker & Guthery 1990) analyzing the effect of "heavy" grazing differed in their definition by a factor of seven. The much-used term "overgrazing" is wrought with controversy and lack of clarity; even specific discussions of overgrazing fail to define it (see Menke & Bradford 1992). This rudimentary state of knowledge interferes with analysis of the role of different grazing practices on biodiversity.

Available evidence indicates that livestock grazing has profound ecological costs. Autecological, synecological, and geomorphological studies have confirmed that native ecosystems pay a steep price for the presence of livestock. Three primary attributes of ecosystems have been elucidated: composition, function, and structure (Franklin et al. 1981). Livestock grazing has a profound impact on all three. The ecological costs of livestock grazing can be summarized as follows:

(1) *Alteration of species composition of communities,* including decreases in density and biomass of individual species, reduction of species richness, and changing community organization.
(2) *Disruption of ecosystem functioning,* including interference in nutrient cycling and ecological succession.
(3) *Alteration of ecosystem structure,* including changing vegetation stratification, contributing to soil erosion, and decreasing availability of water to biotic communities.

Alteration of Species Composition of Communities

That the introduction of a large-bodied herbivore should have dramatic effects on the species composition of plant communities in arid and semiarid regions should not be surprising. Congressional investigation into rangeland conditions on BLM and Forest Service lands showed that over 50% of public rangelands administered by these two agencies were in "poor" or "fair" condition, meaning that less than half the range was 50% similar to the presumed climax community (U.S. General Accounting Office 1988*a*, 1991*a*). Grazing affects the species composition of plant communities in essentially two ways: (1) active selection by herbivores for or against a specific plant taxon, and (2) differential vulnerability of plant taxa to grazing (Szaro 1989). Decreases in density of native plant species and diversity of native plant communities as a result of livestock grazing activity have been observed in a wide variety of western ecosystems (Table 1).

Grazing also can exert a great impact on animal populations, usually due to indirect effects on habitat structure and prey availability (Wagner 1978; Jones 1981; Mosconi & Hutto 1982; Szaro et al. 1985; Quinn & Walgenbach 1990). The deleterious effects of grazing have been observed in all vertebrate classes (Table 2). The response of native wildlife to grazing varies by habitat. Bock et al. (1993*b*) reviewed the effect of grazing on Neotropical migratory landbirds in three ecosystem types and found an increasingly negative effect on abundances of bird species in grassland, riparian woodland, and intermountain shrubsteppe (almost equal numbers of species with positive and negative responses to grazing in grassland; six times as many with negative as positive responses in shrubsteppe). Due to their mobility and visual orientation, birds may be better able to cope with grazed landscapes than mammals are (Bock et al. 1984). Platts (1979, 1981) reviewed the interaction of biological and geomorphological factors that degrade fish habitat.

The relationship of grazing to insect populations is unclear (Table 3). Studies of grasshoppers (Acrididae) on rangelands have yielded contradictory results: some report an increase in grasshopper densities on heavily grazed lands, and others report a decrease (summarized in Welch et al. 1991). Recent research has clarified that duration of grazing, seasonal differences in plant and insect communities, and plant community architecture

Table 1. Deleterious effects of livestock grazing on plant communities in western North America.

Habitat	*Location*	*Effect*	*Authority*
Sonoran Desertscrub	Arizona	Perennial grasses and *Krameria* (palatable shrub) showed dramatic density decreases with grazing	Blydenstein et al. (1957)
Mojave Desertscrub	California	60% reduction in above-ground biomass of annuals, 16–29% decrease in cover of perennial shrubs with grazing	Webb & Stielstra (1979)
Sagebrush Desert	Idaho	Grazed site had ⅓ species richness of ungrazed site	Reynolds & Trost (1980)
Desert Grassland	New Mexico	Grass density increased by 110% after 30 years of protection from grazing	Gardner (1950)
Semidesert Grassland	Arizona	Species richness increased, as did canopy cover for midgrass, shortgrass, shrub, and forb groups, after removal of livestock	Brady et al. (1989)
Semidesert Grassland	Arizona	Woody plants significantly more abundant after removal of livestock	Bock et al. (1984)
Ponderosa Pine Forest	Washington	Decreased species richness on grazed sites	Rummell (1951)
Mountain Canyon	Utah	Absence or near absence of 10 grass species on grazed sites	Cottam & Evans (1945)
Riparian	Oregon	Species richness increased from 17 to 45 species nine years after removal of livestock	Winegar (1977)
Riparian	Arizona	Herbaceous cover of grazed plot less than half that of ungrazed plot	Szaro & Pase (1983)
Riparian	Colorado	Shrub canopy coverage increased 5.5 times, willow canopy coverage 8 times after removal of livestock	Schulz & Leininger (1990)

Table 2. Deleterious effects of livestock grazing on vertebrate animals in western North America.

Organism(s)	*Location*	*Effect*	*Authority*
Small Mammals	Idaho	Density and diversity reduced on grazed sites	Reynolds & Trost (1980)
Small Mammals	Nevada	Density over one-third lower, diversity almost half on grazed sites	Medin & Clary (1989)
Songbirds, Raptors, and Small Mammals	Utah	350% increase in use and diversity after 8 years rest from grazing	Duff (1979)
Ducks and all Terrestrial Nongame Birds	Colorado	All more abundant in ungrazed habitat	Crouch (1982)
Upland Sandpiper (*Bartramia longicauda*)	North Dakota	Nest density reduced on grazed sites	Bowen & Kruse (1993)
Riparian Birds	Montana	Species composition altered by grazing; densities of ⅓ of species differed significantly between heavily and lightly grazed sites—⅔ of these were higher on lightly grazed sites	Mosconi & Hutto (1982)
Riparian Passerines	Southeastern Oregon	Species richness decreased on grazed sites	Taylor (1986)
Willow Flycatcher (*Empidonax traillii*)	Southeastern Oregon	Abundance increased from 0 to 30 when grazing intensity reduced by 4 times	Taylor & Littlefield (1986)
Yellow Warbler (*Dendroica petechia*)	Southeastern Oregon	Abundance increased by 8 times when grazing intensity reduced by 4 times	Taylor & Littlefield (1986)
Dickcissel (*Spiza americana*) and Bell's Vireo (*Vireo bellii*)	Oklahoma	Populations 50% lower on grazed sites	Overmire (1963)
Lizards	California	Abundance 2 times and biomass 3.7 times higher on ungrazed site	Busack & Bury (1974)
Lizards	Arizona	Abundance and diversity higher on ungrazed site in 4 of 5 vegetation types	Jones (1981, 1988)
Wandering Garter Snake (*Thamnophis elegans vagrans*)	New Mexico	5 times more abundant in ungrazed sites	Szaro et al. (1985)
Desert Tortoise (*Gopherus agassizi*)	Western U.S.A.	Livestock trample young tortoises, damage burrows and shrubs used for shelter, and remove critical forage	Berry (1978); Campbell (1988)
Trout (Salmonidae)	Great Basin	Average increase in production of 184% when grazing reduced or eliminated	Bowers et al. (1979)
Trout (Salmonidae)	Idaho	More abundant, larger fish after removal of livestock	Keller & Burnham (1982)
Trout (Salmonidae)	Colorado	Standing crop doubled after removal of livestock	Stuber (1985)

are important factors in determining the effect of grazing on grasshopper populations.

Grazing-induced changes in particular species translate into major conversions of community organization. Grazing is credited with transforming southern New Mexico from grassland to creosotebush (*Larrea*) desert (Whitfield & Anderson 1938; York & Dick-Peddie 1969). Kennedy (1977) noted that grazing thoroughly changed the primary plant species in most Southwest riparian zones. He referred to these changes as "com-

Table 3. Effects of livestock grazing on insects.

Location	*Effect*	*Authority*
Arizona	Grasshopper density 3.7 times greater on protected site in summer, 3.8 times greater on grazed site in fall (different subfamilies, with different food preferences dominant in each season)	Jepson-Innes & Bock (1989)
Australia	Ant abundance increased as sheep density increased; all other groups reduced substantially at highest livestock density	Hutchinson & King (1980)
Colorado	Grasshoppers significantly more abundant on a lightly grazed site than on a heavily grazed site; because there was no difference between the same sites 19 years earlier, a long-term effect of grazing is indicated	Welch et al. (1991)
Oklahoma	Decreases in abundance of most insect groups, dramatic increase in grasshoppers	Smith (1940)
South Dakota	Plant community architecture changed from midgrass/tallgrass to shortgrass, which changed grasshopper species composition	Quinn & Walgenbach (1990)

plete type conversions." Grazing can eliminate a willow stand within 30 years (Kovalchik & Elmore 1992). In Oregon, grazing delayed plant phenology two weeks (Kauffman et al. 1983*b*); such changes could have dramatic effects on communities of pollinators and dispersers. Grazing has also been observed to alter animal foraging guilds (Table 4).

Grazing destabilizes plant communities by aiding the spread and establishment of exotic species, such as tamarisk (*Tamarix*) (Ohmart & Anderson 1982; Hobbs & Huenneke 1992). Livestock help spread exotic plant species by (1) dispersing seeds in fur and dung; (2) opening up habitat for weedy species, such as cheatgrass (*Bromus tectorum;* Gould 1951; Mack 1981), which thrive in disturbed areas; and (3) reducing competition from native species by eating them. As D'Antonio and Vitousek (1992) pointed out, alien grass invasions in North America have been most severe in the arid and semiarid West, where invasion by many species (including *Bromus tectorum, B. rubens, B. mollis, B. diandrus, Taeniatherum asperum,* and *Avena* spp.) was associated with grazing.

Disruption of Ecosystem Functioning

The deleterious effects of livestock on native ecosystems are not limited to changes in species composition. Grazing also disrupts the fundamental ecosystem functions of nutrient cycling and succession.

An often overlooked characteristic of arid and semiarid ecosystems is the presence of microbiotic (or cryptogamic) soil crusts, delicate symbioses of cyanobacteria, lichens, and mosses from a variety of taxa. The essential role of these microbiotic crusts in nutrient cycling of arid ecosystems has been increasingly appreciated. Crusts perform the major share of nitrogen fixation in desert ecosystems (Rychert et al. 1978). The availability of nitrogen in the soil is a primary limiting factor on biomass production in deserts. In the Great Basin Desert, at least, it is second in importance only to the lack of moisture (James & Jurinak 1978). Microbiotic crusts in arid ecosystems have been correlated with increased organic matter and available phosphorus (Kleiner & Harper 1977), increased soil stability (Kleiner & Harper 1972; Rychert et al. 1978), and increased soil water infiltration (Loope & Gifford 1972; Rychert et al. 1978). Crusts also play an important role in ecological succession because they provide favorable sites for the germination of vascular plants (St. Clair et al. 1984).

Given the fragile nature of microbiotic crusts, it follows that they are easily damaged by livestock grazing. In numerous studies, grazing has been correlated with the loss of microbiotic cover (Wullstein 1973; Johansen et al. 1981; Anderson et al. 1982; Jeffries & Klopatek 1987). Crusts can be severely disrupted even while they (Belnap 1993) and the more conspicuous vascular plant communities (Kleiner & Harper 1972; Cole 1990) appear healthy. Microbiotic species richness has also been shown to decrease under grazing pressure (Anderson et al. 1982). Recent studies on the Colorado Plateau have dramatically demonstrated that soil surface disturbances can virtually stop nitrogen fixation. Nitrogenase activity was reduced 80–100% in the microbiotic crust under a single human footprint, as well as under vehicle tracks (Belnap, personal communication; Belnap 1994; Belnap et al. 1994), and nitrogen content in the leaves of dominant plant species was lower in trampled than untrampled areas (Belnap, personal communication; Harper & Pendleton 1993). If a single footprint can bring a local nitrogen cycle almost to a halt, the impact of a century's work of livestock hoofprints can easily be imagined.

Grazing also can disrupt ecological succession. The cumulative impact of long-term livestock use has produced and maintained early seral vegetation throughout much of the West (Longhurst et al. 1982). Glinski (1977) demonstrated that cattle grazing of small seedlings prevented cottonwood (*Populus fremontii*) regeneration in a southern Arizona riparian zone. He concluded that long-term grazing could eliminate or reduce the upper canopy by preventing the establishment of saplings. Carothers et al. (1974) noted the lack of cottonwood regeneration in grazed areas along the Verde River, Arizona. Prevention of seedling establishment due to grazing and trampling by livestock has transformed a variety of Southwest riparian systems into even-aged,

Table 4. Effects of livestock grazing on animal foraging guilds in western North America.

Organisms	*Location*	*Effect*	*Authority*
Riparian Birds	Montana	Flycatching guild, ground-foraging thrush guild and foliage-gleaning insectivore guild affected; bark-foraging guild unaffected	Mosconi & Hutto (1982)
Riparian Birds	Oregon	Grazed sites preferred by insectivores, ungrazed sites by herbivores and granivores	Kauffman et al. (1982)
Lizards	Arizona	More sit-and-wait lizards on grazed sites; open-space foragers and wide-ranging foragers decreased on grazed sites	Jones (1981)
Grasshoppers	South Dakota	Obligate grass-feeders dominated on grazed sites, mixed-forb-and-grass-feeders on ungrazed sites	Quinn & Walgenbach (1990)

nonreproducing vegetative communities (Carothers 1977; Szaro 1989). In Oregon, grazing retarded succession in the willow-black cottonwood (*Salix-Populus trichocarpa*) community, and there was little if any regeneration of alders (*Alnus*) or cottonwoods (Kauffman et al. 1983*b*). Davis (1977) concluded that livestock grazing was "probably the major factor contributing to the failure of riparian communities to propagate themselves."

Ascertaining patterns of ecological succession in xeric rangelands is not easy; thus, the effect of livestock on successional processes is unclear. Traditionally, range management was based upon Clements' (1916) classic model of ecological succession, where seral stages lead to a stable climax. Early on, this concept of predictable, directional succession was applied to range ecosystems (Sampson 1919). This "range succession model" eventually formed the basis of range condition classification, as exemplified by government manuals and early range management textbooks (Stoddart & Smith 1943), and summarized in an extensive review by Ellison (1960). In the arid West, however, vegetation change due to grazing has not followed the prediction of this linear model. Recent evidence suggests that range ecosystems have not evolved as well-balanced communities with stable species compositions (Johnson & Mayeux 1992).

More recently, a less Clementsian view of xeric rangeland succession, referred to as the "state-and-transition model," has been proposed (Westoby et al. 1989). According to this model, relatively stable, discrete vegetation states go through transitions induced by natural episodic events such as fire or by management actions such as grazing (Laycock 1991). As Friedel (1991), Laycock (1991), and others have discussed, transitions between states sometimes cross successional "thresholds." Once certain thresholds have been crossed, as in severe soil erosion, succession may not be reversible except by strong, active management. Although this model is in its infancy, it may someday provide a means to predict if grazing can cause long-term degradation by inducing irreversible succession across thresholds.

Alteration of Ecosystem Structure

The physical structure of ecosystems, including vegetation stratification, is often changed by livestock grazing. In central Washington, grazing was responsible for changing the physical structure of ponderosa pine forest from an open, park-like tree overstory with dense grass cover to a community characterized by dense pine reproduction and lack of grasses (Rummell 1951). Grazing was at least partially responsible for similar structural changes in ponderosa pine forests of northern Arizona (Cooper 1960). Historic records indicate that extensive willow stands once occurred throughout the rangelands of the Intermountain West, which are now almost completely absent (Kovalchik & Elmore 1992). Grazing structurally changed habitat for the wandering garter snake (*Thamnophis elegans vagrans*) through the loss of small trees and shrubs (Szaro et al. 1985). In central Arizona, lizard habitat was changed when livestock reduced low-height vegetation by totally consuming perennial grasses and severely reducing palatable shrubs (Jones 1981). In Oregon, Taylor (1986) noted that lower vegetative strata were affected by grazing. In blackbrush (*Coleogyne ramosissima*) desert habitat, ungrazed sites had significantly more shrub and herbaceous cover (Jeffries & Klopatek 1987). In a high-altitude willow riparian community in Colorado, grazing influenced the spacing of plants and the width of the riparian zone (Knopf & Cannon 1982).

Grazing removes soil litter, which can have both physical and biological effects. Schulz and Leininger (1990) observed twice as much litter in an exclosure as in surrounding grazed habitat. In Oregon, removal of soil litter was thought to be the cause of delayed plant phenology (Kauffman et al. 1983*b*), which in turn could affect communities of animal pollinators.

Researchers have long recognized that grazing contributes to the deterioration of soil stability and porosity and increases erosion and soil compaction. Seventy years ago, Aldo Leopold (1924) declared that "grazing is the prime factor in destroying watershed values" in Arizona. Grazing reduces the roughness coefficient of watersheds, resulting in more surface runoff, more soil erosion, and massive flooding (Ohmart & Anderson 1982). Grazing in the upper Rio Grande changed plant cover, thus increasing flash floods and, consequently, erosion (Cooperrider & Hendricks 1937). As grazing-induced gullying lowered the stream channel along an Oregon stream, associated plant communities changed from wet meadow to the more xeric sagebrush-rabbitbrush (*Chrysothamnos*) type (Winegar 1977). Davis (1977) concluded that removal of upland vegetation by livestock was a major factor in the increase in devastating floods. Numerous authors have noted extreme erosion and gullying when comparing heavily grazed to ungrazed sites (see Cottam & Evans 1945; Gardner 1950; Kauffman et al. 1983*a*). Ellison (1960) concluded that "as a result of some degree of denudation, accelerated soil erosion is inseparably linked with overgrazing on arid lands the world over."

Grazing has also repeatedly been shown to increase soil compaction and thus decrease water infiltration (Alderfer & Robinson 1949; Orr 1960; Rauzi & Hanson 1966; Bryant et al. 1972; Rauzi & Smith 1973; Kauffman & Krueger 1984; Abdel-Magid et al. 1987; Orodho et al. 1990). In arid and semiarid lands where water is the primary ecological limiting factor, major losses of water from ecosystems can lead to severe desertification. Some controversy exists as to whether livestock grazing was the *cause* of increased flooding and erosion or whether the synchrony of increased channel trenching and the introduction of vast livestock herds during the last century was coincidental. Episodes of channel trenching certainly occurred prior to the introduction of livestock (Bryan 1925; Karlstrom & Karlstrom 1987). Most reviewers, however, conclude that, at the least, livestock have been a contributing factor to the entrenching of stream channels in the Southwest (Bryan 1925; Leopold 1951; Hereford & Webb 1992; Betancourt 1992). This interaction of climatic, geomorphic, and biological factors has been summarized as a "trigger-pull": long-term climatic trends were already underway when cattle arrived to serve "as the trigger-pull that set off an already loaded weapon" (Hastings 1959).

Costs of Grazing Magnified: Riparian Habitats in the Arid West

Livestock, like humans, are adapted to mesic habitats, and they select riparian areas for the same reasons we do: shade, cooler temperatures, and water. In addition, riparian areas offer an abundance of food. Many observers have noted that cattle spend a disproportionate amount of their time in riparian zones (Ames 1977; Kennedy 1977; Thomas et al. 1979; Roath & Krueger 1982; Van Vuren 1982; Gillen et al. 1984). That livestock actively select riparian habitats, however, is a cause for ecological concern because these habitats are among the biologically richest in many arid and semiarid regions and are easily damaged. Because livestock spend much of their time in riparian communities, and because the ecological stakes are highest here, many of the adverse impacts of grazing are magnified in these habitats.

Western riparian zones are the most productive habitats in North America (Johnson et al. 1977), providing essential wildlife habitat for breeding, wintering, and migration (Gaines 1977; Stevens et al. 1977; Brode & Bury 1984; Laymon 1984; Lowe 1985). Riparian habitats in the Southwest are home to the North American continent's highest density of breeding birds (Carothers et al. 1974; Carothers & Johnson 1975), rarest forest type, and more than 100 state and federally listed threatened and endangered species (Johnson 1989). Approximately three-quarters of the vertebrate species in Arizona and New Mexico depend on riparian habitat for at least a portion of their life cycles (Johnson et al. 1977; Johnson 1989). Even xeroriparian habitats—normally dry corridors that intermittently carry floodwaters through low deserts—support five to ten times the bird densities and species diversity of surrounding desert uplands (Johnson & Haight 1985).

Sadly, these biological treasures are in extreme danger. The Environmental Protection Agency concluded that riparian conditions throughout the West are now the worst in American history (Chaney et al. 1990). Over 90% of Arizona's original riparian habitat is gone (Johnson 1989). Less than 5% of the riparian habitat in California's Central Valley remains; 85% of that is in disturbed or degraded condition (Franzreb 1987). The degradation of Western riparian habitats began with severe overgrazing in the late Nineteenth Century (Chaney et al. 1990), and grazing remains "the most insidious threat to the riparian habitat type today" (Carothers 1977). An extensive survey of Southwest riparian community types concluded that "livestock may be the major cause of excessive habitat disturbance in most western riparian communities" (Szaro 1989). The Oregon-Washington Interagency Wildlife Committee (1979), composed of biologists from several government agencies, concluded that grazing is the most important factor in degrading wildlife and fisheries habitat throughout the 11 western states. Likewise, ecologists in Montana suggested that livestock grazing is the major cause of habitat disturbance in most western riparian communities (Mosconi & Hutto 1982).

Livestock affect four general component of riparian systems: (1) streamside vegetation, (2) stream channel morphology, (3) shape and quality of the water column, and (4) structure of streambank soil (Platts 1979, 1981, 1983; Kauffman & Krueger 1984; Platts & Nelson 1989). As summarized by Platts (1981), "Grazing can affect the streamside environment by changing, reducing, or eliminating vegetation bordering the stream. Channel morphology can be changed by accrual of sediment, alteration of channel substrate, disruption of the relation of pools to riffles, and widening of the channel. The water column can be altered by increasing water temperature, nutrients, suspended sediment, bacterial populations, and in the timing and volume of streamflow. Livestock can trample streambanks, causing banks to slough off, creating false setback banks, and exposing banks to accelerated soil erosion."

Riparian vegetation is altered by livestock in several ways: (1) compaction of soil, which increases runoff and decreases water availability to plants; (2) herbage removal, which allows soil temperatures to rise, thereby increasing evaporation; (3) physical damage to vegetation by rubbing, trampling, and browsing; and 4) altering the growth form of plants by removing terminal buds and stimulating lateral branching (Kauffman & Krueger 1984; Szaro 1989). Livestock grazing is one of the principal factors contributing to the decline of native trout in the West. Cattle activities especially deleterious to fish are the removal of vegetative cover and the trampling of over-hanging streambanks (Behnke & Zarn 1976). Livestock have been shown to decrease water quality of streams (Diesch 1970; Buckhouse & Gifford 1976). Changes in water chemistry (Jeffries & Klopatek 1987) and temperature (Van Velson 1979), in effect, create an entirely new aquatic ecosystem (Kennedy 1977; Kauffman & Krueger 1984). Insights such as these led the American Fisheries Society to issue a formal position statement calling for an overhaul of riparian zone management (Armour et al. 1991).

Historical and Management Considerations

By virtually any measure, livestock grazing has serious ecological costs in western North America. Grazing has reduced the density and biomass of many plant and animal species, reduced biodiversity, aided the spread of exotic species, interrupted ecological succession, impeded the cycling of the most important limiting nutrient (nitrogen), changed habitat structure, disturbed community organization, and has been the most severe impact on one of the biologically richest habitats in the region. While undoubtedly there are exceptions to this theme of destruction, clearly much of the ecological integrity of a variety of North American habitats is at risk from this land management practice.

In addition to grazing per se, the industry of livestock production entails a number of indirect costs to native biodiversity. Livestock compete with native herbivores for forage ("usurpation") and often consume the most nutritive species ("highgrading"). Fencing, which is a fundamental livestock management tool, creates obstacles for many native wildlife species, such as the pronghorn (*Antilocapra americana*). The livestock industry has played a large role in the elimination of native predators; some of the most vehement opposition to predator reintroduction continues to come from livestock interests. Exotic species, such as crested wheatgrass (*Agropyron cristatum*), are planted as "range improvements." In addition, livestock can transmit disease to native animals (Mackie 1978; Longhurst et al. 1983; Menke & Bradford 1992).

Agency management priorities often overemphasize livestock needs at the expense of wildlife. A recent Congressional study of BLM and Forest Service management confirmed that wildlife receives only a small percentage of available staffing and funding. During fiscal years 1985–1989 the BLM directed only 3% of its total appropriation toward wildlife habitat management, while 34% of its budget went to its three consumptive programs: range, timber, and energy and minerals (U.S. General Accounting Office 1991*b*). Wildlife at national wildlife refuges also suffers from management emphasis on livestock. Cattle grazing and haying occur at 123 refuges; at any given site these activities occupy up to 50% of refuge funds and 55% of staff time. Field studies indicated that these livestock-related activities directly impeded wildlife conservation (Strassman 1987). Strong agency bias in favor of grazing often leads to contradictory management decisions. A recent Forest Service analysis of sensitive vertebrate species identified livestock grazing as one of five factors jeopardizing the northern goshawk (*Accipiter gentilis*) in the Southwest (Finch 1992). Yet the goshawk management recommendations (Reynolds et al. 1992), released by the same office in the same year, did not even mention grazing. Such predilections by agencies reflect similar biases within the range management discipline: a recent 500-page textbook on range management (Holechek et al. 1989) devotes one paragraph to nongame wildlife.

A variety of justifications are heard for grazing in the West. Because livestock has been such a prominent component of Euro-American settlement of the West, some observers see it as a traditional pastime and assume it is appropriate for the land. Some range managers maintain that livestock are actually *necessary* for ecosystem health, that "grass needs grazing" (Chase 1988; Savory 1988). Popular claims such as these are rooted in a scientific debate on the consequences of herbivory on grassland ecosystems. As the "herbivore optimization" hypothesis goes, loss of tissue to herbivores can actually increase total productivity of the grazed plant. Such a response to herbivory is referred to as "overcompensation" by the plant (Owen & Wiegart 1976; Dyer et al. 1982). When different levels of ecological hierarchy (individual, population, community; Belsky 1987) and a wide diversity of ecosystem types, geographic settings, and degrees of management intensity are lumped together into one generalized theory, clarity is lost. Much of the evidence for overcompensation comes from highly productive and intensively managed systems, not from arid rangelands (Bartolome 1993). Few studies have demonstrated overcompensation in western North America (Painter & Belsky 1993), where much of the rangeland resource is not grassland. Observations of native herbivores lend no support to the idea that compensatory growth has any relevance at the community level in western rangelands (Patten 1993). According to Vicari and Bazely (1993), "there is little evidence that the act of grazing per se increases the fitness of grasses, or any other plant species, except under highly specific circumstances."

Other scientists and range managers suggest that livestock, given their capacity for altering so many aspects of ecological organization, could be used as a wildlife management tool (Bokdam & Wallis de Vries 1992; Hobbs & Huenneke 1992). In summarizing a symposium on the topic, Severson (1990) clarified that such applications may be very limited, and that what benefits one species may prove detrimental to another. Because two species in the same community may vary in their response to grazing (Hobbs & Huenneke 1992), determination of its success or failure as a management practice depends on which species is used as a criterion. On many national wildlife refuges, grazing and haying occur with the rationale that these practices will benefit wildlife. Upon review of 123 refuges, Strassman (1987) concluded that "although in theory cattle grazing and haying can be wildlife management tools, as implemented they are tools that do more harm than good."

It is often stated that livestock have merely taken the place of large native herbivores, particularly bison (*Bison bison*). The presettlement abundance of bison on the Great Plains is legendary. West of the Rocky Mountains, however, bison were rare or absent in Holocene times. The species was present in the northern Rockies region, marginally present along the northern and western perimeter of the Great Basin (Hall 1981; Mack & Thompson 1982; Zeveloff 1988; Van Vuren & Deitz 1993) and absent altogether from Arizona (Cockrum 1960; Hoffmeister 1986), western New Mexico (Bailey 1971), as well as most of California (Jameson & Peeters 1988), and Nevada (Hall 1946). The native steppe vegetation of much of the Intermountain West, characterized by caespitose bunchgrasses and a prominent microbiotic crust, reflects the absence of large numbers of large-hooved, congregating mammals. These steppe ecosystems have been particularly susceptible to the introduction of livestock; microbiotic crusts, as mentioned earlier, are easily damaged by trampling. In contrast, the slightly wetter Great Plains grasslands, characterized by rhizomatous grasses and a lack of microbiotic crusts, were well-adapted to withstand herbivory by large ungulates (Stebbins 1981; Mack & Thompson 1982). Theoretically, then, the Great Plains should be better suited to livestock grazing than the arid and semiarid ecosystems west of the Rockies. It should also be noted that the ecological analogy between cattle and bison is incomplete. Cattle, unlike bison, spend a disproportionate amount of time in riparian habitats. In a comparative study of cattle and bison feeding ecology in the Henry Mountains, Utah, Van Vuren (1982) noted that cattle distribution was limited to gentle slopes near water, regardless of forage, while bison roamed widely, seemingly unaffected by slope or proximity to water.

The controversy about flood cycles and arroyo-cutting, discussed earlier, is but one part of a larger controversy concerning the respective roles of climate change and human land use—including livestock grazing—in changing the vegetation of western North America. The international borderlands of southern Arizona and northern Sonora, Mexico, have been the site of the most intensive study of this issue. The appearance of *The Changing Mile* (Hastings & Turner 1985) almost three decades ago promoted the then new idea that the region's dramatic vegetation change during the previous century was due to increasing aridity—to natural climate change—and not to human land-use patterns. Using pairs of photographs, one historic and one recent, *The Changing Mile* visually documented vegetation change and concluded that its cause was an increasingly arid climate. As for livestock, these authors felt the evidence was somewhat ambiguous and concluded that livestock may have contributed to vegetation change in the region "but have not been the primary agent of change" (Hastings & Turner 1965). This work has since been widely quoted by livestock interests to support the idea that historic overgrazing was overstated and, therefore, to justify the continuation of grazing in the region.

Recently vegetation change along the Arizona borderlands has received renewed scholarly attention. This new work reached a very different conclusion: "probably no single land use has had a greater effect on the vegetation of southeastern Arizona or has led to more changes in the landscape than livestock grazing range management programs. Undoubtedly, grazing since the 1870s has led to soil erosion, destruction of those plants most palatable to livestock, changes in regional fire ecology, the spread of both native and alien plants, and changes in the age structure of evergreen woodlands and riparian forests" (Bahre 1991). Moreover, the new analysis (Bahre 1991) states that "the present historic evidence ... casts serious doubt on the hypothesis that a shift toward greater aridity is the primary factor for regional vegetation changes." Bahre (1991) agrees that climatic oscillations since 1870 have resulted in short-term fluctuations in vegetation but insists that long-term directional changes, including degradation of riparian habitats and spread of exotic species, have resulted from human disturbances, including overgrazing by cattle. Bahre challenges the conclusions of *The Changing Mile* on the basis of several factors, including lack of historic evidence to support several key assumptions in the earlier work (for example, that overgrazing had been practiced since the time of the Mexican occupation), and that the majority of historic photographs were taken *after* the worst grazing damage had already occurred. In other words, *The Changing Mile* made comparisons to the wrong baseline data. For now, the best historic evidence seems to support the idea that livestock grazing, interacting with fluctuations in climatic cycles, has been a primary factor in altering ecosystems of the Southwest.

Human intervention is needed to restore the West to ecological health. According to the BLM's own definition, over 68% of its lands are in "unsatisfactory" condition (Wald & Alberswerth 1989; U.S. General Accounting Office 1991*a*). Approximately 464 million acres of American rangeland have undergone some degree of desertification (Dregne 1983). Attempts at restoration of livestock-damaged ecosystems have offered both good and bad news: riparian areas often show rapid recovery upon removal of livestock, but more xeric uplands demonstrate little inherent capacity for healing.

Riparian areas appear to be relatively resilient. At a Sonoran Desert spring, Warren and Anderson (1987) documented dramatic recovery of marsh and riparian vegetation within five years of livestock removal. All nine aspects of trout habitat studied along Summit

Creek, Idaho, improved within two years of livestock removal (Keller et al. 1979). Mahogany Creek, Nevada, also showed major improvement in fisheries habitat after only two years of exclosure (Dahlem 1979). Beaver and waterfowl returned to Camp Creek, Oregon, within nine years of cattle exclosure (Winegar 1977). However, the aquatic component of riparian systems often is the quickest to show improvement. Szaro and Pase (1983) observed extremely limited recovery of a cottonwood-ash-willow association in Arizona after four years. Knopf and Cannon (1982) noted that a willow community was slower to heal than the adjacent stream: 10–12 years was insufficient for recovery of the former.

The U.S. General Accounting Office (1988*b*) recently reviewed riparian restoration efforts on BLM and Forest Service lands in the West and concluded (1) that even severely degraded habitats can be successfully restored and (2) that successful restoration to date represents only a small fraction of the work that needs to be done. They noted that successful techniques varied considerably from site to site, and that many sites could repair themselves, given respite from livestock. Successful riparian restoration efforts are summarized by the U.S. General Accounting Office (1988*b*) and Chaney et al. (1990).

In numerous studies of riparian grazing impact, investigators concluded that total removal of livestock was necessary to restore ecosystem health. Along Mahogany Creek, Nevada, reduction in grazing had little benefit; only a complete removal brought about habitat improvement (Dahlem 1979; Chaney et al. 1990). Ames (1977) found that even short-term or seasonal use is too much and compared mere reductions in livestock numbers to letting "the milk cow get in the garden for one night." In a recent comparison of 11 grazing systems, total exclusion of livestock offered the strongest ecosystem protection (Kovalchik & Elmore 1992). As Davis (1982) put it, "If the overgrazing by livestock is one of the main factors contributing to the destruction of the habitat, then the solution would be to ... remove the cause of the problem."

The vast majority of damaged rangeland acreage is on arid and semiarid lands, where the prognosis for restoration is poor (Allen & Jackson 1992). To rehabilitate arid lands is somewhat analogous to trying to grow a garden without water. Perhaps because there is little chance of rapid success, land managers have been slow to take up the challenge of restoring arid rangelands. Cooperrider (1991) noted that "the principal purpose of most rangeland rehabilitation projects has been restoration of livestock forage. Such projects typically end up reducing plant and animal species diversity." Some dryland restoration projects touted as success stories (such as the Vale project in southeastern Oregon; Menke & Bradford 1992), actually have entailed large-scale plantings of exotic species. Such activities restore livestock forage, not native ecosystems.

Is there an ecologically sustainable future for livestock grazing in western North America? This ultimately is a question of human values, not of science. We must decide how much we really care about native diversity and ecosystem processes and what we are willing to do to sustain them. Ecological science and conservation biology have a key role to play in helping society make a wise decision. Scientific input into grazing issues has come laden with resource extraction assumptions: one of the primary goals of range management is to maximize livestock production (Stoddart & Smith 1943; Bell 1973; Menke & Bradford 1992) or to "improve the output of consumable range products" (Holechek et al. 1989). Given this economic underpinning, the ecological merit of livestock in the West has generally gone unchallenged. It is time that conservation biologists take a careful look at the most pervasive land use in western North America and scrutinize the practice described as "the single most important factor limiting wildlife production in the West" (Smith 1977) and "one of the primary threats to biological diversity" (Cooperrider 1991). Whatever decision society reaches, it will be a wiser, more informed one if the conservation biology community contributes its insights to the debate.

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Reducing the impact of summer cattle grazing on water quality in the Sierra Nevada Mountains of California: a proposal

Robert W. Derlet, Charles R. Goldman and Michael J. Connor

ABSTRACT

The Sierra Nevada Mountain range serves as an important source of drinking water for the State of California. However, summer cattle grazing on federal lands affects the overall water quality yield from this essential watershed as cattle manure is washed into the lakes and streams or directly deposited into these bodies of water. This organic pollution introduces harmful microorganisms and also provides nutrients such as nitrogen and phosphorus which increase algae growth causing eutrophication of otherwise naturally oligotrophic mountain lakes and streams. Disinfection and filtration of this water by municipal water districts after it flows downstream will become increasingly costly. This will be compounded by increasing surface water temperatures and the potential for toxins release by cyanobacteria blooms. With increasing demands for clean water for a state population approaching 40 million, steps need to be implemented to mitigate the impact of cattle on the Sierra Nevada watershed. Compared to lower elevations, high elevation grazing has the greatest impact on the watershed because of fragile unforgiving ecosystems. The societal costs from non-point pollution exceed the benefit achieved through grazing of relatively few cattle at the higher elevations. We propose limiting summer cattle grazing on public lands to lower elevations, with a final goal of allowing summer grazing on public lands only below 1,500 m elevation in the Central and Northern Sierra and 2,000 m elevation in the Southern Sierra.

Key words | cattle grazing, eutrophication, non-point pollution, Sierra Nevada, water quality, watersheds

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INTRODUCTION

Cattle grazing has been a part of the landscape in remote regions of the western United States since the 1850s. In the past, much of this land was not cultivatable and not inviting to human settlement due to the harsh climate, rugged terrain or inaccessibility (Young & Sparks 1985). Thus cattle grazing over an otherwise unusable landscape served a purpose in the development and advancement of the West. However, as far back as the 1880s the detrimental effect of cattle on alpine water quality was noted, and cited as one of the reasons to establish Yosemite National Park in 1890 (Farquhar 1965; Runte 1992).

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The greatest economic value of the Sierra Nevada Mountains is derived from the provision of abundant quantities of fresh water for California (Goldman 2000). Since 1900, California's population has increased from 1.5 million to over 36 million persons (US Census Bureau 2008). This large increase has placed high demands on the limited available supply of clean drinking water (Carle 2004). California's population will soon approach 40 million, and protected watersheds serve the purpose of providing a clean and unpolluted water source. The Sierra Nevada watersheds provide 50% of California's fresh water for domestic use (Carle 2004). Reno, Nevada is also heavily dependent upon the out flow of the Sierra Nevada, primarily from Lake Tahoe which restricted grazing many years ago.

The unique geographic features of the Sierra Nevada have resulted in challenges to maintain water quantity and quality for this essential source. Melting snow must pass through a fragile ecosystem prior to runoff into lowland reservoirs. Much of the watershed consists of surface or near surface granite or metamorphic bedrock, with little topsoil and has little buffering capacity (Moore 2000). As a result small amounts of environmental pollution may have a significant impact on aquatic life since there is little or no biogeochemical retention, transformation, or fixation of trace elements or reduction of major nutrients such as nitrogen and phosphorus. Therefore relatively small amounts of nutrient addition or habitat disturbance leads to significant impacts on nutrient flux and subsequent impacts on the aquatic ecosystems of lakes and streams. Much of this watershed encompasses roadless, remote back-country wilderness areas at high elevations that without pollutant sources should yield outstanding water quality. However, over the past 150 years there have been ongoing threats to water quality from cattle grazing that have continued despite the renewed national focus on source watershed protection and non-point pollution. The 1996 amendments to the 1974 U.S. Safe-Drinking Water Act now require that the states conduct a source water assessment, including non-point pollution monitoring and enforcement (Environmental Protection Agency 1996). The EPA placed additional regulations on specific pathogens in 2006, including *Cryptosporidium,* a protozoon pathogen commonly found in cattle. This act provides a strong legislative mandate to ensure that Sierra Nevada headwaters are not polluted from cattle grazing, or threatened from other domesticated animals. The importance of source watershed protection in the Sierra Nevada is also exemplified by the cooperative agreement between the City of San Francisco-Hetch-Hetchy Authority and Yosemite National Park. Signs exist on public hiking trails in the Tuolumne Meadows area of Yosemite outlining the need for source watershed protection (Derlet 2009, unpublished data).

DEMOGRAPHICS OF CATTLE

California has 5.5 million head of cattle, of which nearly 2 million are cows used in dairy operations (USDA 2008). Many of remaining 3.5 million head are involved primarily in beef production. Most are raised in feedlots. Some ranchers in California practice “transhumance”, by transporting livestock by truck from valley lowlands to the USDA Forest Service and U.S. Bureau of Land Management (BLM) lands in the Sierra Nevada in the summer in relation to forage availability (Sulak & Huntsinger 2002). In California, winter lowland range generally consist of valley and foothill grasslands and oak savanna, less than 300 m in altitude, while the summer Sierra ranges are usually high elevation mountain meadows, at altitudes up to 3,100 m, and are snow covered much of the year. Based on available Forest Service data fewer than 40,000 head of cattle are moved to Sierra Nevada mountain areas for summer pasturing. Such use of mountain range grazing on public lands in the Sierra Nevada predates the establishment of the National Forests in 1906, but was institutionalized and is now controlled through the granting of summer grazing permits by the U.S. Forest Service. To accommodate differences in forage productivity with ecotype and use of rangelands by different types of livestock, the Forest Service and other federal agencies allocate grazing privileges on public lands based on an Animal Unit Month (AUM) or “head month” which is the amount of forage that a mature cow and her calf (or the equivalent, in sheep or horses) can eat in 1 month. In California as a whole, the Forest Service allows livestock grazing on about 12.4 million acres of forest land that have the potential to provide about 486,384 AUM of forage of which some 374,089 AUM (76.9%) were used in 2004 (GAO 2005). Grazing allotments in the Sierra are less than half of the entire state. The Forest Service charges livestock operators $1.35 per AUM to graze livestock on the federal lands, or about $4.05 per cow for the summer. This is heavily subsidized since the actual cost to the Forest Service is $12.26 per AUM just to recover the costs of administering the grazing program (GAO 2005). While some ranchers may experience a cost benefit of inexpensive grazing land, long-term societal costs are higher, in terms of both ecological and public health costs.

The ecological costs of grazing on public lands can be dramatic, and include loss of diversity, lowering of population densities for a variety of taxa, disruption of nutrient recycling and succession, and changes in the characteristics of terrestrial and aquatic habitats (Fleischner 1994). The problems of cattle grazing on many of natural resources and ecosystems, especially degradation of aquatic habitats are well documented (Belsky *et al.* 1999), as are impacts to water quality (Derlet & Carlson 2006). We believe the public health costs to California of summer livestock grazing in the Sierra Nevada exceed its benefits. As far back as 1965, experts on the Sierra Nevada recognized that there was no real net economic benefit to the cattle industry to summer cattle grazing in the Sierra Nevada (Farquhar 1965). Despite discussions on the impact of cattle grazing in the Sierra Nevada, the some ranchers have recently pressured the USDA Forest Service to expand cattle grazing tracts (USDA Forest Service 2006).

EUTROPHICATION OF THE WATERSHED

Globally, concern has been raised about serious threats to the planet's drinking water supply from eutrophication of watersheds (Conley *et al.* 2009). Over the past 150 years, deposition of rate-limiting substances such as phosphorus (P) and nitrogen (N) compounds has resulted in eutrophication of much of the Sierra Nevada, with increases in phytoplankton species and biomass (Goldman 2000). Cattle manure contains high amounts of both N and P compounds, and 100 head of cattle will collectively deposit 50 kg of N and 25 kg of P each day on the range, based on a mean animal weight of 400 kg (Ohio State University 2006). Thus, fecal matter from cattle with N and P as well as other nutriments contributes to the eutrophication process (Belsky *et al.* 1999). In addition, this has promoted conditions which increase bacteria, other microorganisms, and the frequency of algal blooms (Yers *et al.* 2005; Conley *et al.* 2009). Non-point pollution from cattle waste poses a serious eutrophication threat to both surface and ground water sources at both higher and lower elevations (Klott 2007). This promotes imbalance in the ecosystems with accelerated eutrophication through fertilization of algae favoring the undesirable cyanobacteria at the expense of the more desirable diatoms and green algae (Horne & Goldman 1994). Cyanobacteria have been linked to the death of over 100 head of cattle in alpine regions of Switzerland as a result of excessive growth of this algae and secretion of the toxin microcystin in normally oligotrophic lakes (Mez *et al.* 1997).

In lowland grassland areas many nutrients and toxic substances are fixed or adsorbed to soil particles and soil fungi, which can greatly reduce nutrient loading of surface waters. Although P is adsorbed to soil particles and tends to be retained by the earth, N in contrast moves easily through soil, which then flows into ground water, which in turn reaches stream drainage and eventually to California's lakes and reservoirs (Horne & Goldman 1994). However, because much of the Sierra Nevada is granite with only a thin layer of soil in some areas, the adsorptive ability for P by soil is limited, thus allowing P to enter lakes and streams (Goldman 2000). Cattle grazing can also impact aquatic life. A study in the Golden Trout Wilderness which compared grazed with non-grazed areas showed a decreased fish biomass in grazed areas (Knapp & Matthews 1996). Mountain insects have also been found to have been affected in cattle grazing areas (Del Rosario *et al.* 2002).

Some cattle are grazed in specially designated Wilderness areas of the Sierra Nevada, where over-night human visitation is restricted to limit impact on the wilderness eutrophication by humans. However, the focus on humans is misguided. Range cattle excrete a mean of 50 kg/day of wet weight manure into the alpine landscape (Ohio State University 2006). In contrast, healthy human waste is only 0.10 to 0.15 kg/day (Rendtorff & Kashgarian 1967). Thus, each head of cattle produces up to 500 times as much waste as a single human in a single day and therefore each animal impacts the environment far more than each human.

HARMFUL MICROORGANISMS

Cattle excrete microorganisms which can be harmful to humans (Berry *et al.* 2006). Our studies have also shown significantly higher levels of both heterotrophic and pathogenic microorganisms in the Sierra Nevada areas where cattle graze, compared with non-grazing areas (Derlet & Carlson 2004, 2006; Derlet *et al.* 2008).

In watersheds where cattle have grazed, 96% of surface water samples contained significant indicator levels of *E. coli* of 100 CFU/100 ml or more, placing these waters at high risk for harboring the large variety of harmful microorganisms (Derlet *et al.* 2008). In contrast, the California water board does not allow more than 2.2 CFU/100 ml of *E. coli* in water used to irrigate vegetable crops. Thus, Sierra water in cattle grazing watersheds may contain 40 times as many *E. coli* as would be allowed to be used on vegetable crops. In contrast, adjacent non-grazed watersheds had a prevalence of less than 10% medically significant *E. coli*. *E. coli* and coliform bacteria have long been established as indicators of fecal pollution of watersheds and water supplies (American Public Health Association 1998). Diseases such as entero-invasive *E. coli*, *Giardia*, *Cryptosporidium*, *Salmonella*, *Campylobacter*, *Yersenia* species and other microbial pathogens, some that can survive for extended periods in the environment, are likely to be among those present. (Harvey *et al.* 1976; Byappanahalli *et al.* 2003). Cattle serve as asymptomatic carriers for many of these organisms. One recent study found as many as over 50,000 *Giardia* cysts/gram of cattle manure in asymptomatic infected cattle (Gow & Waldner 2006). Thus over 2 billion cysts may be excreted from an infected animal each day based on 50 kg of manure/day, enough to infect several million persons with the minimal infective dose of 10 cysts. Removal of the entire list of pathogenic bacteria by municipal water districts is an expensive multi step process. In Milwaukee, municipal water intake of accidental sewage spillage near intake pipes led to nearly one-third of the city population becoming infected with *Cryptosporidium*, despite standard water treatment (Mackenzie *et al.* 1994). Drought conditions increase the prevalence of pathogens and substrate, which may make some municipal purification processes less effective by concentrating pathogens (Derlet *et al.* 2008). Understanding factors that impact the water quality from any watershed is essential for intelligent and effective land management decisions.

Finding medically significant coliforms in surface water below cattle grazing areas is not unique to the Sierra Nevada, as several studies from other areas of the U.S. have demonstrated a high prevalence of coliforms in watersheds grazed by cattle (Yers *et al.* 2005). A study of South Carolina watersheds found non-point pollution with *E. coli* to be high in cattle grazing areas (Klott 2007). Miller found up to 14,000 *Giardia* cysts per liter of water in storm surface water below coastal California dairies (Miller *et al.* 2007). Cattle are also noted to carry the shiga toxin containing *E. coli* strain O157:H7 at a rate of 1 to 30%, which can be acquired from drinking partially treated or untreated water and cause illness and death in humans (Swerdlow *et al.* 1992; Renter *et al.* 2003). Shiga toxin containing *E. coli* may also be acquired from swimming, thus placing children who unknowingly play in the water downstream from remote grazing areas at risk for a disease (McCarthy *et al.* 2001). Studies on this strain have also shown it to survive in cold water so characteristic of high Sierra lakes and streams (Want & Doyle 1998). In addition as previously noted cattle manure contains high amounts of N, P and other growth factors for algae. These particulate and dissolved organic substances also create an aquatic environment that supports survival of pathogenic microorganisms (Horne & Goldman 1994; Miettinen *et al.* 1997; Jasson *et al.* 2006; Tao *et al.* 2007). Despite these human health concerns, the US Forest Service initially increased cattle grazing tracts in a Sierra Nevada Wilderness (USDA Forest Service 2006).

IMPACT TO WATERSHED GROUND VEGETATION

Livestock grazing and livestock grazing operations may severely disrupt sensitive ecological communities which in turn affect water quality (Belsky & Blumenthal 1997; Belsky *et al.* 1999). Some authors attribute significant impacts to "overgrazing" implying there is a level of livestock grazing that has less significant impacts (Allen-Diaz *et al.* 1999). Cattle degrade habitat by trampling and eating vegetation, compacting soils, impacting riparian systems, and affecting water quality. When livestock degrade habitat, they also impair the survival of many animal and some of the plant species upon which they depend. For example, aspen groves in the Sierra Nevada forests are rare but important areas of high biodiversity (Rogers *et al.* 2007) that enhance watershed capacity by storing seven times more water than conifers that have been changed in abundance and distribution by livestock grazing (Bartos & Campbell 1998). Kay & Bartos (2000) found that although elk and deer graze on aspen most herbivory of aspen was from livestock not from wildlife. Recent conservation recommendations include reintroduction of top predators to the Sierra Nevada (Rogers *et al.* 2007) but this would require an end to domestic livestock grazing. Aspen restoration has become a priority for California Department of Fish and Game's wildlife management and habitat conservation programs.

Livestock trampling has both direct and indirect effects on vegetation, soils and water runoff. (Abdel-Magid *et al.* 1987). The natural replacement of aged conifers is jeopardized, as new seedlings are trampled to death after germination. The Lens-pod Milk-vetch, *Astragalus lentiformis*, is a rare endemic plant that is only found in one district of Plumas National Forest in the northern Sierra Nevada range. The Forest Service has documented 55 occurrences of the Lens-pod Milk-vetch most of which are located in grazing allotments. Plants in the *Astragalus* family tend to be unpalatable to livestock but the Lens-pod Milk-vetch is susceptible to trampling and, as various Forest Service botanical evaluations admit, "The trend for this narrow endemic is unknown". Despite this, in the past 2 years the Forest Service has reauthorized cattle grazing on nine allotments that account for 49% of the known occurrences of the Lens-pod Milk-vetch without analyzing the cumulative impacts to the plant. Water runoff from snowmelt or rain through trampled areas carry eutrophic substances into lakes and streams.

Impacts to aquatic wildlife may occur at the individual and at the population level. On example is the Yosemite toad, which is a rare amphibian found in high elevation meadows in the central Sierra Nevada that is a candidate for listing under the Endangered Species Act (USDI 2002). Outbreaks of red-leg disease and infection with a Chytrid fungus have contributed to die-offs of Yosemite toad populations (Davidson & Fellers 2005). The occurrence of the toad in high altitude meadows that are National Forest rangeland puts individuals at risk of being trampled by the herds of grazing cattle that concentrate there. Small toads may even get trapped and die in deep hoof prints or under fecal matter. However, population level impacts may also occur. Alterations to meadow hydrology such as lowering of the groundwater table and summer flows can strand tadpoles or make breeding sites unsuitable; lowering of the water table in meadow habitat through stream incision resulting in breeding habitats drying out prior to metamorphosis of the tadpoles; cattle may negatively affect upland habitat through grazing and trampling of willows that are used for refuge, foraging, and over wintering; cattle may also trample and collapse rodent burrows that are used for over-wintering or seasonal refuge. Because cattle move between meadows, they may act as vectors to transmit infective pathogens between different populations. Spores of *Batrachochytrium dendrobatidis*, the fungus linked to the toad die-offs, can survive for at least 7 weeks in water (Johnson & Speare 2003). Livestock carrying mud on their hooves and moving between meadows are likely to spread the pathogenic fungus (Parris 2006). Because cattle routinely move between meadows and may be herded through an entire meadow system during the season, they could move the fungus between meadows leading to local extirpation of the species.

IMPACT OF CLIMATIC CHANGE

The American Society for Microbiology has become concerned about increasing surface water temperatures (Dixon 2008). Predicted increases in temperatures from climatic change will warm streams creating more favorable conditions for growth of toxic algae and pathogenic microorganisms (Coats *et al.* 2006). A number of studies have correlated increased water temperatures with increases in algae growth (Paerl & Huisman 2008). Toxins from species-specific algae have been implicated in waterfowl deaths and human illness and are not removed by standard municipal water disinfection processes (Falconer & Humpage 2005; Lopez-Rodas *et al.* 2008). Several researchers at the University of California, Davis have shown that surface water temperatures in the Sierra like many lakes in the Northern Hemisphere are increasing (Coats *et al.* 2006). Lake Tahoe's entire water column has increased one degree in the last 30 years and surface waters have warmed by four degrees. Climate models predict that the warming trend will continue. Visible algae in many High Sierra lakes and streams has increased over the past 20 years (Goldman & Derlet 2009, unpublished data). In a recent study of Lake Tahoe specifically, planktonic diatom numbers were found to have increased from 1982 to 2006, and after controlling for multiple factors, increased water temperature was shown to be the single factor behind this increased form of algae (Winder *et al.* 2009). This trend may only intensify the problems already related above by increasing the rate of eutrophication and providing an ideal environment for toxic cyanobacteria. Furthermore the predicted increase in rapid melting of the Sierra Nevada annual snow pack will harm ecosystems (Coats *et al.* 2006). In this regard, conifer shading is even more important to slow snowmelt and preserve the "snow pack reservoir" function of these mountains. The tramping of seedlings by cattle can prevent new conifer growth thereby reducing shading vegetation so soil and snow is exposed to direct solar radiation and rapid melting and runoff.

A PROPOSAL TO ENHANCE WATERSHED PROTECTION

We propose limiting summer-time cattle grazing in the Sierra Nevada Mountains on public lands to lower elevations. Our proposal is based on collective research as discussed above and the authors' observations on watershed geology, climate, precipitation, snowmelt, flora and fauna of the alpine regions of these mountains. Summer cattle grazing at the end of a five-year phase in period should be restricted to areas below 1,500 m elevation in the Central and Northern Sierra and 2,000 m elevation in the Southern Sierra. We define Southern Sierra as Sierra south of the Kings-Sequoia NP boundary by latitude, and land north of this as Central and North Sierra. To achieve this goal, a step-wise phase out should occur over a five-year period. As higher elevations are the most ecologically sensitive, it would be preferable if cattle could first be removed from grazing above 2,500 m in the Central and North, and 3,000 m in the South. Each succeeding year the elevation limits should be lowered 200 m until the final goal is achieved. Thus 5 years would have elapsed from initiation to achieving a phase out at these elevations. This will protect the most vulnerable and valuable portion of the Sierra Nevada watershed. In the Lake Tahoe basin, grazing has nearly been phased out with improvement in surface water quality flowing into the lake (Goldman 2008, unpublished data). Certain exceptions to the proposal may be reasonable, for example the large flatlands east of the Sierra crest such as the Bridgeport Valley east of Yosemite and Sierra Valley northwest of Reno. These two large grassland valleys have multiseason use from cattle and other agricultural usage.

As an alternative, the phase out of alpine grazing on public lands in the Sierra Nevada could be accomplished by a permit buyout process. Adoption of a moratorium on issuance of any new permits for currently vacant grazing allotments at altitudes above 2,000 m could be combined with a buy-out option for existing permittees in a voluntary relinquishment program. Funding for these buy-outs could come from federal land and Water Conservation funds, conservation organizations, mitigation agreements, and other federal, state and local government agencies. There are many examples of the success of such programs. For example, the California Desert Protection Act of 1994 allowed for the voluntary relinquishment of permittees to end livestock grazing in the expanded and newly created units administered by the National Park Service. Since that time, most of the permits have been acquired and retired largely through the activity of various conservation organizations. The recent Owyhee Initiative, signed into law in the Omnibus Appropriations Act of 2009, allows for buy out and voluntary relinquishment of grazing privileges to protect Wilderness Areas. While a buy-out process is likely to be slower and less coordinated, both the affected resources and the local ranching communities would benefit, creating a win/win scenario. Impacts to sensitive plants, animals and their habitats would be reduced, water quality enhanced, and ranchers would have the funding to move their operations to more appropriate and productive areas. The long-term cost savings to the Forest Service would be considerable.

Phase out proposals should be adopted as soon as possible to ensure long-term protection for this crucial source of water for California, which from recent reports may face the development of water shortages which will worsen in the face of global climate change.

CONCLUSION

Cattle have a negative impact on high elevation watersheds of the Sierra Nevada Mountains. Phasing out high elevation summer cattle grazing from source watersheds should improve water quality. Restricting cattle from the higher elevations will affect less than one of every hundred head of cattle in California. As a result the impact of this proposal on California's cattle industry would be relatively small and the potential benefits larger to the safety and health of children and adults in the State.

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ORIGINAL RESEARCH

Risk Factors for Coliform Bacteria in Backcountry Lakes and Streams in the Sierra Nevada Mountains: A 5-Year Study

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CME **Category 1 Continuing Medical Education credit for WMS member physicians is available for this article. Go to http://wms.org/cme/cme.asp?whatarticle=1922 to access the test questions.**

Objective.—To provide a 5-year longitudinal assessment of risk of acquiring disease from Sierra Nevada Wilderness area lakes and streams. This study examines the relative risk factors for harmful water microorganisms, using coliforms as an indicator.

Methods.—Streams and lakes in the backcountry of Yosemite and Kings Canyon National Parks and neighboring wilderness areas were selected and water was analyzed each year over a 5-year period. A total of 364 samples from lakes or streams were chosen to statistically differentiate the risk categories based on land usage, as follows: 1) areas rarely visited by humans (Wild), 2) human day-use-only areas (Day Hike), 3) areas used by backpackers with overnight camping allowed (Backpack), 4) areas primarily impacted by horses or pack animals (Pack Animal), and 5) cattle and sheep grazing tracts (Cattle). Water was collected in sterile test tubes and Millipore coliform samplers. Water was analyzed at the university microbiology lab, where bacteria were harvested and then subjected to analysis using standardized techniques. Statistical analysis to compare site categories was performed utilizing Fisher exact test and analysis of variance.

Results.—A total of 364 sampling sites were analyzed. Coliforms were found in 9% (4/47) of Wild site samples, 12% (5/42) of Day Hike site samples, and 18% (20/111) of Backpacker site samples. In contrast, 63% (70/111) of Pack Animal site samples yielded coliforms, and 96% (51/53) of samples from the Cattle areas grew coliforms. Differences between Backpacker vs Cattle or Pack Animal areas were significant at $P \leq .05$. All samples grew normal aquatic bacteria.

Conclusion.—Surface water from watersheds below cattle areas and those used by pack animals is at high risk for containing coliform organisms. Water from Wild, Day Hike, or Backpack sites poses far less risk for contamination by coliforms.

Key words: water, Yosemite National Park, Kings Canyon National Park, Sierra Nevada Mountains, cattle, Coliforms

Introduction

The Sierra Nevada Mountain Range in California serves as an internationally recognized recreational area and an important natural resource, in that it provides 50% of the state's drinking water.[1,2] The Sierra extends from Tehachapi Pass in the south 400 miles northward to Soldier Meadows, near Lassen National Park.[3] Much of the land still retains wilderness character, with roughly 4 000 000 acres of land designated as official wilderness by the National Park Service or the US Department of Agriculture (USDA) Forest Service, and is protected from development, logging roads, and motor vehicles.[4] Most

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of these protected areas range from 1800 to 4200 m in elevation. Surface-water quality at high-elevation headwaters is important to hikers, backpackers, and fishermen, as well as downstream urban water districts.[2,5] Non–point source pollution may result in contamination of surface waters with harmful substances, including both microbial organisms and toxic substances.[2] Therefore, the issue of potential microbial pollution from day hikers, backpackers, horses and pack animals, and commercial cattle and sheep grazing is important. Microorganisms include coliforms, pathogenic bacteria, and protozoa such as *Giardia* or *Cryptosporidium*.[6] Although concerns have been raised regarding *Giardia* in the Sierra, many authors have suggested that other fecal pathogens, such as enterotoxic *Escherichia coli*, may play a greater role in mountain-acquired illness.[6–10]

The unique geographic features of the Sierra have resulted in challenges to water ecology and quality. Much of the watershed consists of granite or metamorphic bedrock, with little topsoil.[11] As a result, soil buffering capacity is extremely low, providing little or no biogeochemical retention or transformation of nutrients such as nitrogen and phosphorus.[5] Relatively small amounts of nutrient addition or habitat disturbance can lead to significant impacts on nutrient flux and subsequent impacts on water quality and aquatic ecosystems.[12] Pollution from soap, sunscreens, food particles, and human and animal waste may enter the waterways. These substances include nutriments known to increase rates of surface-water eutrophication, in turn prompting conditions that lead to increased survival or growth of microorganisms such as bacteria and algae.[13–15]

Monitoring for each type of microorganism is expensive and difficult; this difficulty is compounded by the high alpine geography that requires multiple hiking days to access remote sites. As an alternative to testing for all microorganisms, testing for coliforms can provide an index of risk for pathogenic waterborne disease.[16,17] Coliform bacteria have been established as indicators of fecal pollution or contamination, including *Giardia*, of waterways in the United States.[17] In wilderness areas, coliforms may originate from one or a combination of sources including 1) wild animals endemic to the area; 2) humans visiting during daylight; 3) backpackers who camp overnight; 4) stock or pack animals, such as horses and mules; and 5) cattle or sheep grazing. Coliform pollution of wilderness areas by humans may occur through inadequate burial and disposal of fecal material. In addition, bathing or swimming in lakes may also result in microbial pollution.[18] Pack animals may pollute by deposition of manure either directly into lakes and streams or indirectly by deposition of manure onto trails or meadows, and these animals have been documented to import *Giardia* into the Sierra wilderness.[19,20] This manure may be washed into waterways by either summer storms or annual snowmelt.[21,22] The USDA Forest Service leases tracts in wilderness areas for cattle grazing.[23] Both cattle and pack animal manure are known to potentially contain microbes that are pathogenic to humans, including viruses; protozoa such as *Giardia* and *Cryptosporidium*; and bacteria such as *E coli* and *Salmonella*.[24–27] Finally, some coliform and other bacteria potentially may originate from natural wild animal and bird zoonotic reservoirs.[28]

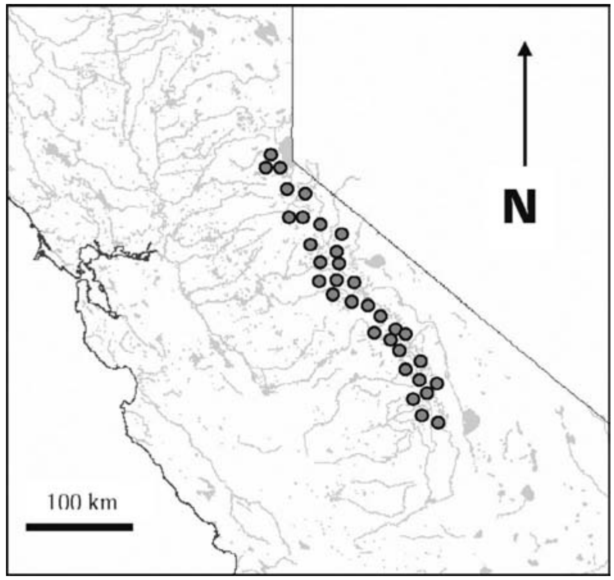

Figure. Study area and sample collection sites. Sites were located throughout the majority of the Sierra Nevada range. In some cases each dot represents more than one sampling site because some sites were too close to display individually.

We have surveyed the surface water of Sierra Nevada wilderness areas during selected summers in past years, but debate still continues regarding the impact of backpackers, cattle grazing, or livestock on the watersheds in wilderness areas.[23] In this report, we use results from previously published surveys (years 2003 through 2006) and combine them with new results reported here to create a continuous 5-year data set.[29–31] The goal of this paper is to determine the relationship between land use patterns and the prevalence of coliforms in the Sierra Nevada surface water.

Methods

FIELD SITE SELECTION

Sites were selected that include all common types of land use in wilderness areas of Kings Canyon, Sequoia, and Yosemite National Parks, as well as the following

USDA Forest Service wilderness areas: Carson-Iceberg, Emigrant, Hoover, and John Muir (the Figure). The Hall Natural Research Area, adjacent to the eastern boundary of Yosemite and the southern boundary of the Hoover wilderness, was also included. No overnight camping or motor vehicles are allowed in the Hall area. Sites were selected randomly from areas representative of different use patterns. Relative differences in the number of sites in each category reflect the prevalence of land use patterns along the various trails. Risk classifications included 1) natural areas not visited by humans or domesticated animals (Wild); 2) day hike areas used only by humans and in which overnight camping was not allowed (Day Hike); 3) areas used by backpackers with overnight camping allowed (Backpacker); 4) areas traversed by animals such as horses and mules (Pack Animal); and 5) cattle and sheep grazing tracts (Cattle). Site characteristics were stratified with the assistance of the National Park Service and the USDA Forest Service based on use described by the risk classifications of this study. Cattle grazing is not permitted in National Parks, so all samples in cattle grazing tracts were taken from within Forest Service wilderness areas.

FIELD WATER COLLECTION

Water samples were collected from June through September for the 5-year period ranging from 2002 to 2006. For sites subject to repeated analysis, samples were taken during the same week each year. Water was not collected within 3 days of thundershowers to prevent skewing of results from trail runoff. Samples were not taken in the real-time visible presence of pack animals or cattle. Water was collected in 1) sterile test tubes, 2) Millipore total coliform count samplers (Millipore Corporation, Bedford, MA), and 3) Millipore heterotrophic bacteria count samples. All samples were collected in duplicate. Although the manufacturer suggests immediate incubation, this was not possible as a result of the remote wilderness conditions of the study. Our control studies have shown that colony survival is not affected for up to 1 week at temperatures below 30°C, a condition to which we adhered in the field by monitoring the temperature of the sample container and returning to the laboratory within 7 days of all sampling (R. W. Derlet, MD, unpublished data, May 2002). To prevent deterioration from higher temperatures during transport from trailhead to laboratory (a trip taking, on average, 8 hours), samples were kept in a cooler at 5°C. Each sample device measured bacteria for 1 mL of sample. This was multiplied $\times 100$, as per standardized procedure of reporting colony-forming units (CFU)/100 mL in the water literature.[17,30] The mean value of duplicate samples is reported. Water temperature was measured at each site using a stream thermometer (Cortland Line Company Inc, Cortland, NY). Location and elevation were determined using US Geographical Society topographical maps, guide books, and backcountry rangers.

ANALYSIS OF WATER SAMPLES

Details of analysis for bacteria have been described in detail elsewhere.[28,29,32] The analysis for coliform counts and total bacterial counts required incubating Millipore counting plate paddles at 35°C for 48 hours. Bacterial colonies were counted, then harvested and subplated for further analysis, following standardized procedures.[32] Colonies were plated onto Sheep Blood, MacConkey, and Sorbitol agars (Reel Inc, Lenexa, KS). Lactose fermenting colonies from MacConkey plates were presumed to be coliform bacteria and were subject to further testing. Further screening and initial identification was done by subplating onto Eosin Methylene Blue (EMB Levine), Cefsulodin Irgasan Novobiocin, and Hektoen agars. The color and morphology of the colonies were recorded. Controls and samples, including coliform-inoculated and coliform-free water, were subjected to simulated field conditions and tested to provide quality assurance of methods.

DATA ANALYSIS

The entire data set was analyzed to compare the results of water analysis to the different land use patterns. A subset of sites that had been subject to an annual analysis for at least 4 of the 5 years was analyzed separately to determine if these specific sites produced consistent results each year. Coliform-positive samples were correlated with water temperature and elevation. For this purpose, very low temperature was arbitrarily categorized as 0°C to 10.9°C, low as 11°C to 15.9°C, mild as 16°C to 20.9°C, and warm as 21°C and higher. Elevation was compared in 500-m intervals from 2000 m to 3500 m. Statistical significance between groups was calculated with Fisher exact test and analysis of variance (ANOVA) utilizing STATA Software (College Station, TX). Data are reported with 95% confidence intervals, unless otherwise stated.

Results

Sample sites are illustrated in the Figure, and results are summarized in Tables 1 through 6. A total of 364 samples were collected from 105 different streams or lake sites. Coliforms were found in 4 of 47 Wild sites (8.5%, CI 1.8–15.2), 5 of 42 Day Hike sites (11.9%, CI 3.1–

Table 1. Percentage of coliform-positive sites by land use and raw data (positive sites/total sites)

Land use	*2002*	*2003*	*2004*	*2005*	*2006*	*Totals*
Wild sites	25 (1/4)	0 (0/4)	7 (1/15)	18 (2/11)	0 (0/13)	9 (4/47)
Day hiker	0 (0/5)	25 (2/8)	17 (1/6)	18 (2/11)	0 (0/12)	12 (5/42)
Backpack	18 (6/34)	22 (7/23)	7 (1/15)	14 (3/21)	17 (3/18)	18 (20/111)
Pack animals	66 (12/18)	55 (18/33)	80 (12/15)	56 (14/25)	70 (14/20)	63 (70/111)
Cattle	100 (7/7)	88 (7/8)	100 (15/15)	92 (13/14)	100 (9/9)	96 (51/53)
Totals	38 (26/68)	45 (34/76)	45 (30/66)	39 (32/82)	36 (26/72)	41 (150/364)

18.9), and 20 of 111 backpacker sites (18.0%, CI 12.0–24.0). In contrast, 70 of 111 Pack Animal sites (63.1%, CI 55.5–70.5) yielded coliforms, and 51 of 53 Cattle sites (96.2%, CI 91.5–100) grew coliforms. The differences between Wild, Day Hike, or Backpacker and either Pack Animal sites or Cattle sites were statistically significant ($P \geq .05$, Fisher exact test).

With regard to temperature, 9 of 23 samples at very low temperature were positive (39.1%, CI 12.2–66.8), and 59 of 158 samples at low temperatures were positive (37.3%, CI 17.9–38.2). For mild temperatures, 65 of 160 samples were positive (40.6%, CI 29.9–51.3), and 2 of 5 samples from warm temperatures were positive (40.0%, CI 4–76). There was no significant difference between coliform growth and temperature range ($P =$ .56, ANOVA). For elevations between 2000 and 2499 m, 24 of 51 samples were positive (47.0%, CI 27.0–67.0), and for elevations between 2500 and 2999 m, 60 of 162 samples were positive (37.0%, CI 24.3–49.7). For elevations above 3000 m, 66 of 151 samples were positive (43.7%, CI 30.4–57.0). No significant difference in coliform growth and elevation range was detected ($P =$ .57, ANOVA). Coliform counts in positive samples ranged from 100 to 500 $CFU \cdot mL^{-1}$.

Subanalyses performed on sites that were sampled at least 4 of the 5 years are listed in Tables 2 through 6. These sites were sampled at similar times during 4 of 5 summers. A total of 58 of these sites provided 246 samples for analysis. Coliforms were found in a similar frequency when compared to the total analysis. In this subanalysis, coliforms were found in 2 of 38 Wild samples (5.0%, CI 0–11), 3 of 42 Day Hike samples (7.1%, CI 0.6–13.6), 11 of 62 Backpacker samples (17.7%, CI 9.2–24.9), 40 of 65 Pack Animal samples (61.5%, CI 51.5–70.9), and 35 of 37 Cattle samples (94.5%, CI 87.6–100).

Heterotrophic bacteria were also identified from the samples. Concentrations ranged from 400 to 12 200 CFU/100 mL. Although not statistically significant, total bacterial counts for positive samples tended to be lower at the Wild and Day Hike sites, with a combined mean of 2333 CFU/100 mL (CI 1562–3105), compared with 5248 CFU/100 mL (CI 2838–7650) for Backpacker sites, 5819 CFU/100 mL (CI 3010–8628) for Pack An-

Table 2. Wild sites: Number of coliforms at each site by year (colony-forming units [CFU]/100 mL)

Wilderness area	*Place*	*Elevation, m*	*2002*	*2003*	*2004*	*2005*	*2006*
Yosemite	Johnston Pass Creek	2780	100	None	*	None	None
Yosemite	Raymond Pass Creek	2943	None	100	*	None	None
Yosemite	Upper Yosemite Creek—Side Creek	2501	None	None	None	None	None
Yosemite	Hoffmann Creek	2560	None	None	*	None	None
Yosemite	Upper Middle Dana-Gibbs Creek	3016	None	None	None	None	None
Kings Canyon	Bago Springs Creek	2840	*	None	None	None	None
Kings Canyon	Spring, north of Glen Pass JMT†	3353	*	None	None	None	None
Kings Canyon	Creek above Rae Lake Ranger Station	3231	*	None	None	None	None
Kings Canyon	Creek draining Lake 10 315	2768	*	None	None	None	None

*No data.
†John Muir Trail.

Table 3. Day hike only sites: Number of coliforms at each site by year (colony-forming units [CFU]/100 mL)

Wilderness area	*Place*	*Elevation, m*	*2002*	*2003*	*2004*	*2005*	*2006*
Yosemite	Budd Creek	2622	None	*	None	200	None
Yosemite	Gaylor Lake	3150	None	*	None	None	None
Yosemite	Upper Gaylor Creek	3155	None	*	None	None	None
Yosemite	Lower Gaylor Creek	2835	None	*	None	None	None
Yosemite	Granite Lake	3176	None	*	None	None	None
Yosemite	North Fork Tuolumne River, headwaters	2438	*	None	None	None	None
Yosemite	Dana Fork of Tuolumne River	2941	100	None	None	200	None
Kings Canyon	Bull Frog Lake	3231	*	None	None	None	None
Emigrant	Blue Lake Creek	3048	*	None	None	None	None
Hall Area	Green Treble Lake—lower	3010	None	None	None	None	None

*No data.

imal sites, and 5732 CFU/100 mL (CI 2947–8517) for Cattle sites.

Field collection observations confirmed the characterization of land use categories. Wild areas had no trails or visible evidence of human or domesticated animal use upstream of the sampling site; Day Hike areas were posted as such or were posted with ‘‘No camping’’ signs. Backpacker areas had no evidence of recent or remote pack animal manure on trails, but they did show evidence of campsites. Pack Animal areas had animal manure on the trails, and in Cattle areas cow pies were observed in meadows and woodland. No manure was observed directly in lakes or streams at the time of sampling.

Discussion

In our 5-year analysis, overall consistency was found each year with respect to the prevalence of coliforms overall and also in each designated land use area. This consistency and reproducibility of results is an important finding of this 5-year analysis and has implications for validating single-year data. Total coliform prevalence ranged from 36% to 45% each year. Total annual precipitation was similar each of the years sampled, with no drought years.[33] Only a few other studies have examined backcountry water in the Sierra, providing few data with which to compare our findings.[7–9] We believe that analyzing the data by land use areas provides a useful prospect of impact on water quality.

Table 4. Backpacking sites: Number of coliforms at each site by year (colony-forming units [CFU]/100 mL)

Wilderness area	*Place*	*Elevation, m*	*2002*	*2003*	*2004*	*2005*	*2006*
Yosemite	Yosemite Creek	2278	None	100	None	None	None
Yosemite	Booth Lake	3001	*	100	None	None	None
Yosemite	Townsley Lake	3154	*	None	None	None	None
Yosemite	Vogelsang Lake	3147	*	None	None	None	100
Yosemite	Ten Lakes #2	2813	None	None	*	None	None
Yosemite	Ten Lakes #3	2750	None	None	*	None	None
Yosemite	Ten Lakes #4	2727	100	None	*	300	400
Yosemite	East Ten Lakes	2865	None	None	*	None	None
Kings Canyon	East Creek at confluence of Bubbs Creek	2494	*	100	None	None	None
Kings Canyon	Charlotte Creek	2219	None	100	200	100	None
Kings Canyon	Charlotte Lake near ranger station	3165	*	None	None	None	None
Kings Canyon	Upper Rae Lake	3213	*	None	None	None	None
Kings Canyon	60 Lakes Drainage Creek	2926	*	100	None	None	None
Kings Canyon	South Fork Kings River at Upper Paradise	2134	*	None	None	None	None
Kings Canyon	North Fork Woods Creek	2621	*	None	None	None	None

*No data.

Table 5. Pack animal sites: Number of coliforms at each site by year (colony-forming units [CFU]/100 mL)

Wilderness area	*Place*	*Elevation, m*	*2002*	*2003*	*2004*	*2005*	*2006*
Yosemite	Tuolumne River (Lyell Canyon)	2804	200	100	200	None	200
Yosemite	Rafferty Creek	2673	100	None	*	100	100
Yosemite	Fletcher Lake	3095	700	None	None	None	None
Yosemite	Fletcher Creek	3060	500	100	100	100	None
Yosemite	Dog Lake	2804	100	200	*	100	100
Kings Canyon	Bubbs Creek at confluence of Kings River	1560	100	None	*	None	None
Kings Canyon	Bubbs Creek at Junction Meadow	2469	200	None	*	None	200
Kings Canyon	Bubbs Creek at Vidette Meadow	2896	100	None	*	200	None
Kings Canyon	Arrow Lake	3154	*	100	350	None	None
Kings Canyon	Arrow-Dollar Creek Trail Crossing	3145	*	100	200	None	100
Kings Canyon	Dollar Lake	3115	*	100	None	100	300
Kings Canyon	Rae Lake (middle)	3211	*	None	None	None	200
Kings Canyon	South Fork Kings at Lower Paradise	2011	0	100	500	100	300
Kings Canyon	Copper Creek	1555	100	100	300	None	None
Kings Canyon	Lewis Creek	1219	200	100	*	200	None

*No data.

CATTLE AREAS

We have found that areas frequented by cattle had the greatest degree of coliform contamination into the wilderness watershed, ranging from a prevalence of 88% to 100% for each year sampled over the 5-year period. We are not surprised at the finding of coliforms below cattle grazing areas. On traditional US rangelands, coliforms can be expected to be found in the watershed.[34] A recent study of South Carolina watersheds found non-point pollution with *E coli* to be high in cattle grazing areas.[35] In some respects, finding coliforms below grazing areas serves as a positive control for the study. However, until recently, data on the impact of cattle on Sierra water have been limited.[30] Cattle harbor and excrete many microorganisms capable of causing disease in humans, including protozoa, bacteria, and viruses.[25–27] Miller and colleages[36] found up to 14 000 *Giardia* cysts per liter of water in storm surface water below coastal California dairies. Cattle are also noted to carry *E coli* strain O157: H7 at a rate of 1% to 30%, placing persons who drink untreated water below established cow pastures at risk for very serious disease.[26] Studies on this strain have also shown it to survive in cold water.[37] In addition, cattle manure contains large amounts of nitrogen, phosphorus, and other growth factors for algae.[14] These substances also create an aquatic environment that supports pathogenic microorganisms.[12–15] Each wilderness ‘‘cow use day’’ is equivalent to 100 to 120 human use days in terms of environmental impact with respect to waste pollution.[38,39] Despite these concerns, the US Forest Ser-

Table 6. Cattle risk watershed sites: Number of coliforms at each site by year (colony-forming units [CFU]/100 mL)

Wilderness area	*Place*	*Elevation, m*	*2002*	*2003*	*2004*	*2005*	*2006*
Carson	Upper Clark Fork River	2072	*	100	250	None	400
Carson	Lower Clark Fork River	2316	*	100	300	100	600
Carson	Disaster Creek	2366	*	200	350	300	550
Carson	Arnot Creek	2000	*	100	100	200	100
Carson	Woods Creek	1976	*	100	100	250	100
Emigrant	Kennedy Creek	2244	*	None	*	300	200
Hoover	Buckeye Creek	2377	200	200	500	300	450
Hoover	Molydunite Creek	2773	100	300	400	300	200
Hoover	South Fork Walker River (Burt Canyon)	2719	None	200	250	200	200

*No data.

vice has recently increased proposed cattle grazing tracts in the Sierra Wilderness.[23]

PACK ANIMAL–IMPACTED AREAS

The finding of a high prevalence of coliforms in wilderness areas frequented by pack animals is important. Very few other studies have attempted to analyze land use patterns and risk for finding pathogenic microorganisms in the high-elevation areas of the Sierra Nevada.[8,9] A report on the Rae Lakes region of Kings Canyon National Park found that water from lakes and streams with higher human activity tended to have a higher prevalence of coliforms.[8] However, these areas were also subject to pack animal traffic. In that study, lakes and streams found free of coliforms were inaccessible to horses and mules. Pack animals produce high volumes of manure, which is deposited directly onto the surface of trails, soil, or meadows.[24,38,40] In contrast to human waste, pack animal manure is not buried in the soil. Manure deposited on the ground can be swept into streams during summer rains or spring snow runoff.[21,22] The National Park Service is concerned about manure contamination of surface waters because of its effect on water.[40,41] Fecal contamination, as indicated by the finding of coliforms, would place the watershed at risk for harboring microbes capable of causing human disease. As is the case with cattle, these threats include certain pathogenic strains of *E coli*, *Salmonella*, *Campylobacter*, *Aeromonas*, and protozoa such as *Giardia*. Pack animals entering the High Sierra have been subject to analysis, and *Giardia* has been found in their manure.[20] The organism *Hafnia alvei* was found in one study conducted along the John Muir Trail in the Sierra Nevada, even in old manure.[24] *H alvei* can cause diarrhea in humans.[42] The pack animal areas studied were also traversed by humans. Therefore, it is possible that some of the coliforms found at these sites originated from humans. An examination of results from the Backpack sites helps to clarify this issue. In comparison to Pack Animal sites, only a small percentage of Backpacker sites had coliforms. This finding would support the conclusion that most of the microbial contamination in pack animals areas is a result of pack animal manure. Furthermore, in Day Hike areas in which pack animals are not allowed to travel, only low levels of coliforms were found.

BACKPACK-ONLY SITES

Coliform was found in an average of 18% of these sites. Wilderness regulations require that human waste must be buried at least 100 feet from waterways.[40,41] Discussions with wilderness backcountry rangers indicate that there is generally good compliance with these regulations. When disposed of properly in humus topsoil, which contains a multitude of bacteria and fungi, these environmental microbes degrade many of the pathogens. Some Wilderness areas now also ask backpackers to carry out their toilet paper.

WILD SITES

In contrast to the other site types, coliforms were found in only 9% of Wild sites. The source of coliforms found in the wild is speculative. Coliforms may be present as a result of waste contamination from the many species of birds and native mammals. Environmental coliforms have been reported in the environmental literature.[43]

Heterotrophic, aquatic bacteria are part of a normal ecosystem of lakes and streams.[44] Indeed, if bacteria were absent, the normal food chain from frogs to fish, as well as the ecological balance, would be in jeopardy. A prior study identified many species, including *Achromabacter* species, *Pasteurella haemolytica*, *Rahnella* species, *Serratia* species, *Yersinia intermedia*, *Yersinia* species, and *Pseudomonas* species in wilderness surface water.[29] We found total bacterial counts to be lower at Wild and Day Hike sites, compared to other categories in this 5-year analysis. This may result from the effects of camping, which include the deposition of bacteria from skin contact into surface water and also the stirring up of bacteria-rich bottom sediment in lakes and streams.[39]

LIMITATIONS

Multiple confounding factors may affect wilderness field findings. Annual precipitation varied during the years of the study. Wind, water flows, and cloud cover may affect results. Although samples were taken during summertime traffic by humans and domesticated animals, these represent single–point-in-time samples; additional samples at different times may have increased the accuracy and significance of findings. Data in this report are applicable only to Sierra Nevada Wilderness Areas and not to areas with human habitation. Finally, overall use patterns were not quantified (backpacker use in terms of persons/night; animal use in terms of heads of livestock/acre, etc).

RECOMMENDATIONS

In wilderness areas where cattle or pack animals have been present, we recommend that drinking water be treated. In Sierra Nevada wilderness areas, water from alpine sidestreams that are free from upstream domesticated animal use have a very low risk of harboring coliforms and we believe have a minimal risk of illness if drunk untreated.

Conclusion

In this 5-year analysis, coliform prevalence in Sierra Nevada Alpine wilderness water varied by land-usage patterns of humans and domesticated animals. Water in areas of cattle grazing or in areas used by pack animals has a high probability of containing coliform organisms. Water from lakes and streams of Wild, Day Hike, or Backpack watersheds bears significantly less risk of harboring coliforms.

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ORIGINAL RESEARCH

Coliform Bacteria in Sierra Nevada Wilderness Lakes and Streams: What Is the Impact of Backpackers, Pack Animals, and Cattle?

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Objective.—The presence of coliform bacteria indicates a watershed risk for harboring microbes capable of causing human disease. We hypothesized that water from watersheds that have different human- or animal-use patterns would have differing risks for the presence of coliform bacteria.

Methods.—Water was collected in wilderness areas of the Sierra Nevada range in California. A total of 60 sites from lakes or streams were selected to statistically differentiate the risk categories: 1) high use by backpackers, 2) high use by pack animals, 3) cattle- and sheep-grazing tracts, and 4) natural areas rarely visited by humans or domestic animals. Water was collected in sterile test tubes and Millipore coliform samplers during the summer of 2004. Water was analyzed at the university microbiology lab, where bacteria were harvested and then subjected to analysis by standardized techniques. Confirmation was performed with a Phoenix 100 bacteria analyzer. Statistical analysis to compare site categories was performed with Fisher exact test.

Results.—Only 1 of 15 backpacker sites yielded coliforms. In contrast, 12 of 15 sites with heavy pack-animal traffic yielded coliforms. All 15 sites below the cattle-grazing areas grew coliforms. Differences between backpacker and cattle or pack-animal areas were significant ($P \leq .05$). Only 1 of the 15 wild sites rarely visited by humans grew coliforms. All coliforms were identified as *Escherichia coli*. All samples grew normal aquatic bacteria of the genera *Pseudomonas*, *Ralstonia*, and *Serratia* and nonpathogenic strains of *Yersinia*. No correlation could be made with temperature or elevation. Sites below cattle-grazing tracts and pack-animal usage areas tended to have more total bacteria.

Conclusions.—Alpine wilderness water below cattle-grazing tracts or areas used by pack animals are at risk for containing coliform organisms. Areas exclusively used by backpackers were nearly free of coliforms.

Key words: water, Yosemite National Park, Kings Canyon National Park, Sierra Nevada, *Escherichia coli*

Introduction

The Sierra Nevada range snowpack serves as an important water source for California; its watershed provides nearly 50% of the state's freshwater supply.[1] It is important that this watershed be protected from microbial, chemical, and toxic pollution for users both downstream and upstream.

Within the Sierra Nevada range, over 3 000 000 acres of land have been designated as official wilderness by the National Park Service or United States Department of Agriculture (USDA) Forest Service and protected from development, logging roads, and motor vehicles.[2,3] Some wilderness areas have quotas to limit overnight camping by backpackers and use by pack animals. Most of these protected areas are in high alpine regions between 2000 and 4200 m in elevation. These high alpine lakes and streams are an especially important watershed for California because of presumed purity of water and a large quantity of precipitation in the form of snow. The water is important for not only the distant water users but also the local water users such as backpackers, campers, fishermen, and the USDA Forest Service and

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National Park Service. However, this land is potentially subject to pollution by day hikers, backpackers, horses and pack animals, and also commercial cattle and sheep grazing. Pollution may occur from potential harmful substances that include microbial organisms or toxic substances.[4] Microbial organisms that may cause illness in humans include pathogenic bacteria such as coliforms and protozoa such as *Giardia* or *Cryptosporidium*.[5] Chemicals or toxins may be imported or synthesized by microbes, zooplankton, or phytoplankton from precursors imported by humans. Debate has ensued on the impact of backpackers, cattle grazing, or livestock such as mules and horses polluting the watersheds in wilderness areas. We completed 2 studies in a previous year that surveyed remote Sierra Nevada lakes and streams.[6,7] However, these studies did not provide the statistical power to show significant differences for risk factors. This current study was designed to provide a direct comparison of risk factors.

Coliform bacteria have been established as indicators of fecal pollution or contamination of waterways in the United States.[8,9] Coliforms may originate from a single source or a combination of sources: 1) backpackers, 2) pack animals, 3) grazing animals (cows, sheep), and 4) wild animals. Coliform pollution of wilderness areas by humans occurs through inadequate burial and disposal of fecal material. In addition, bathing or swimming in alpine lakes may also result in microbial pollution.[9] Pack animals may pollute by deposition of manure either directly into lakes and streams or indirectly onto trails or meadows, from which it may be washed into waterways by summer storms and annual snowmelt. The USDA Forest Service ‘‘leases’’ tracts in wilderness areas for cattle grazing.[2] As a result, a high density of cattle manure may be found in certain alpine watersheds, either in meadows or as a result of direct deposit into streams or lakes. Finally, coliform or other bacteria may originate from natural, wild animal zoonotic reservoirs.

We hypothesized that wilderness freshwater from watersheds that have different human- or animal-use patterns would have differing risks for the presence of coliform bacteria. Therefore, the purpose of the study was to analyze wilderness freshwater samples for coliforms and compare results from watersheds that have different use patterns among the following groups: 1) backpackers, 2) horses and mules (pack animals), 3) cattle grazing, and 4) isolated areas affected only by natural wild animals.

Methods

FIELD SITE COLLECTION

Sixty sites were prospectively selected to differentiate among environmental risks for different types of bacterial contamination in wilderness areas of Kings Canyon National Park, Sequoia National Park, and Yosemite National Park as well as the following USDA Forest Service wilderness areas: Mokelumne, Carson-Iceberg, Emigrant, Hoover Wilderness, Adams, John Muir, and Golden Trout. The Hall Natural Research Area, adjacent to the eastern boundary of Yosemite National Park and the southern boundary of Hoover Wilderness, was also included. No overnight camping or motor vehicles are allowed in the Hall area, and the remote areas have minimal visits by humans.

Risk classifications included 1) high use by backpackers, 2) high use of pack animals, 3) cattle-grazing tracts, and 4) natural sites (wild ecologies) not likely contaminated by humans or domesticated animals. Sites were risk stratified with the assistance of the National Park Service and USDA Forest Service on the basis of user nights by backpackers, pack animals, and cattle allotments in grazing tracts. Cattle grazing is not permitted in national parks, so all samples in cattle-grazing tracts were taken from within USDA Forest Service wilderness areas.

FIELD WATER COLLECTION

Water samples were collected from May through September in 2004. Water was collected in sterile test tubes and Millipore total coliform count samplers (Millipore Corporation, Bedford, MA). All samples were collected in duplicate, cooled according to standardized procedures, and transported to the University of California, Davis.[10] Sample devices measured bacteria for 1 mL of sample. This was multiplied by 100 as per standardized procedure of reporting colony-forming units per 100 mL in the water literature. Water temperature was measured at each site with a stream thermometer (Cortland Line Company Inc, Cortland, NY).

BACTERIAL ANALYSIS OF WATER SAMPLES

Details of analysis for bacteria have been described elsewhere.[6,7] The analysis for coliform counts and total bacterial counts required incubating Millipore counting plate paddles at 35°C for 24 hours. Bacterial colonies were counted and then harvested for further analysis. Colonies were initially plated onto sheep blood and MacConkey agars (Remel Inc, Lenexa, KS). Lactose fermenting colonies from MacConkey plates were presumed to be coliform bacteria and were subject to further testing. Further screening and initial identification was performed by subplating onto C.I.N. (*Yersinia*) agar, Sorbitol-MacConkey agar, L.I.A., and T.S.I. tubes. Precise identification of bacteria genera and species analysis

Table 1. Sites with heavy backpacking*

Wilderness area	*Place*	*Elevation (m)*	*Temperature (℃)*	*Escherichia coli CFU/100 mL*	*Other bacteria CFU/100 mL*
Yosemite	Yosemite Creek	2278	11.1	None	200
Yosemite	Budd Creek	2701	7.8	None	600
Yosemite	Townsley Lake	3154	13.3	None	5200
Emigrant	Wire Lakes	2694	19.4	None	3800
Emigrant	Blue Lake	3048	17.8	None	1100
Mokelumne	Round Top Lake	2834	17.2	None	800
Kings Canyon	East Lake	2493	13.9	None	6400
Kings Canyon	North Fork Woods Creek	2621	11.1	None	1900
Kings Canyon	South Fork Kings River (Upper Basin)	3078	12.2	None	4400
John Muir	Chicken Foot Lake (Little Lakes Valley)	3288	11.6	200	2900
John Muir	Ruwau Lake	3366	12.2	None	4100
Golden Trout	Chicken Spring Lake	3429	15.6	None	4600
Sequoia	Upper Rattlesnake Creek	3169	14.4	None	1100
Sequoia	Kern River	2031	16.7	None	3800
Desolation	Meeks Creek	2133	17.8	None	8900

*CFU indicates colony-forming units.

were performed by standardized automated laboratory procedures. In addition, analysis was also performed with a Phoenix 100 bacteria autoanalyzer. Strains were grown on Colombia agar with 5% sheep red blood cells for 16 to 24 hours at 37°C, replated, and grown again for 16 to 24 hours at 37°C just before testing. A suspension of 0.5 McFarland (accepted range, 0.5–0.6) was prepared in the identification (ID) broth (Becton Dickinson, Erembodegem, Belgium) and poured within 30 minutes into the panel, which was then loaded into the instrument within 30 minutes. Four quality-control strains (*Escherichia coli* ATCC 25922, *Klebsiella pneumonia* ATCC 13883, *Klebsiella pneumoniae* ATCC 700603, and *Pseudomonas aeruginosa* ATCC 27853) were loaded with each study batch, which always met quality-control criteria. The Phoenix instrument gives an ID result when a species or group of species is identified with more than 90% confidence. The confidence value is a measure of the likelihood that the issued ID is the only correct ID. The average time required to reach an ID result ranged from 3 to 12 hours. The autoanalyzer provided a computer printout identifying the bacteria. *E coli* colonies were also subjected to analysis to determine the presence of *E coli* O157 by using latex agglutination methodology.

Statistical significance among groups was calculated with Fisher exact test by STATA 8 Software (STATA Corporation, College Station, TX).

Results

The results are summarized in Tables 1 through 4. Significant differences were found among sample groups. All 15 samples that were taken below areas in which cattle grazed or had recently grazed were positive for coliform growth. From areas frequented by pack animals, 12 of 15 samples had coliforms. In contrast, coliforms were found in only 1 of 15 areas of heavy backpacking. Only 1 of 15 sites rarely visited by humans or pack animals contained coliforms. Backpacker and natural-site groups had significantly fewer sites with coliforms when compared with the cattle-grazing group ($P \geq .01$). Likewise, the pack-animal group had significantly more sites with coliforms when compared with the backpacker and natural areas ($P \geq .05$). No statistical differences were found in numbers of coliform bacteria according to water temperature or elevation.

Noncoliform aquatic bacteria were also identified from the samples. The most common bacteria found included *Achromabacter* species, *Pasteurella haemolytica*, *Rahnella aquatilis*, *Ralstonia paucula*, *Serratia odorifera*, *Serratia plymthica*, *Yersinia intermedia*, *Yersinia kristensenii*, *Yersinia frederiksenii*, *Pseudomonas putida*, and *Pseudomonas fluorescens*. No correlation could be made between site use and types of noncoliform bacteria or total bacteria counts, except for the Hall Natural Research Area, where the total bacteria range was the lowest of any group of samples. Total bacteria in the Hall

Table 2. Sites with stock (horses and pack animals)*

Wilderness area	*Place*	*Elevation (m)*	*Temperature (°C)*	*Escherichia coli CFU/100 mL*	*Other bacteria CFU/100 mL*
Hoover	W. Walker River	2262	11.1	250	3100
Emigrant	Horse/Cow Meadow Stream	2686	10.0	200	3000
Emigrant	Grouse Lake inlet stream	2179	5.0	550	2500
Emigrant	Piute Creek—Groundhog Meadows	2286	7.8	300	2000
Emigrant	Spring Meadow Creek	2590	23.3	900	10 000
Kings Canyon	Arrow Lake	3154	17.2	350	2100
Kings Canyon	Kings River—Paradise Valley	1981	14.4	500	1500
Yosemite	Fletcher Lake	3095	15.0	None	5800
John Muir	Long Lake (Bishop Pass Trail)	3277	12.2	150	5000
John Muir	Rock Creek at Wilderness Boundary	3154	11.1	300	8200
Yosemite	Tuolumne River (Lyell Canyon)	2804	16.1	200	3000
Kings Canyon	Dollar Lake	3115	17.2	None	1800
Kings Canyon	Rae Lake (middle)	3211	16.7	None	3100
Golden Trout	Horseshoe meadow	3017	10.0	300	1500
John Muir	Cottonwood lakes	3383	8.9	200	10 000

*CFU indicates colony-forming units.

Natural Research Area ranged from 200 to 500 per 100 mL. Temperature or elevation was not a factor, as other sites with similar temperature and elevation had higher baseline levels of aquatic bacteria. The marked absence of human impact distinguished this area.

Discussion

In this study, areas frequented by cattle or pack animals had the greatest degree of fecal contamination into the wilderness watershed. We are not surprised at the finding of coliforms below cattle-grazing areas. In most of these areas, moderate amounts of cattle manure were observed during field collections. We identified all coliforms in our study as *E coli*. In some respects, finding coliforms below grazing areas serves as a positive control for the study. One might expect coliforms in watersheds with high densities of cattle.[11] However, we are surprised at the finding of coliforms in areas frequented by pack animals. National parks and the USDA Forest Service have strict requirements on management of livestock in wilderness areas. It is not possible to exclude a human contribution to this finding, as high-volume pack-animal areas are also used by humans. In previous years we have examined Sierra Nevada water for coliform bacteria.[6,7] However, those studies were from water taken primarily from watersheds polluted by both pack animals and humans, and we were unable to fully determine associated risks for coliform pollution. This current study identified and included sampled sites used exclusively by backpackers and not pack animals. In addition, this current study added sites that were unused by humans, pack animals, or cattle. The absence of coliforms in most of those areas used exclusively by humans and the absence of pack animals would suggest that pack animals are most likely the source of coliform pollution. Pack animals produce high volumes of manure, which is deposited directly onto the surface of trails, soil, or meadows.[12,13] Manure deposited on the ground may be swept into streams during summer rains or spring snow runoff. During the field operations of the study, pack animals were observed on several occasions to be defecating directly into bodies of freshwater. Fecal contamination as indicated by the finding of coliforms would place the watershed at risk for harboring microbes capable of causing human disease. Some of these infections are a potential threat to humans. This includes certain pathogenic strains of *E coli*, *Salmonella*, *Campylobacter*, and *Aeromonas* and protozoa such as *Giardia*, all of which have animal reservoirs. The organism *Yersinia enterocolitica* has been previously cultured in high alpine areas of the Sierra Nevada range and may have a natural reservoir in small mammals and birds.[14] Pack animals entering the High Sierra have been subject to analysis, and *Giardia* samples were found in their manure.[15]

E coli and other pathogenic bacteria can survive in aquatic environments for long periods depending on the nutriment availability, pH, and water temperature. The number of years that *E coli* can survive in aquatic environments has been debated.[16] A study of Lake Michigan shore water showed that *E coli* may sustain itself indefinitely in appropriate environmental situations.[17]

Table 3. Cattle-grazing sites*

Wilderness area	*Place*	*Elevation (m)*	*Temperature (℃)*	*Escherichia coli CFU/100 mL*	*Other bacteria CFU/100 mL*
Carson	Upper Clark Fork River	2072	11.2	250	10 000
Carson	Lower Clark Fork River	2316	8.9	300	2600
Carson	Disaster Creek—north fork	2366	10	350	1300
Carson	Disaster Creek—east fork	2438	10.6	200	5700
Carson	Arnot Creek	2000	11.1	100	4600
Carson	Woods Gulch	1976	11.7	100	5200
Hoover	Buckeye Creek (Big Meadows)	2274	12.8	500	3800
Hoover	Buckeye Creek side creek	2377	8.9	450	4700
Hoover	Molydunite Creek	2773	11.1	400	3400
Hoover	South Fork Walker River (Burt Canyon)	2719	11.1	250	2800
Golden Trout	Mulkey Meadows	2840	15.6	100	3500
Golden Trout	Little Whitney Meadow	2560	16.7	100	3500
Emigrant	Borland Lake	2264	8.9	250	8400
Adams	East Fork Chiquito Creek	2212	14.5	100	5200
Adams	Cold Creek	2503	14	150	4600

*CFU indicates colony-forming units.

Open-range cattle are noted to carry *E coli* strain O157:H7 at a rate of 1%, placing humans who drink untreated water below established cow pastures at risk for a very serious disease.[13] Studies on this strain have also shown it to survive in cold water.[18] In addition, many non-O157 *E coli* are capable of inducing serious disease in humans.[10] Although it is possible to genetically differentiate human from animal and ecologic *E coli*, these techniques are very expensive and available only in limited laboratories in the United States.

Finally, we wish to comment on the noncoliform bacteria found in the study. Aquatic bacteria are part of a normal ecosystem of lakes and streams.[19] Indeed, if bacteria were absent, the normal food chain from frogs to fish, as well as the ecological balance, would be in jeopardy. The most common bacteria we found was *R aqua-*

Table 4. Low-impact sites: rare visits by humans*

Wilderness area	*Place*	*Elevation (m)*	*Temperature (℃)*	*Escherichia coli CFU/100 mL*	*Other bacteria CFU/100 mL*
Hall area	Green Treble Lake—lower	3115	10	None	300
Hall area	Green Treble Lake—upper	3116	10	None	400
Hall area	Maul Lake	3117	10.6	None	200
Hall area	Spuller Lake	3132	11.1	None	500
Kings Canyon	Avalanche Creek	1554	8.9	None	5000
Yosemite	Middle Dana Fork Creek	3016	12.8	None	1200
Yosemite	Parker Pass Creek	2971	13.9	None	1500
Yosemite	Granite Lake	3167	14.5	None	1200
Kings Canyon	Cunningham Creek	2621	14.0	None	2300
Sequoia	Upper Buck Creek	2209	16.7	None	3400
John Muir	Little Cottonwood Creek	2996	14.5	None	1900
Kings Canyon	North Guard Creek	2895	14.0	None	2600
Sequoia	Side Spring Creek Franklin Pass Trail	3078	5	None	1200
Sequoia	Laurel Creek	2063	13.9	None	4700
Yosemite	Miguel Creek—upper north fork	1503	12.8	100	1800

*CFU indicates colony-forming units.

tilis. Several nonpathogenic species of *Yersinia* were also cultured. Many bird species can be carriers of nonpathogenic species of *Yersinia* and *Y enterocolitica*.[20] Previous studies of wilderness water suggest a correlation between total bacterial counts and usage by backpackers.[6,7] Freshwater from remote alpine areas has been shown to be a source of *Campylobacter*, *Salmonella*, and *Y enterocolitica*, although these were not found in the current study.[21,22,23]

Conclusion

The risk for finding coliform bacteria in alpine wilderness water was determined by the use of the adjacent watershed. Water in areas used extensively by pack animals or for cattle grazing was routinely contaminated, whereas water in those areas used exclusively by backpackers or rarely visited by humans was rarely contaminated.

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Cow manure in headwater streams: tracing aquatic insect responses to organic enrichment

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Abstract. Cattle grazing in riparian areas can affect the structure of aquatic insect assemblages by adding nutrients (manure) to the stream, or by physically altering the habitat through trampling or foraging. Although cattle grazing is a well-described source of disturbance in stream habitats, the effects of manure inputs have not been previously isolated from effects of the physical disturbance. We traced the responses of aquatic insects representing 5 different functional-feeding groups to this introduced food source. We simulated manure enrichment from light grazing intensity and hypothesized that insects that feed directly on imported organic matter (shredders, filterers, gatherers) would have higher assimilation rates and densities than predators or periphyton-grazers in response to manure enrichment. We expected insect responses to increase with incremental enrichment over time, and decrease with increasing distance from the manure input. We introduced manure (composed of undigested C_4 plant particles) from corn-fed cows into 7 streams that drain forests of C_3 plants in Coastal and Sierra Nevada regions in California. Stable C isotope ratios indicated all feeding groups assimilated the isotopically distinct manure. In the Coastal streams, groups assimilating the most manure were gatherers (net increase of 21% towards C_4 plant signal) and filterers (20%), whereas shredders (9%) assimilated the least. In the faster-flowing Sierran streams, assimilation by each group was $\leq$9%. Temporal increases in manure uptake were detected in the Coastal mayfly gatherer *Paraleptophlebia pallipes*, suggesting increased manure assimilation over time. Manure uptake by insects was spatially localized within 3 m downstream of the site of manure input. Densities of all 6 genera representing 5 functional-feeding groups were not significantly altered in response to manure enrichment. However, chironomids increased >5-fold in densities after 4 wk of enrichment. The composition of chironomid genera did not shift, and insect taxa richness in the enriched treatments did not change in response to manure enrichment. In our simulation of enrichment effects, which approximated low-density grazing of 6 cows visiting each stream reach weekly for 8 wk, we found that in the absence of physical disturbances from cattle grazing, manure is an important food source for gatherers in particular, and elicited responses from chironomids that are characteristic of organic enrichment.

Key words: aquatic insects, organic enrichment, carbon isotopes, streams, cattle grazing.

Streams, particularly those in shaded headwaters, are predominately heterotrophic systems that depend on allochthonous material as an important source of food for primary consumers (Minshall 1967, Fisher and Likens 1973). Sources of imported organic matter in streams are numerous and range from effluents from paper mills (e.g., Mayack and Waterhouse 1983), sewage treatment plants (e.g., Hynes 1960, Hawkes 1979), and a variety of other anthropogenic sources (e.g., Rosenberg and Resh 1993) to natural inputs of leaf litter (e.g., Webster and Benfield 1986), wood debris (e.g., Aumen et al. 1990, Benke and Jacobi 1994), and animal carcasses (e.g., Durbin et al. 1979, Wipfli et al. 1999). The role of organic matter as a primary energy source for headwater stream communities has been extensively studied, particularly with a focus on aquatic insect responses to allochthonous inputs (e.g., Webster and Benfield 1986, Benke and Jacobi 1994).

One common source of organic pollution in streams is manure from cattle grazing. Aside from the large-scale contribution of manure inputs to streams by feedlots and dairies, manure from cattle also enters streams directly as cattle frequent riparian areas. Grazing by livestock, particularly cattle, is the most widespread land management practice in the United States (Council for Agricultural Science and Technology 1974). Cattle grazing has important implications for stream ecology because the distribution of cattle on forested range is primarily

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determined by the location of water (Roath and Krueger 1982, Kie and Boroski 1996). Cattle preferentially occupy riparian habitats in headwater streams in the arid west, which provide water, thermoregulation, shade, and nutritious forage (Roath and Krueger 1982, Pinchak et al. 1991). It has been estimated that 1 range cow can deposit an average of 2.3 kg (wet mass)/d of manure directly into a stream (Moore 1988). Because cattle graze in herds (Roath and Krueger 1982), this source of manure input to streams can be highly concentrated.

Livestock grazing effects on stream ecology result in changes in streamside vegetation, channel morphology, and water quality (Platts 1981, Krueger and Waters 1983, Fleischner 1994, Strand and Merritt 1999). However, previous studies are predominately qualitative and do not separate the effects of physical disturbance caused by cattle grazing (e.g., trampling and foraging) from the enrichment effects caused by cow manure that is directly deposited into streams.

Manure imports to streams are available to aquatic insects in many forms: as coarse and fine particles; as a substrate on which algae and biofilm grow; and through prey that have assimilated the manure. Uptake by aquatic insects of experimentally added manure can be detected using stable C isotope ratios and changes in density of different functional-feeding groups. Stable C isotope ratios are often used in lotic foodweb studies to trace an animal's food source because the isotope ratios of animals are primarily determined by diet (Deniro and Epstein 1978, Rounick et al. 1982). Furthermore, little change in the isotope ratios of consumers occurs as C moves through food chains (Rounick and Winterbourn 1986, Peterson and Fry 1987). We hypothesized that insects that feed directly on imported organic matter (shredders, filterers, gatherers) would have higher assimilation rates and densities than predators or periphyton-grazers in response to manure enrichment. We expected insect responses to increase with incremental enrichment over time, and decrease with increasing distance from the manure input.

Methods

Study sites

We compared manure-enrichment effects on aquatic insect populations in streams that were chemically and physically distinct, using field experiments in headwater streams in 2 montane regions of California (Fig. 1). We tested our hypotheses during summer baseflow. We used 3 headwater streams (each draining ~2.0 km²) in the Coast Range in Marin County (lat 37°57'N, long 122°43'W). We used 4 headwater streams (each draining 10–20 km²) in the Sierra Nevada in El Dorado County (lat 38°55'N, long 120°40'W). The streams were representative of grazed streams, and drained forests of plants using the C_3 photosynthetic pathway. The vegetation along the Coastal streams is a mixed evergreen forest dominated by coast live oak (*Quercus agrifolia* Nee), California bay (*Umbellularia californica* (Hook & Arn.) Nutt.), madrone (*Arbutus menziesii* Pursh), and coastal redwoods (*Sequoia sempervirens* (D. Don) Endl.). Sierran streams drain catchments of mixed conifer forests and are lined by riparian vegetation that includes incense-cedar (*Calocedrus decurrens* (Torrey) Florin), pacific yew (*Taxus brevifolia* Nutt.), bigleaf maple (*Acer macrophyllum* Pursh), white alder (*Alnus rhombifolia* Nutt.), willow (*Salix* spp.), and dogwood (*Cornus nuttallii* Audubon, *C. sessilis* Durand).

Treatments

We manipulated only the level of manure inputs in the streams. Each stream had 3 treatments: 1) manure-enriched, 2) placebo (i.e., a procedural control that simulated experimental manipulation without manure enrichment), and 3) control. Each of the treatments was separated by ≥30 m (Fig. 2). We introduced ~14 kg wet manure to a pool in the downstream reach in each stream once a week for 8 wk. This amount is equivalent to deposition by 6 cows in a 1-d period (Moore 1988), and simulated relatively light grazing intensity. The amount more than doubled the mass of standing organic matter (measured as coarse and fine particulate organic matter following the techniques of Wallace and Grubaugh 1996) within a 13-m reach in the Sierran streams (RBD, unpublished data).

As the placebo, we introduced an artificial, non-nutritive substrate (two 38-cm diameter clay saucers) to simulate the localized effects of manure deposition on velocity and disturbance to the local community. The control was an unmanipulated upstream reach, which we used to monitor changes in conditions during the

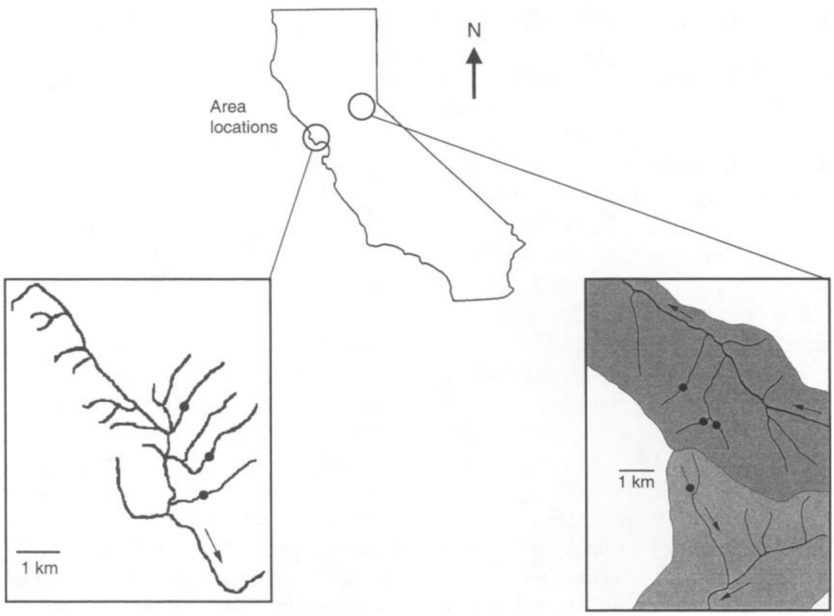


FIG. 1. Location of study sites in 3 Coastal (left panel) and 4 Sierran (right panel) California streams. Dots indicate study sites. Arrows indicate direction of flow.

course of the experiment. In this fixed-block design, placebo and control treatments were randomly assigned, whereas manure-enriched treatments were kept furthest downstream to prevent downstream contamination.

The manure had a distinct C isotope ratio ($\delta^{13}C = -15.8‰ \pm 0.1$ SE) compared to the detritus and algae available to consumers. The $\delta^{13}C$ of terrestrially derived detritus (e.g., leaf litter) in Coastal streams was $-28.2‰$ (± 0.1) and in Sierran streams was $-28.9‰$ (± 0.2). The $\delta^{13}C$ of filamentous algae in Coastal streams was $-33.9‰$ (± 0.1) and in Sierran streams was $-39.4‰$ (± 0.1). The manure was composed of 39% total C, 2% total N, 10 ppm NH_4-N, 10 ppm NO_3-N, 0.73% total P (TP), and 3500 ppm orthophosphate. Manure was obtained from 17 penned Holstein cattle that were fed corn silage (i.e., C_4 plant) for at least 7 continuous d (i.e., the time recommended by Jones et al. 1979). Cattle ingestion of corn silage involved a net fractionation in $\delta^{13}C$ of $\sim -4.2‰$ between corn silage ($-11.6‰$) and manure ($-15.8‰$). Fresh manure was collected over 3 d, immediately packaged in polyethylene bags, sealed, and kept at 1°C from 1 to 65 d. Each week, 7 bags were randomly removed from the cold storage and applied to the 7 streams. During a pilot study, cold storage of the manure for 9 mo resulted in a 0.5‰ increase in $\delta^{13}C$.

Stream characteristics

We measured physical and chemical characteristics of stream water at 4 stations within each reach (3 m upstream, 0.5 m, 3 m, and 10 m downstream of treatments and within the control) (Fig. 2). We collected samples 2, 4, and 8 wk after the start of the experiment. We measured stream discharge, velocity, depth, and width within each reach. We measured water temperature, pH, dissolved oxygen concentration, and conductivity in situ using portable meters 2 m downstream from either the manure or clay saucers, and within the control reach. We collected water samples for NO_3-N and NH_4-N determination (diffusion–conductivity analyzer, APHA 1998) and TP (ascorbic acid reduction, APHA 1998). We filtered these samples in the field, and refrigerated them at 4°C before analysis. We used these nutrient measurements to explain whether shifts in insect species composition (e.g., grazer abundance and diversity) were caused by algal response to manure or direct nutrient and organic enrichment from manure.

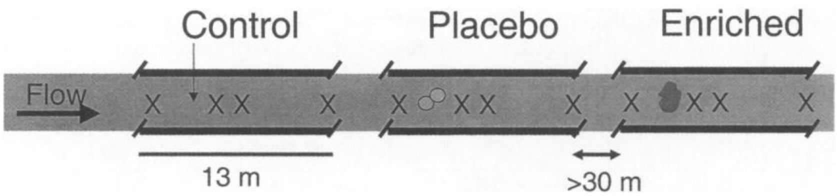


FIG. 2. Location of sampling stations in 3 replicate Coastal streams and 4 replicate Sierran streams. Three samples of insects were collected from each station (x) on 3 dates: 2, 4, and 8 wk after the start of the experiment. In this design, placebo and control treatments were randomly assigned, whereas manure-enriched treatments were kept furthest downstream to prevent downstream contamination.

Responses of stream insects

Insect densities and richness.—We used a modified Hess sampler (177 cm^2, 63-μm mesh) to collect 3 samples of aquatic insects from each station within a reach. We took samples randomly from the left, center, and right areas of the channel to include representative habitats within each station. The samples were pooled and preserved in 95% ethyl alcohol. Each sampled area was flagged to avoid sampling the same area twice. All insects from each pooled sample were sorted using a dissecting microscope. Manure-enrichment effects on insect populations were measured by comparing densities of insects among treatments at both temporal and spatial scales.

Functional-feeding groups.—Five genera of insects were used to represent 5 functional-feeding groups. Taxa from the Coastal streams included the shredder caddisfly *Lepidostoma* spp. (includes *L. unicolor* (Banks), *L. canthum* (Ross), *L. cinereum* (Banks), *L. jewetti* (Ross), *L. lotor* (Ross)); the gatherer mayfly *Paraleptophlebia pallipes* (Hagen); the grazer mayfly *Cinygma dimicki* (McDunnough); the filterer caddisfly *Hydropsyche* spp. (includes *H. californica* (Banks), *H. oslari* (Banks), *H. cockerelli* (Banks), *H. philo* (Ross)); and the predator stonefly *Calineuria californica* (Banks). Taxa from the Sierran streams included the shredder stonefly *Malenka* spp. (includes *M. depressa* (Banks), *M. californica* (Claassen); the gatherer mayfly *Paraleptophlebia* spp. (includes *P. debilis* (Walker), *P. pallipes* (Hagen)); the grazer mayfly *Cinygma dimicki* (McDunnough); the filterer caddisfly *Hydropsyche* spp. (*H. cockerelli* (Banks), *H. oslari* (Banks), *H. occidentalis* (Banks)); and the predator stonefly *Calineuria californica* (Banks).

Tracing manure uptake.—We quantified how manure assimilation was related to insect distance from the site of manure input and length of exposure to the enrichment by analyzing the ^{13}C signal. Animals feeding on organic matter rich in ^{13}C will have a high $\delta^{13}C$ value. For example, C_4 plants such as corn have higher $\delta^{13}C$ values (i.e., $\sim -12‰$) compared to C_3 plants (i.e., $\sim -27‰$) because C_3 plants discriminate more against ^{13}C during photosynthesis (Tieszen and Boutton 1988). The reliability of detecting shifts in an animal's diet is made more robust if the potential food sources each have a distinct (e.g., $\geq 5‰$ different) C isotope signal (Tieszen and Boutton 1988). Therefore, introduced organic matter from C_4 plants, such as manure from cattle that were fed corn silage, will have a C isotope signal that is distinct from terrestrially derived detritus in streams draining watersheds of C_3 plants.

We examined whether insects assimilated the manure by collecting insects using a Surber sampler (930 cm^2, 63-μm mesh) immediately downstream of each station within a reach. We randomly chose insects from each sample, allowed them to clear their guts for 3 d, dried them at 50°C for 24 h, and ground them to a powder. Each sample consisted of 4 to 10 homogenized insects. We combined insects collected from control and placebo treatments in each stream. We identified the C isotope ratios of potential food sources in the study streams by clipping and collecting 2.5 cm distal filaments of algae and collected terrestrially derived detritus (e.g., leaf litter, twigs) using a Surber sampler in each treatment reach in the Coastal and Sierran streams. We composited samples of algae and detritus by treatment area, dried them at 50°C for 24 h, and ground them to a powder. We also homogenized 5 samples of the manure and processed them as above. Isotope ratios of insects, terrestrially derived detri-

TABLE 1. Mean (±SE) chemical and physical parameters of 3 Coastal streams and 4 Sierran streams. Significant regional differences for all parameters were detected (unpaired t-tests, $p < 0.06$), except for NH_4-N and total P. Data were from placebo and control reaches for all 3 sampling times.

Parameter	Coastal ($n = 18$)	Sierran ($n = 24$)	p-value	Power
NH_4-N (ppm)	0.02 ± 0.01	0.60 ± 0.02	0.06	0.45
NO_3-N (ppm)	0.15 ± 0.04	0.05 ± 0.02	0.02	0.69
Total P (ppm)	0.01 ± 0.01	0.03 ± 0.03	0.47	0.12
Conductivity (μS/cm)	237 ± 5	22 ± 4	<0.001	1.0
pH	7.7 ± 0.1	6.6 ± 0.1	<0.001	1.0
Temperature (°C)	13.7 ± 0.3	9.7 ± 0.2	<0.001	1.0
Dissolved oxygen (mg/L)	8.2 ± 0.9	7.1 ± 0.8	<0.001	0.99
Velocity (cm/s)	11.1 ± 2.1	22.7 ± 2.2	<0.001	0.96
Depth (cm)	6.6 ± 0.5	11.6 ± 0.6	<0.001	1.0
Width (m)	0.75 ± 0.06	0.99 ± 0.10	0.04	0.14
Discharge (m^3/s)	0.64 ± 0.18	2.80 ± 0.64	0.01	0.99

tus, filamentous algae, and manure were determined using a Europa 20–20 continuous-flow isotope ratio mass spectrometer. Ratios of C isotopes of the samples were relative to the international standard, Vienna PDB, Belemnite fossil of the Pee Dee formation in South Carolina.

Data analysis

Stream characteristics.—We compared physical and chemical parameters between the Coastal and Sierran streams using unpaired t-tests. We examined manure enrichment effects on each of the physical and chemical parameters using a 2-factor ANOVA (treatment, week) for each region.

Tracing manure uptake.—We used a 2-factor ANOVA (treatment, week) to determine if individual taxa in each region assimilated the manure during 8 wk of enrichment, and (taxon, treatment) to determine if taxa in each region differentially assimilated the manure. For taxa that assimilated manure, we examined temporal patterns of assimilation using linear regression. We used a 2-factor ANOVA (treatment, proximity to manure input) using data collected on the 8^{th} wk of the study to examine spatial patterns of manure assimilation.

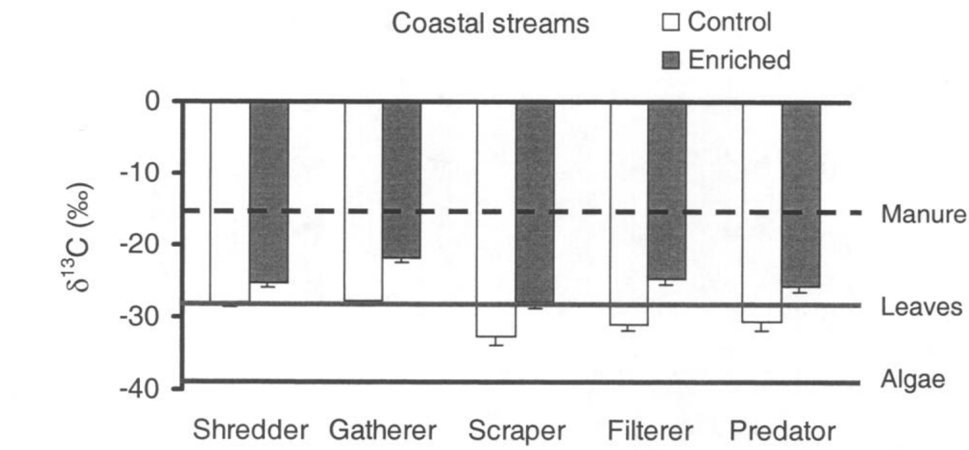
Insect densities and richness.—We used a 2-factor ANOVA (treatment, week) to detect enrichment effects on taxa densities, total insect density, and taxa richness in Coastal streams. For Sierran streams (for which only samples from the 8^{th} week were processed), we used ANOVA (treatment) to detect enrichment effects on taxa densities, total insect density, and taxa richness using data collected on the 8^{th} week of the study. We analyzed temporal and spatial effects of manure enrichment on Coastal stream densities using a factorial ANOVA (treatment, week, proximity). Taxa densities were transformed using log(x + 1). We used a Tukey–Kramer test for all multiple comparisons following ANOVA. NCSS statistical software (1998. NCSS 2000 Number Cruncher Statistical Systems, NCSS, Kaysville, Utah) was used for all parametric analyses. Power was calculated at $\alpha = 0.05$.

Results

Regional characteristics

Streams were significantly different by region for all physical and chemical parameters except NH_4-N and TP (Table 1). For example, the Coastal streams had 1 unit higher pH, 4°C higher temperature, 15% higher dissolved oxygen concentrations, 3-fold higher NO_3-N, and 10-fold higher conductivity. The Sierran streams had double the velocity, greater depth and width, and 4-fold higher discharge.

None of the physical and chemical stream parameters within either region significantly changed as a result of manure enrichment (ANOVA, includes NH_4-N: $F = 3.31$, $p = 0.06$, power $= 0.43$; NO_3-N: $F = 0.65$, $p = 0.53$, power $= 0.13$; TP: $F = 1.03$, $p = 0.38$, power $= 0.16$; conductivity: $F = 0.16$, $p = 0.86$, power $= 0.07$; pH: $F = 0.43$; $p = 0.65$, power $= 0.10$; temperature: F



	Shredder	Gatherer	Scraper	Filterer	Predator
$\delta^{13}C$ change (%)	9.4	21.4	14.4	20.4	16.4
F	5.3	20.6	5.4	37.0	11.2
df	1, 38	1, 27	1, 18	1, 12	1, 10
p	0.03	0.006	0.001	0.001	0.03
Power	0.56	0.98	0.47	0.98	0.50

FIG. 3. Mean (−SE) $\delta^{13}C$ values of taxa of different functional-feeding groups from Coastal streams showing how each taxon assimilated the introduced manure over the 8-wk period. Mean $\delta^{13}C$ values are shown (lines) for manure (dashed), leaves (solid gray), and algae (solid dark). Mean $\delta^{13}C$ values are based on insects collected downstream from manure input (and comparable locations from control reaches) on all 3 sampling dates. Each sample consisted of 4 to 10 insects. Statistical data are from a 2-factor ANOVA (treatment, week).

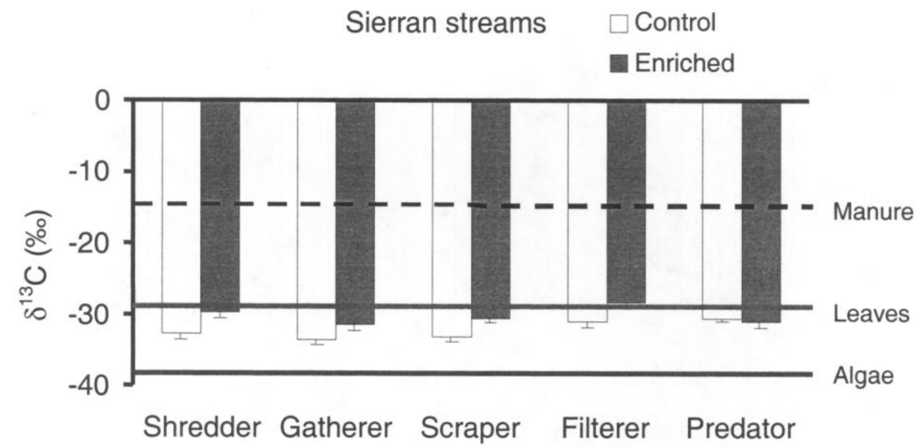
= 3.35, p = 0.06, power = 0.46). In particular, NH_4-N, NO_3-N, and dissolved oxygen concentrations in the treatment reaches did not change at any time during the experiment.

Tracing manure uptake

Aquatic insects in both Coastal and Sierran regions assimilated the manure, with the various functional-feeding groups assimilating it at different rates. Coastal taxa representing the 5 functional-feeding groups in the enriched treatments had a net increase of 9 to 21% in insect $\delta^{13}C$ values (ANOVA, Fig. 3). In the Coastal streams, the gatherer *Paraleptophlebia* assimilated more manure than the scraper (*Cinygma*), filterer (*Hydropsyche*), or shredder (*Lepidostoma*) (2-factor ANOVA [taxon, week], F = 8.58, df = 4,105, p < 0.001). In the faster-flowing Sierran streams, 4 of the 5 taxa assimilated the manure but uptake was only 6 to 9% in insect $\delta^{13}C$ values (ANOVA, Fig. 4). Sierran taxa did not have significantly different rates of assimilation (F = 2.06, df = 4,126, p = 0.09, power = 0.58).

Manure uptake did not increase significantly over time for most Coastal taxa in the enriched treatments except for the gatherer *Paraleptophlebia* (linear regression, p < 0.01). Its rate of assimilation averaged 1.2‰/wk (linear regression, Fig. 5). None of the taxa in the Sierran streams increased their assimilation of manure over time (linear regression, p > 0.69).

A spatial pattern of localized manure uptake was detected using isotope ratios when all taxa were combined. For example, after 8 wk of enrichment, insects 0.5 m and 3 m downstream of the site of manure input in Coastal streams had significantly higher $\delta^{13}C$ values than those in comparable locations in control treatments (ANOVA, Fig. 6). The same insects collected upstream and 10 m downstream, however, did not significantly assimilate the manure. This spatial pattern was significant when considering all taxa combined for each region (2-factor ANOVA [treatment, proximity to manure], F = 5.02, df = 3,53, p = 0.004 for Coastal; F = 3.87, df = 3,60, p = 0.01 for Sierran), but it was not statistically significant when individual taxa were examined separately (e.g., Coastal *Paraleptophlebia*, ANOVA, F = 1.71, df = 2,25, p = 0.21, power



	Shredder	Gatherer	Scraper	Filterer	Predator
$\delta^{13}C$ change (%)	8.8	6.1	7.8	9.0	-1.1
F	5.8	4.3	6.2	11.7	na
df	37	39	30	14	6
p	0.02	0.05	0.02	0.005	na
Power	0.60	0.52	0.60	0.88	na

FIG. 4. Mean (−SE) $\delta^{13}C$ values of taxa of different functional-feeding groups from Sierran streams showing how each taxon assimilated the introduced manure over the 8-wk period. Mean $\delta^{13}C$ values are shown (lines) for manure (dashed), leaves (solid gray), and algae (solid dark). Mean $\delta^{13}C$ values are based on insects collected downstream from manure input (and comparable locations from control reaches) on all 3 sampling dates. Each sample consisted of 4 to 10 insects. na = not applicable because of insufficient number of replicates. Statistical data are from a 2-factor ANOVA (treatment, week).

= 0.28; Coastal *Lepidostoma*, $F = 0.40$, df = 2,28, $p = 0.68$, power = 0.10).

Population and community responses

Densities of the examined taxa representing different functional-feeding groups did not change in response to manure enrichment at any time during the experiment in either Coastal or Sierran streams. Assemblage responses to manure enrichment in terms of densities and richness were region specific. Insect assemblages in the Sierran streams were unresponsive to enrichment. For example, Sierran total insect

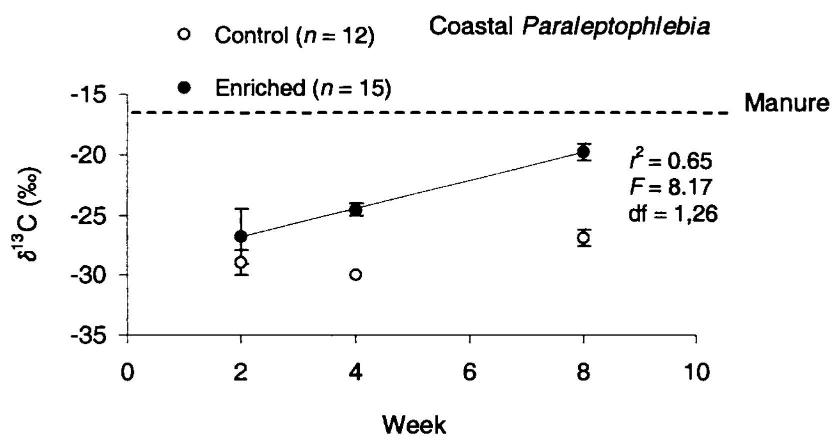


FIG. 5. Temporal trend of increasing manure uptake by Coastal *Paraleptophlebia* during the 8-wk period of enrichment. Mean (±SE) $\delta^{13}C$ of manure is indicated by dashed line.

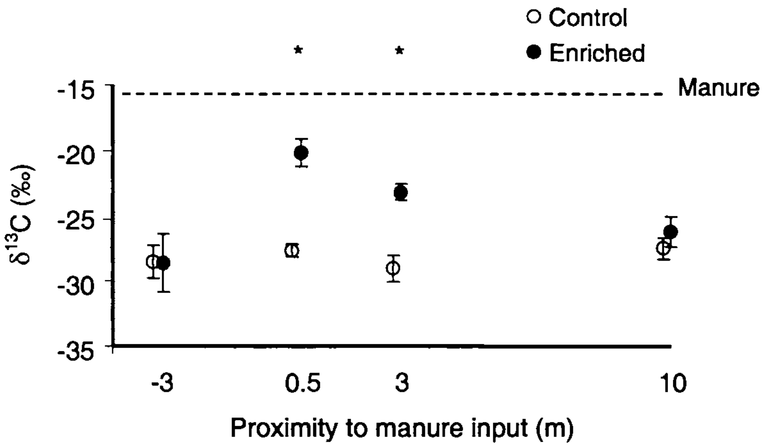


FIG. 6. Mean (±SE) C isotope ratios of taxa from Coastal streams showing localized manure uptake after 8 wk of enrichment. Mean $\delta^{13}C$ of manure is indicated by dashed line. * = significant difference (ANOVA) at $p < 0.004$.

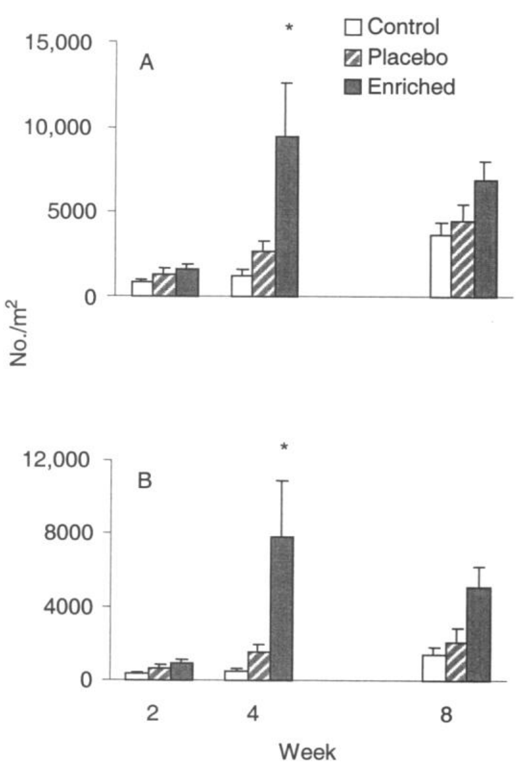


FIG. 7. Mean (+SE) densities of total insects (A) and chironomid midges (B) in 3 Coastal streams, temporally for each treatment. * = significant difference (ANOVA) at $p < 0.001$.

densities (ANOVA, $F = 0.38$, df = 2,48, $p = 0.69$, power = 0.11), chironomid densities (ANOVA, $F = 1.35$, df = 2,48, $p = 0.27$, power = 0.28), and taxa richness (ANOVA, $F = 1.09$, df = 2,48, $p = 0.35$, power = 0.23) did not change after 8 wk of enrichment. In contrast, Coastal total insect densities and chironomid densities were significantly higher in the enriched treatments. In particular, after 4 wk of enrichment, total insect densities were >3-fold greater in enriched compared to either the placebo or control treatments (Fig. 7A; ANOVA, $F = 17.17$, df = 2,108, $p < 0.001$).

Enrichment effects on Coastal densities after 4 wk were spatially localized. Total insect densities within 3 m downstream from the site of manure input were significantly higher compared to those in placebo or control treatments (Fig. 8A; ANOVA, $F = 11.15$, df = 2,108, $p < 0.001$). For example, in the enriched treatments, insect densities 0.5 m downstream ($\bar{x} = 1236/m^2$) were >9 times higher, and densities 3 m downstream ($\bar{x} = 293/m^2$) were >1.5 times higher than in respective locations in the placebo and control reaches. Taxa richness in the Coastal streams, however, did not change as a result of enrichment (ANOVA, $F = 0.23$, df = 2,108, $p = 0.80$, power = 0.08).

The numerically dominant group, the Chironomidae, influenced both temporal and spatial patterns of increased densities in the Coastal

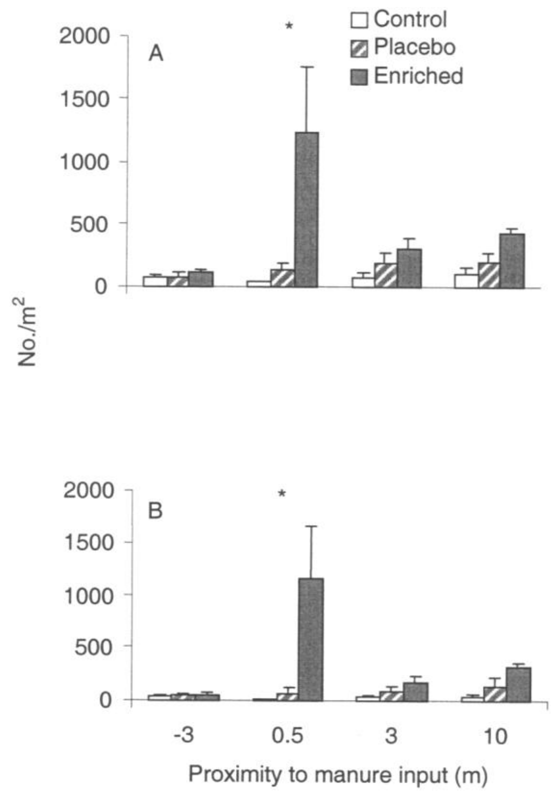


FIG. 8. Mean (+SE) densities of total insects (A) and chironomid midges (B) in 3 Coastal streams, spatially after 4 wk of treatments. * = significant difference (ANOVA) at $p < 0.001$.

streams. For example, chironomids increased >5-fold after 4 wk of enrichment (Fig. 7B; ANOVA, $F = 25.70$, df = 2,107, $p < 0.001$). In the enriched treatments, chironomid densities 0.5 m downstream ($\bar{x} = 1150/m^2$) were >17 times higher, and densities 3 m downstream ($\bar{x} = 181/m^2$) were >2 times higher than those in respective locations in the placebo and control reaches (Fig. 8B; ANOVA, $F = 19.65$, df = 2,107, $p < 0.001$). However, despite their numerical dominance, the composition of chironomid taxa was not altered by manure enrichment. For example, 7 of the most common genera of chironomids in Coastal streams were consistently present in both the control and enriched sites. These common midges increased in densities by 11-fold (e.g., *Microtendipes, Micropsectra, Parametriocnemus*), >30-fold (e.g., *Rheocricotopus, Thienemannimyia* group, *Brillia*), and up to 80-fold (e.g., *Polypedilum*) in the enriched treatments.

Discussion

Regional differences in assimilation

Insects having different food-gathering mechanisms assimilated the introduced manure. This pattern was seen in both Coastal and Sierran streams, despite the inherent regional physical and chemical differences (i.e., discharge, water chemistry, and nutrient concentrations) of the 2 groups of streams. Assimilation of the manure was more pronounced in the Coastal streams where lower discharge allowed for relatively more manure particles to settle and remain on the streambed and thereby become available to consumers. In contrast, the higher discharge of the Sierran streams may have diluted nutrient concentrations so that dosing was different or may have shortened the residence time of the manure. Field data showed the relationship between discharge and amount of manure on the streambed. For example, after 1 d of enrichment, organic matter collected 0.5 m downstream from the site of manure input consisted of 52% manure in streams with low discharge (e.g., 0.006 m^3/s), whereas manure composed 18% of the organic matter in streams with 6-fold higher discharge (e.g., 0.04m^3/s).

Assimilation pathways

Insects may have assimilated the manure by at least 3 pathways. First, insects may have incidentally ingested the manure in the same manner that fine inorganic particles (e.g., sediment) are commonly found in gut contents (e.g., Coffman et al. 1971). Second, consumers of periphyton or biofilm may have assimilated the manure through feeding on heterotrophic microorganisms that use detrital exudates (i.e., in this case manure particles) as their primary source of C (Winterbourn and Rounick 1985). Third, insects may be feeding on the undigested corn particles in the manure itself. However, if insects feed directly on these corn particles, the differences in isotope ratios in the manure components (e.g., corn kernels versus corn leaves) may lead to different isotope ratios in consumers because seeds generally have higher $\delta^{13}C$ values than leaves (O'Leary 1981). Taxa representing the different functional-feeding groups may have used ≥1 of these pathways; nevertheless, our study indicated that insects, regardless of the food-gathering mechanism used, assimilated the manure.

Functional-feeding groups

Our hypothesis that gathering, shredding, and filtering insects would have higher assimilation rates of the introduced manure compared to predators and grazers was supported by only 1 taxon, the Coastal gathering mayfly *Paraleptophlebia*. The general absence of a temporal trend of increasing manure assimilation by insects suggests that insects relied on a set proportion of the manure for their diet. For Coastal *Paraleptophlebia*, its isotope ratio increased at a rate of 1.2‰/wk, suggesting that manure represented an increasing % of its diet. This field result was supported by a laboratory study in which *Paraleptophlebia* was fed manure as 100% of its diet for 14 d (RBD, unpublished data), and showed an increased rate of manure assimilation (a change in $\delta^{13}C$ of 2.2‰/wk, from −32.97‰ to −28.52‰ after 14 d). These results illustrate the role of manure as potential food resources for aquatic insects, and suggest that manure represents an increasing % of the diets of gatherers, whereas manure only represents a fixed proportion of the diet for shredders, filterers, grazers, and their predators.

However, the length of the study may not have allowed for appropriate monitoring of assimilation rates in the slower-developing stoneflies and caddisflies. Caddisflies such as *Lepidostoma* typically require 6 to 8 wk to develop between instars (Grafius and Anderson 1979, 1980). In contrast, mayflies have from 15 to 25 instars (Butler 1984). The relatively fast instar development of the mayfly *Paraleptophlebia* (which feeds directly on imported organic matter in contrast to the periphyton-grazer *Cinygma*) may explain why it was most responsive to manure enrichment. Fry and Arnold (1982), for example, observed that turnover of C is closely tied to growth and not time, which may explain why increased assimilation was not detected in this study for the slower-growing stonefly and caddisfly specimens.

The spatial pattern of localized manure uptake demonstrated that insects within 3 m downstream of the manure input assimilated a significant amount of the manure. Manure availability in streams was also spatially distinct; the amount of available manure decreased further downstream. For example, manure composed 28% of the organic matter 0.5 m downstream in the fast-flowing streams, but the amount of manure available to insects decreased to 9% at 3 m downstream, and to 2% at 10 m downstream.

Responses to organic enrichment

Our hypothesis that gatherers, shredders, and filterers would have higher densities compared to predators and grazers in response to manure enrichment was not supported by this study. Aquatic insect assemblages typically respond to organic enrichment (e.g., from effluents) through changes in species composition, increased densities of taxa tolerant to enrichment, and decreased densities or elimination of taxa sensitive to enrichment (Hynes 1960). Although we demonstrated that insects assimilated the manure, no significant changes in densities or composition were detected for most of the taxa examined, which may be attributable to relatively insufficient enrichment load or length of monitoring of insect responses.

However, the Chironomidae, a characteristically organic-enrichment tolerant group (Johnson et al. 1993) had the classic response to manure inputs—multifold increases in density—after only 4 wk of enrichment. Despite the pronounced changes in their densities, manure enrichment did not alter chironomid genera composition or result in numerical dominance by any one taxon. Rather, the 7 common genera of chironomid midges in the Coastal streams were consistently abundant in both control and enriched treatments. Each of the common midge genera are moderately tolerant to organic pollution, sharing similar tolerance values (USEPA 1999), which likely explains the lack of numerical dominance by any genera.

In conclusion, we separated the effects of manure enrichment on aquatic insect assemblages from the physical disturbances to the stream caused by cattle grazing. Our simulation of manure enrichment approximating light grazing intensity (6 cows visiting streams once a wk for 8 wk) revealed that insects, regardless of their feeding mode, assimilated the manure. The role of manure as a food source and patterns of manure uptake were detected in streams regionally distinct in their hydrology and chemistry. Cattle grazing effects on stream ecosystems have been well documented (see reviews by Kauffman and Krueger 1984, Fleischner 1994, Strand and Merritt 1999), but our findings suggest that, in the absence of physical disturbances by cattle grazing, manure by itself elicits responses from aquatic insects that are characteristic of organic enrichment.

Acknowledgements

We thank Steve Fend, Marilyn Myers, and the late William Peters for assistance in identifying chironomids, caddisflies, and mayflies, respectively. We also thank Peter Robinson, Doug Gisi, and Dan Kominek of the University of California at Davis, and Dan Giacomini, for coordinating and facilitating manure collection for this study. Paul Brooks assisted in isotope analyses. We are grateful to Jacques Finlay, William Leach, and Marilyn Myers for many discussions that led to this project and for their helpful comments on this manuscript. We also thank Mark Munn, David Rosenberg, and 2 anonymous reviewers for their comments. Blodgett Forest Research Station and Point Reyes National Seashore allowed us to conduct this research at their respective field sites.

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Chapter 6

Timber Harvesting, Silviculture, and Watershed Processes

T. W. Chamberlin, R. D. Harr, and F. H. Everest

Waters in forested lands of western North America are major producers of anadromous salmon and trout. The size of the fishery resource is large, but it is diminishing as a result of human activities and currently is only a fraction of its original size. Western forested watersheds also produce an array of other natural resources, including a variety of wood products. Areas that produce both timber and salmonids coincide over much of western North America (Figure 6.1), and the increasing public demand for both of these resources creates frequent management conflicts. Under most circumstances, both timber and fish can be successfully managed in the same watershed if measures to protect water quality and fish habitat are carefully coordinated with timber management operations.

This chapter is confined to the effects of timber management activities on stream ecosystems, particularly streams with anadromous salmonids. Lakes and estuaries are also vital to the life cycle of many anadromous salmonids, but consideration of those realms is left to other authors (e.g., Tschaplinski 1988).

Timber management activities discussed in this chapter include felling and yarding of trees, site preparation by burning or scarification, fire hazard reduction, forest regeneration by planting or seeding, reduction of competition by brush removal and tree thinning before commercial harvest, and some effects of road building on the hydrologic and sediment systems discussed in the chapter. Other chapters of this book treat road building (Furniss et al. 1991) and forest chemicals (Norris et al. 1991) in detail.

Numerous models allow the standing crop of fish to be estimated from habitat variables. Fausch et al. (1988) reviewed 99 models of various types, and Hicks et al. (1991, this volume) summarize the response of salmonids to changes in habitat. The diverse habitat requirements of many salmonids are discussed in this volume by Bjornn and Reiser (1991). We will concentrate, therefore, on how timber management influences hydrologic and sediment transport processes and thereby affects the amount and quality of flowing water, gravel substrates, cover, and food supplies required by all salmonid species.

Relations between Timber Management and Salmonid Habitat

Because the several species and life phases of salmonids have diverse habitat requirements, streams that support them productively must sustain a varied complex of hydraulic and geomorphic conditions (Sullivan 1986) that are distrib-

Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats
American Fisheries Society Special Publication 19:181–205, 1991

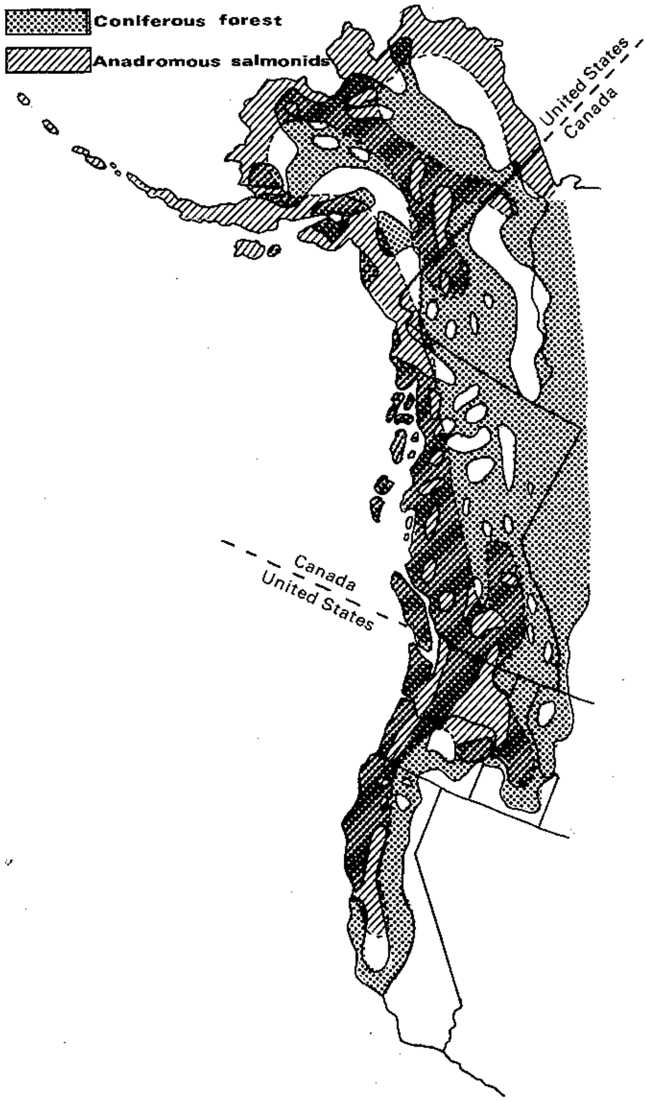


FIGURE 6.1.—Areas of timber and anadromous salmonid production in western North America.

uted along the stream continuum (Vannote et al. 1980). The close relation between watershed (basin) properties and stream characteristics has been repeatedly emphasized (Lotspeich 1980), and serves as a good approach for understanding how forest management influences fish habitat.

Figure 6.2 is a conceptual model of the linkages between timber management activities and fish. In this model, the influences of these activities are transmitted through changes in watershed processes and structures that, in turn, modify the habitat elements described by Bjornn and Reiser (1991). Hartman (1988) provided an excellent synthesis of these complex relationships for the watershed of

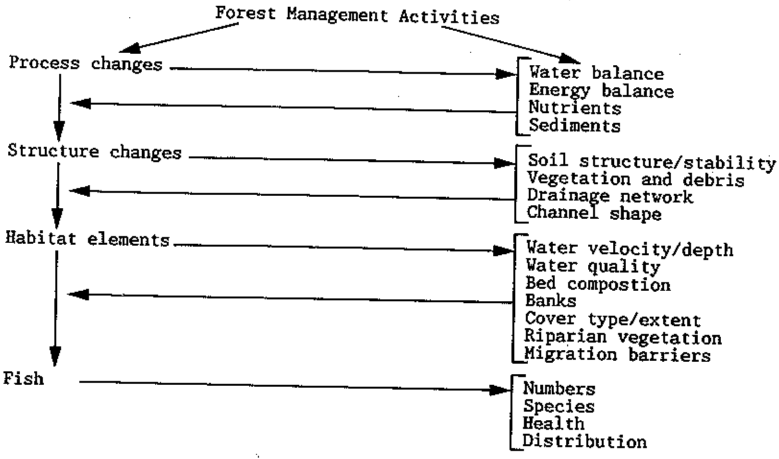


FIGURE 6.2.—Linkages between timber management activities and fish production.

Carnation Creek, British Columbia, and reemphasized the importance of understanding basic watershed processes.

Water plays a central role in watershed processes, but equally important are the sediments it moves and the structure imposed on stream channels by bedrock and the trees, roots, and logs of the riparian ecosystem. The land–water ecosystem must be managed through space and time as an integrated whole if productive fish habitat is to be maintained.

Importance of Small Streams

Salmonids occupy a wide variety of streams that range in size from tiny headwater tributaries to the mainstream Columbia River. Some species even migrate to, and spawn and rear for a while in, first-order streams that may become intermittent or dry in summer.

Most spawning and rearing in forested watersheds, however, takes place in second- to fourth-order streams; coho salmon and trout often are found further upstream than other salmonid species. Such small streams account for the majority of total aggregate stream length available to salmonids in most watersheds.

Even when small streams are not accessible to migrating fish because of barriers or steep gradients, they are vitally important to the quality of downstream habitats. The channels of these streams carry water, sediment, nutrients, and wood debris from upper portions of the watershed. The quality of downstream habitats is determined, in part, by how fast and at what time these organic and inorganic materials are transported.

Small streams are responsible for a high proportion of salmonid production in a basin, and they influence the quality of habitat in larger tributaries downstream. They also are the streams most easily altered by forest management activities. Small streams are intimately associated with their riparian zones and are highly responsive to alterations in riparian vegetation and the surrounding watershed.

Vegetative crown cover is often complete over first- through third-order streams. Because small streams depend largely on litter fall for organic energy input (Murphy and Meehan 1991, this volume), any manipulation of the canopy or streambank vegetation will influence the stream's energy supply. Likewise, road building or other activities that increase sediment supplies or modify local runoff may have greater effects on smaller streams than they would on larger systems.

Although larger streams generally have a greater capacity to buffer the effects of changes to the riparian zone, salmonid fry often preferentially inhabit the lower-velocity margins and back channels of such streams (Ptolemy 1986). Forest harvesting and other land-use impacts can accumulate over time to cause substantial changes in stream-edge environments, even along very large rivers (Sedell and Froggatt 1984).

Hydrologic Effects

Water defines fish habitat more than any other factor does. Hence, changes in the quantity, quality, or timing of streamflow caused by timber harvesting and silviculture are a primary focus for timber–fish interactions. The basic components of the hydrologic cycle—precipitation, infiltration, evaporation, transpiration, storage, and runoff—have been introduced by Swanston (1991) in this volume. Here we discuss how timber management activities influence those components and some of the consequent effects on salmonid habitats.

Regional Variations in Streamflow Response

Regional differences in runoff patterns, ranging from rain-dominated to snowmelt-dominated systems, are illustrated by Swanston (1991). Coastal watersheds with high-elevation ranges may have a mixture of runoff types with gradual transition zones; as one moves from south to north along the Pacific coast, the summer "dry" period becomes increasingly wet.

This regional variability makes it very difficult to generalize about the hydrologic effects of forest management, but can help resource managers to focus on those parts of the hydrologic cycle that will have the most influence on fish habitats in their areas. For example, in rain-dominated coastal systems, frequent high winter floods make the maintenance of side channels a primary habitat protection activity. In interior snow-dominated watersheds, by contrast, management practices to augment low late-summer rearing flows are encouraged.

Regional variation in streamflow behavior will also influence management practices related to sedimentation. In the interior basins, spring breakup and snowmelt are responsible for most of the movement of road and channel sediments, but, along the coast, frequent rains provide sufficient energy to transport material during many months of the year.

In general, forest management activities influence salmonid habitats when they alter the normal regional streamflow pattern at the extremes—that is, by increasing or decreasing the normal levels or occurrences of very high or very low flows. Management actions to manipulate these changes in beneficial directions may be possible, and are furthered by an understanding of how timber harvesting affects each component of the water balance.

Influences on the Water Balance

Timber management activities do not normally change the total amount of precipitation entering a watershed. A possible exception occurs in areas where fog drip from forest foliage adds substantially to water input but is lost when forest vegetation is removed (Harr 1982). Harvesting may, however, substantially alter the spatial distribution of water and snow on the ground, the amount intercepted or evaporated by foliage, the rate of snowmelt or evaporation from snow, the amount of water that can be stored in the soil or transpired from the soil by vegetation, and the physical structure of the soil that governs the rate and pathways by which water moves to stream channels. Within this complex of elements in the water balance, the effects of harvest and silviculture can be grouped into three major categories that form the basis for most runoff analyses: influences on snow accumulation and melt rates; influences on evapotranspiration and soil water; and influences on soil structure that affect infiltration and water transmission rates.

Snow accumulation and melt.—The forest canopy intercepts snowfall, redistributes snow, shades the snowpack, and lowers wind velocities. Harvesting affects these processes in various ways, depending on the temperature, precipitation, and wind patterns characteristic of a region.

In the colder, drier winter climate of the interior, intercepted snow is easily blown from the canopy. During prolonged windless periods, snow may sublimate and be lost from the snowpack.

In warmer, more moist climates of the transient-snow zones, snow is wetter and sticks longer in the forest canopy. Warm air (above 0°C) melts intercepted snow, causing it to reach the ground as meltwater or in wet clumps. Snowpacks under mature forest are thus variable in depth, discontinuous, and wetter than snowpacks in the open (Berris and Harr 1987). Under younger tree canopies, however, snow may be deep because tree branches are more flexible and bend downward, causing snow to slide off onto donut-shaped piles around individual boles (Berris 1984).

Forest openings alter wind patterns and trap snow. Small openings (up to eight tree heights in diameter) trap snow more effectively than large ones, although more snow will be available for melt even in large openings than in forested terrain. In the West Kootenays of British Columbia, snow accumulation in openings up to 42 tree heights in diameter was 37% greater than in the forest and melted 38% faster (Toews and Gluns 1986). Troendle and King (1985) found that peak snow water equivalent (depth of water that results when snow melts completely) averaged 9% higher, and peak snowmelt flows averaged 20% greater, after a forest was logged in small patches.

In dry interior climates, the rate at which snow melts from openings depends principally on energy from shortwave solar radiation (i.e., sunshine). Hence, the loss or creation of shade patterns can significantly affect the rate of melt and the timing of runoff peaks. During cloudy, rainy, and windy weather characteristic of winter storms on the Pacific coast, in contrast, sunshine is a minor heat source for melt compared to the convective transfer of sensible and latent heat from moist air to the snowpack. When rain falls on snow, the melt rate increases in proportion to the wind speed and the air temperature (U.S. Army Corps of Engineers 1956).

Forest harvesting that opens up stands to stronger winds can thus increase the melt rate. For example, Harr (1986) and Berris and Harr (1987) found that more heat was available to melt snow in a recent clear-cut than in an adjacent old-growth Douglas-fir stand during a rain storm of an intensity that recurs at roughly 2-year intervals. The greater amount of heat, coupled with 2–3 times more water in the snowpack, resulted in 22% more water (rain plus snowmelt) flowing from the clear-cut than from the forested plot. Likewise Golding (1987) documented a 13.5% increase in peak winter storm flows after only 19.2% of a coastal British Columbia watershed was clear-cut.

Whether increased water outflow from a logged site causes an increase or a decrease from the whole basin depends on where the site is with respect to other elevations, aspects, and distances from the channel mouth. Shallow snow in the transient-snow zones melts and runs off fairly quickly during rain storms, for example, but deeper snowpacks such as those in the Sierra Nevada may not translate melt into runoff changes as directly (Kattelmann 1987). In the snow zone, models such as those of Leaf and Brink (1973) and Kattelmann (1982) can help forest managers design logging plans that synchronize or desynchronize the runoff of snowmelt at different locations in a basin, and thereby contribute to fishery management objectives.

In the transient-snow zones of the Pacific coast, the effects of harvest on runoff are variable and not well documented, primarily because the montane relief and meteorology are themselves so variable. Still, plot studies on both small and large paired watersheds have shown increases in the size of peak flows after logging (Harr and McCorison 1979; Christner and Harr 1982; Harr 1983), suggesting that timber harvests can affect fish habitat in these areas as well.

Influences on evapotranspiration and soil water.—Clear-cutting, shelterwood cutting, or thinning eliminates or reduces a substantial area of leaves and stems that would otherwise intercept precipitation and allow it to be evaporated when sufficient energy was available. Likewise, fewer tree roots reduce the amount of water that would otherwise be extracted from the soil and hence be unavailable for streamflow. These two factors cause soil water content (and sometimes groundwater) and runoff to be higher in logged than in unlogged areas, and the effect increases as the percentage of stems removed increases. When stands are only thinned, the residual stand may increase its use of water (Hibbert 1967), so changes in streamflow following thinning are likely to be less than might be expected from counts of trees alone.

Table 6.1 shows some examples of changes in annual runoff that have been observed after timber harvesting. The increases have been largest (in absolute terms) during the growing season, when substantial precipitation also occurs. In western Oregon, the greatest portion of the annual increase occurred during the early part of the fall–winter rainy season, when rain rapidly filled the soil pores in cleared areas and then had to run off as surface flows (Rothacher 1971; Harr et al. 1979). Later in the rainy season, even soils under mature canopies became saturated, so runoff from logged and unlogged areas was roughly the same.

These generalizations apply to clear-cut areas that have not been further disturbed by roads, yarding, or burning, and that are not subject to major rain-on-snow events. Compounding effects of such physical disturbances are discussed in the next section.

TABLE 6.1.—Examples of changes in annual runoff after timber harvest. (From Hibbert 1967.)

Location	Species	Treatment	Increase in water yield in the first year (%)
Coweeta, North Carolina	Hardwoods	100% clear-cut	40
Coweeta, North Carolina	Hardwoods	35% selective	40
H. J. Andrews, Oregon	Conifers	40% clear-cut	[a]
Wagon Wheel Gap, Colorado	Mixed	100% clear-cut	22
Fool Creek, Colorado	Conifers	40% clear-cut	30

[a] Small increase in low flow.

Increased late-summer or fall runoffs can increase available fish habitat. They may also moderate the increases in stream temperature that result from removal of shade vegetation. Summer flows have doubled and tripled immediately after clear-cutting and broadcast burning of logging slash in small watersheds (Rothacher 1971; Harr et al. 1979), although increases were short-lived. Rapid regrowth of riparian vegetation may reduce summer streamflows below prelogging levels (Harr 1983).

Increases in soil water content and groundwater levels can indirectly affect fish habitat in other ways. On logged hill slopes, moist soil is vulnerable to mass movements (O'Loughlin 1972; Swanston 1974). On the other hand, higher groundwater levels after harvesting may expand the area of floodplain habitats accessible to fish, particularly during summer low-flow periods (Hetherington 1988).

Influences on soil structure.—The third group of influences that timber harvesting and silviculture can have on the water balance involves the entry of water into soil and its downslope movement to streams. Most undisturbed forest soils can accept water much faster than normal rates of rainfall or snowmelt (Dyrness 1969; Harr 1977), and virtually all water that reaches such soils enters them and follows subsurface routes to stream channels. Substantial surface runoff occurs only when storms are unusually heavy or rainy seasons are especially long, as noted above for western Oregon.

Forest management activities that disturb the soil such as road building, yarding, burning, or scarification can alter the pathways water takes to stream channels, and hence increase (or decrease) the volume of peak streamflows. Soil can be compacted by logging equipment (Greacen and Sands 1980) or by logs dragged over the ground during yarding and site preparation (Dyrness and Youngberg 1957). When surface soils are exposed, their pores can be clogged by fine sediment and their structure can be broken down by the energy of falling raindrops (Lull 1959). If the infiltration capacity of the soil is sufficiently reduced, water runs off over rather than through the soil. Higher peak flows and increased sediment transport result.

In general, yarding exposes the least amount of soil when it is done with balloons or helicopters, and the most when logs are skidded with tractors (Figure 6.3). In steep terrain, high-lead cable yarding has disturbed soils over

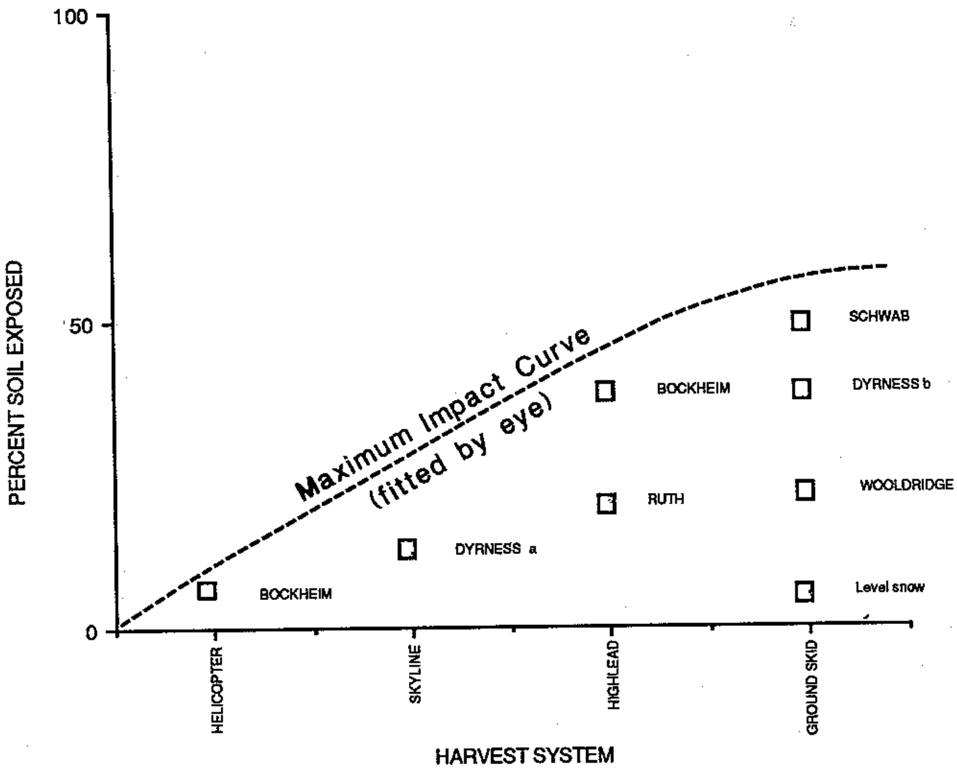


FIGURE 6.3.—Areal extent of mineral soil exposed by alternative yarding techniques. Squares represent empirical studies by the authors indicated and the well-known protection afforded soils by snow cover. Logs moved by helicopter and skyline cables do not touch the ground during transportation. In high-lead yarding, one end of a log is suspended from an overhead cable and the other end drags along the ground. Ground-skidded logs are dragged full length over the terrain. References: Bockheim = Bockheim et al. (1975); Dyrness a = Dyrness (1967b); Dyrness b = Dyrness (1965); Ruth = Ruth (1967); Schwab = Schwab (1976); Wooldridge = Wooldridge (1960).

30–60% of the logged areas (Smith and Wass 1980); on flat terrain or over snow cover, however, even tractor skidding may cause negligible disturbances (Bockheim et al. 1975; Klock 1975). These findings indicate that soil disturbance will be minimized when the harvesting system is well matched to particular site characteristics.

Internal changes in soil structure also take place after logging, as tree roots die, sediment fills soil pores, and compaction occurs. The role of large subsurface pathways in the rapid transmission of water has been shown by Cheng et al. (1975), de Vries and Chow (1978), and Hetherington (1988). The collapse or blockage of these "macropores" forces water to flow over the surface, which may accelerate erosion.

Roads and landings have relatively impermeable surfaces, and water runs off them rapidly. Ditches along roads not only collect surface runoff, they can intercept subsurface flow and bring it onto the surface (Megahan 1972). The effects of roads alone on basin hydrology have not received much study, but there is some evidence that roads can accelerate storm runoff and cause higher peak flows in small basins (Harr et al. 1975, 1979; Harr 1979).

Soil properties on upper slopes are remote from the concerns of most managers of fish habitat. Nevertheless, soil disturbances there usually speed up water movement; if disturbances are extensive, the size of peak flows will increase. Only the maintenance of intact surface and subsurface soil structure can assure "normal" hydrologic behavior. Stream and upland managers, loggers, forest hydrologists, soil scientists, and terrain specialists should consult broadly with one another to avoid introducing long-lasting and undesirable hydrologic changes when trees are harvested anywhere in a watershed.

Summary of water balance influences.—Timber management activities can affect streamflow by altering the water balance or by affecting the rate at which water moves from hillsides to stream channels. The more severe an alteration of the hydrologic cycle is, the greater the effect on streamflows, and hence on fish habitats, will be.

Changes in flow condition depend on many factors. The expected effects of soil disturbance on flow dynamics are illustrated in Figure 6.4. Another aid for analyzing the net effect of timber harvesting is the "water resources evaluation of non-point silvicultural sources" (WRENSS: U.S. Forest Service 1980a). Discussions by Isaacson (1977) and Toews and Gluns (1986) also are helpful in this regard.

Beyond the semiquantitative modeling techniques that must be calibrated to the characteristics of a specific watershed, the following broad generalizations usually apply.

- Harvesting activities such as roadbuilding, falling, yarding, and burning can affect watershed hydrology and streamflow much more than can other management activities such as planting and thinning.
- Clear-cutting causes increased snow deposition in the openings and advances the timing and rate of snowmelt. The effect lasts several decades until stand aerodynamics approach those of the surrounding forest. Snowmelt can be accelerated by the large wind-borne energy inputs of warm rain falling on snow.
- Harvested areas contain wetter soils than unlogged areas during periods of evapotranspiration and hence higher groundwater levels and more potential late-summer runoff. The effect lasts 3–5 years until new root systems occupy the soil.
- Road systems, skid trails, and landings accelerate slope runoff, concentrate drainage below them, and can increase soil water content.
- Hydrologic models such as WRENSS help predict the net effect of a harvesting pattern and sequence on runoff, but each basin must be analyzed to ensure that the most important hydrologic processes are understood.

Influences on Water Quality

The principal water quality variables that may be influenced by timber harvesting are temperature, suspended sediment, dissolved oxygen, and nutrients. Elsewhere in this volume, forest chemicals are discussed by Norris et al. (1991), the important role of the riparian zone in controlling energy inputs (temperature

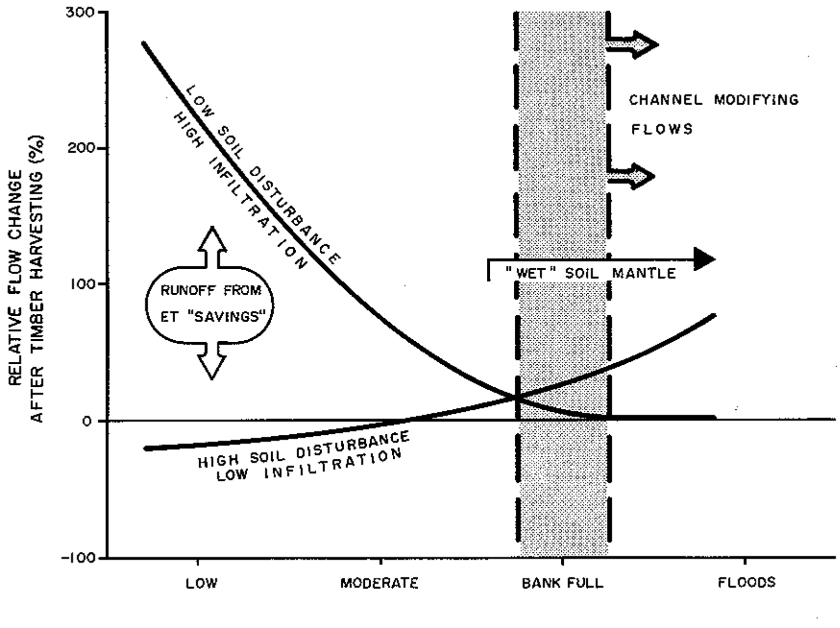


FIGURE 6.4.—Alterations in relative flow after forest harvesting as functions of flow condition when soil disturbances have been small or large. When flow is low, as it may be in late summer, and timber clearance has caused only slight disturbance to the soils, precipitation falling on the soil will infiltrate normally but less will be lost to evapotranspiration (ET) than previously, leaving more water for a sustained augmentation of flow. If soils have been compacted and otherwise disturbed, little rainfall will infiltrate and recharge soil water; most will run off on the surface, causing a transitory peak in flow, and sustained flow will not benefit from an ET "savings." At higher flows, which reflect wetter soil mantles in the watershed, the ET savings become a smaller proportion of the total water budget, so clear-cutting has less relative effect on flow when soil structure remains intact; compacted soils, however, deliver increasing amounts of surface runoff.

and nutrients) is treated by Murphy and Meehan (1991), and the interaction between water quality and fish response is covered by Hicks et al. (1991).

Temperature.—Solar energy is the largest component of energy available to warm stream water in summer. When streamside vegetation is removed, summer water temperatures usually increase in direct proportion to the increase in sunlight that reaches the water surface. Water has a high heat capacity, so a stream's volume, depth, and turbulence affect the actual temperature at any point in the water column. Forest harvesting can cause mean monthly maximum stream temperatures to increase as much as 8°C and mean annual maxima to rise 15°C (Brown and Krygier 1970), but specific stream and watershed conditions cause wide variation in the processes affecting temperature increases (see Beschta et al. 1987 for a comprehensive review).

Figure 6.5 suggests that smaller streams have a greater potential for increases in temperature from streamside harvesting than do larger streams, because a greater proportion of their surface areas will be newly exposed to the sun. However, they may be shaded by smaller trees or deciduous vegetation. Planned openings along

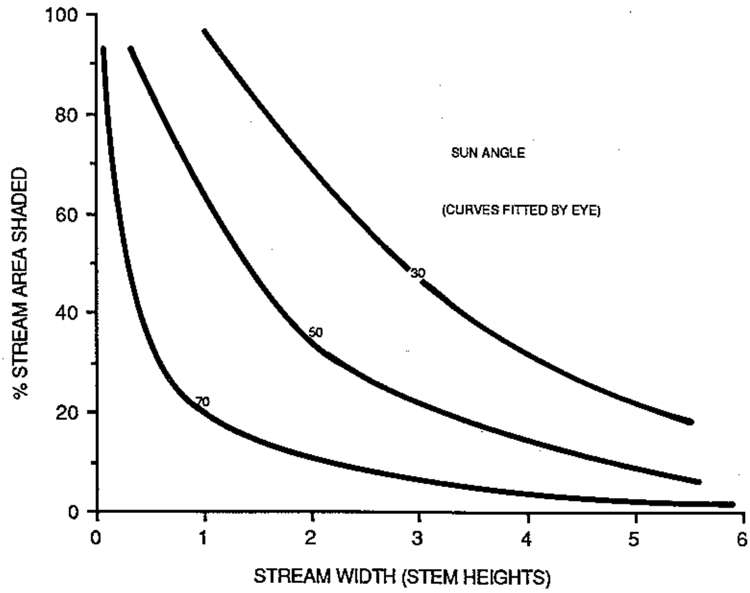


FIGURE 6.5.—Percentage of stream area shaded as functions of stream width and latitude or season. Stream width is indicated as multiples of the height of the prevailing riparian vegetation. Latitude or season is indexed by the noon angle of the sun above the horizontal.

cold coastal streams might enhance fish productivity if other habitat requirements were maintained. In this as in other habitat manipulations, however, caution is required because modest changes in water temperature can change the time required for salmonid eggs to develop and hatch (Holtby 1988a). In northern areas, removal of over-stream coniferous vegetation may lower winter stream temperatures because a net outward energy flow may result, causing slower egg development, deeper surface ice, and bottom-ice formation on gravels. Further, if logging leads to a higher groundwater table in the valley bottom, this will influence the thermal regime of flows in winter (Hetherington 1988)—or, indeed, in any season. Only a careful analysis of the energy balance, including groundwater influences, can indicate the likely direction and magnitude of changes in water temperature that forest harvesting will cause. Techniques are available for predicting changes in a stream's heat budget and consequent changes in water temperature (e.g., Brown 1980; Beschta et al. 1987).

Suspended sediment.—Forest harvesting and silviculture can influence suspended sediment concentrations in a variety of ways, all related to the erosion and sedimentation processes discussed earlier. Most streams carry some sediment, and the amount varies seasonally, but we are most concerned with forest management activities that substantially change the magnitude, timing, or duration of sediment transport and overwhelm the ability of salmonids to cope with or avoid the resulting stress.

Poorly designed roads and skid-trail systems are persistent sources of sediment, but so are open slopes whose soils have been exposed by yarding activities, mass movements, scarification, or intense fire. In cold climates, removal of insulating vegetation promotes formation of ice lenses in and frost heaving of soils, facilitating soil movement during spring thaws (Slaney et al. 1977).

Few studies have identified the component of suspended stream sediments originating from harvesting activity alone (without road influence). Some have illustrated that careful, well-planned logging can take place without appreciable sediment production (Packer 1967), whereas others have documented very high sediment levels (Reinhart et al. 1963) as a result of unplanned activity. Furniss et al. (1991) discuss in more detail the very important role that roads have in sediment production.

We cannot overemphasize the importance of maintaining the integrity of the riparian zone during harvest operations. In addition to disturbing surface soil, activities near the stream bank may destabilize channel margins, releasing sands that settle in and clog the streambed gravels (Scrivener 1988a).

Dissolved oxygen.—Concentrations of dissolved oxygen in intergravel spaces may be reduced if fine organic debris accumulates on and in the streambed. The high chemical and biological oxygen demands of such debris and the bacteria on it may persist for long periods until the bottom material is removed by high flows. Logging and skidding near or across small streams obscured by snow are particularly likely to contribute fine organic debris to watercourses during spring runoffs.

Clogging of surface gravels by fine inorganic sediments can restrict intergravel flow enough to lower dissolved oxygen concentrations. This problem usually occurs only when large or persistent volumes of sediment emanate from active road systems, mass soil movements, bank slumps, or destabilized upstream channels (Scrivener and Brownlee 1989).

During extremely low flows, dissolved oxygen concentrations decline in streams (Bustard 1986). Turbulent exchange of gases with the air decreases. Fish and other respiring organisms—including those associated with organic debris—become concentrated in a few channels and in pools that are nearly or completely isolated. If channels are aggraded and pools shallow, the reservoir of dissolved oxygen is small. In summer, high temperatures both accelerate respiration and lower the solubility of oxygen. In winter, ice cover may prevent diffusion of oxygen from air to water. Harvest activities that impose large oxygen demands on streams exacerbate the normal stresses that low flows place on fish.

Nutrients.—Concentrations of inorganic nutrients (e.g., N, P, K, Ca) in streams may increase after logging, but usually by moderate amounts and for short periods (Fredriksen 1971; Scrivener 1982). Likewise, 5- to 10-fold increases in nutrient releases after slash burning have shown rapid returns to earlier levels. The mobilization of nutrients is tempered by their adsorption onto soil particles and by their uptake by microorganisms that decompose stream detritus (Murphy and Meehan 1991).

Streams in which algal production is limited by a particular nutrient (e.g., phosphorus) may have major algal blooms in response to minor increases of that nutrient, if temperature and flow conditions permit. These blooms can harm salmonid production if their remnants settle into interstitial gravel space. For this reason, forest fertilizers, like pesticides, should not be applied within buffer strips along streams.

Effects of Harvests on Erosion and Sedimentation

Forest harvest activities can influence both upland erosional processes and the way that forest streams process sediment in their channels. The degree of influence varies with geology, climate, vegetation, dominant geomorphic processes, and land uses (H. W. Anderson 1971). The episodic climatic history of western basins over hundreds or thousands of years also makes time an important consideration in the analysis of forestry practices (Benda et al. 1987).

Sediments entering stream channels can affect channel shape and form, stream substrates, the structure of fish habitats, and the structure and abundance of fish populations. In the following discussion, we assume that the goal of forest managers is to maintain streams in their "normal" configurations by minimizing changes in the amount of sediment entering and passing through the systems. Although natural stream processes vary substantially from year to year, and streams change inexorably with time, stream reaches retain characteristic properties over much longer times than those encompassed by forest management cycles. It is against those basic properties that the effects of harvest practices are measured.

Changes in Erosional Processes

Swanston (1991) discusses in this volume how sediment originates either from surface erosion of exposed mineral soil or from mass movements such as landslides, debris torrents, slumps, and earthflows. Furniss et al. (1991) add to the discussion of road-related sediment production.

Surface erosion.—The potential for surface erosion is directly related to the amount of bare compacted soil exposed to rainfall and runoff. Hence, road surfaces, landings, skid trails, ditches, and disturbed clear-cut areas can contribute large quantities of fine sediments to stream channels. Not all hillside sediment reaches the stream channel, but roads and ditches form important pathways. For example, gravel-surfaced logging roads increased sediment production by 40% when they were heavily used by logging trucks (Reid and Dunne 1984). In the Clearwater River basin of Washington, the amount of material less than 2 mm in diameter that washed off roads equalled the amount produced by landslides and has contributed to poor gravel quality for spawning coho salmon.

The quality of management planning strongly influences sediment production from forest-harvesting activity, as illustrated by the classic study of Reinhart et al. (1963) on the Fernow Experimental Watershed, West Virginia. Sediment production varied over 3 orders of magnitude according to the degree of planning and care with which the skidder logging was conducted.

As a general rule, surface erosion results from the exposure of mineral soil, and, as we discussed under soil structure above, it is minimized by the use of yarding systems that are well matched to the terrain and soil types. Packer (1967) reviewed several additional examples and concluded that the best erosion control practices are to avoid operations in very wet seasons, to maintain vegetative buffer zones below open slopes, to skid over snow, and to ensure prompt revegetation.

Silvicultural activities that require scarifying the ground or burning can increase sediment production if buffer strips are not left between treated areas and stream channels. Even when burns do not expose mineral soil, a water-repellent layer can form and reduce the ability of water to infiltrate into the soil (Krammes and DeBano 1965; Bockheim et al. 1973), increasing the runoff available for surface erosion.

An indirect effect of burning is loss of the insulating layer of organic matter. In northern latitudes where soils freeze or permafrost occurs, modifying the freeze–thaw relationship can have serious and long-lasting effects on soil structure and sediment production.

Mass movements.—Mass movement of soil is the predominant erosional process in steep high-rainfall forest lands of Oregon, Washington, British Columbia, and Alaska. The frequency of mass erosion events in forested watersheds is strongly linked to the type and intensity of land treatment in the basin (Rood 1984). Although most mass movements are associated with roads and their drainage systems, many originate on open slopes after logging has raised soil water tables and decreased root strengths (O'Loughlin 1972).

The increase in mass soil movement due to clear-cutting varies widely, ranging from 2–4 times in Oregon and Washington (Ice 1985), to 31 times in the Queen Charlotte Islands of British Columbia (Rood 1984). An increase of up to 6.6 times found by Howes (1987) in the southern Coast Mountains of British Columbia is probably closer to the norm. Although this is much lower than the increases associated with roads, the greater total area of clear-cuts may balance the net result on a weighted-area basis (Swanson and Dyrness 1975).

Much can be done to identify slopes susceptible to mass movements through the use of aerial photography and engineering analyses (Swanson et al. 1987). Howes (1987) developed a procedure—based on terrain mapping and slide occurrence—for quantitatively predicting landslide susceptibility after harvesting. It is usually impossible to harvest unstable hillsides without increasing mass movements, however, except perhaps when careful selective logging with helicopter yarding can be done.

When soils are mass-wasted into stream channels, their effects on salmonid habitats depend on the sediment-processing capability of the stream. They might be beneficial if they bring stable rubble and woody debris complexes to "sediment-poor" channels (Everest and Meehan 1981b). Many mass movements bring soil to higher-gradient reaches, however, and the sediment is carried downstream to a deposition zone where it severely impairs the stream's ability to support fish rearing and spawning.

Remedial measures are available to correct surface erosion problems, but they are costly and far from perfect. Correcting the effects of accelerated mass movements may require tens or hundreds of years, because it involves replacement of stable root systems and the creation of new soil. Measures to accelerate revegetation in severely disturbed areas should include planting deciduous trees, shrubs, and grasses; hydroseeding; and mechanically stabilizing gully systems (Megahan 1974; Heede 1976; Swanston 1976; Carr 1985).

Changes in Sediment Processing by Forest Streams

Sediment transport in forest streams involves the detachment and entrainment of sediment particles, their transport, and their deposition. The process repeats whenever flow velocities are high enough to move the stream's available material.

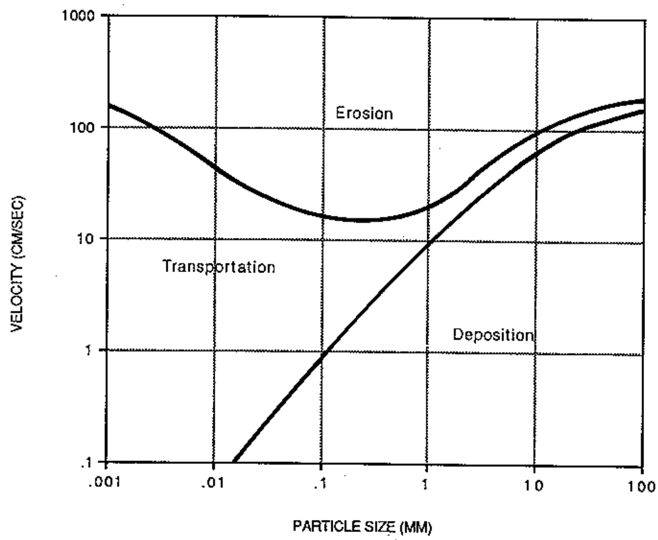


FIGURE 6.6.—Effects of water velocity on sediment particle sizes that are eroded, transported, and deposited in streams. The upper curve indicates the minimum velocities that erode particles of various sizes. Particles of 0.1–1 mm in diameter are the easiest to erode; smaller particles pack more tightly and are bonded by molecular forces, and the mass of larger ones makes them harder to move. The lower curve indicates the minimum velocities that keep particles moving once they already are in motion.

Forest harvesting directly affects these processes when it increases (or decreases) the supply of sediment, when it alters the peak flow or the frequency of high flows, and when it changes the structure of the channel by removing the supply of large woody debris that forms sediment storage sites (Megahan 1982). Bank erosion and lateral channel migration also contribute sediments if protective vegetation and living root systems are removed (Scrivener 1988a).

When additional fine sediments are placed in transport, the intrusion or infiltration of some of the particles into relatively clean or porous surface layers of streambed gravels occurs (Beschta and Jackson 1979). If the source persists, increased amounts may settle deeper into the streambed (Scrivener 1988a) and have longer-lasting effects on egg and fry survival (Hartman et al. 1987).

If the resupply of small sediments from upstream sources is reduced, such as below a debris jam, the gravels become more coarse. In the case of drastic losses of sediment supply, such as below a dam or road sediment trap, downcutting of the channel can occur (Church and Kellerhals 1978).

In streams with gravel beds, most bedload transport takes place during the few "channel-modifying" flow days of each year (see Figure 6.6). Analyses of the hydrologic effects of harvesting on sediment movement and channel change in these streams must focus on whether or not the frequency of such flows will be increased or decreased by the proposed harvest pattern.

Effects of Harvests on Channel Forms and Geomorphic Processes

The fluvial environment is part of a larger watershed ecosystem that includes the floodplain, living vegetation and root systems, and organic debris in and adjacent to the channel. Fish habitats result from a complex interaction among water, sediment, and channel structure. Forest management can affect all of these components, as well as the hydrologic and sediment transport processes discussed above.

Integrating Hydrology, Sediment, and Channel Structure

To anticipate the effects of forest management on fish habitats, one must project changes in the hydrologic and sediment processes against the structural framework of the channel. No single technique exists for this very complex task, although component models have been attempted (e.g., Simons et al. 1982; Sullivan et al. 1987). Descriptive studies, such as the Carnation Creek watershed study in British Columbia (Hartman et al. 1987; C. D. Harris 1988), have shown the results of integrated changes, but quantitative prediction remains difficult because of wide variability in forest streams and a general lack of data. Nevertheless, some important interactions among geomorphic processes are understood, and at least the qualitative magnitude and direction of harvesting effects can be anticipated. For example, streams in which structural elements such as embedded logs have been removed have lost stored sediment to downstream reaches and have generally degraded. When there are fewer "steps" in the stream's profile, more energy is released to move sediment, resulting in a simpler, higher-gradient channel with poorer salmonid habitat. Bisson et al. (1987) extensively reviewed the hydrologic role of large woody debris in channels.

Channel environments are very broadly of two types: alluvial channels, whose form is controlled by a balance between flow regime and the sediments of the valley bottom; and bedrock-controlled channels, whose form is dictated by external structure (bedrock). In both types of channel, large woody debris and tree roots can be secondary controlling structures.

Forest harvesting can affect alluvial systems by weakening channel banks, removing the source of large woody debris, altering the frequency of channel-modifying flows, and changing sediment supply. Unlike bedrock-controlled channels, the alluvial system must change its form in response to geomorphic changes until a new balance between aggradation and degradation has been achieved (Leopold et al. 1964). In alluvial channels, both the removal of bank vegetation and increased sediment supply cause channels to become wider and shallower with fewer pools and more riffles.

Channels with more structural control, such as bedrock in the streambed or banks, large tree root systems in the banks, or armor layers (large rocks), are more stable with respect to fluctuations in flow and sediment supply, and maintain narrower and deeper channels. Even very stable channels can be radically modified, however, by the catastrophic effect of debris torrents.

Off-channel fish habitats in the floodplain such as side and flood channels, ponds, and swamps also can be strongly influenced by forest harvesting. Even in large rivers such as the Willamette River in Oregon, the loss of debris jams and related multiple floodplain channels has vastly reduced channel and shoreline area (Sedell and Froggatt 1984; Figure 6.7).

A special case of side channels occurs in glacial systems, where the clear-water sections fed by groundwater or valley-wall runoff provide the only nonturbid

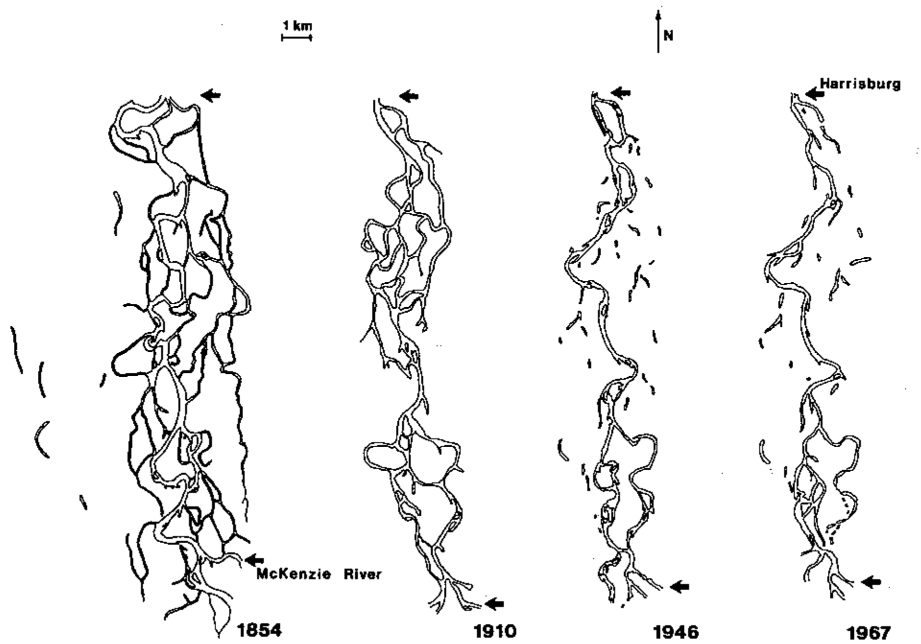


FIGURE 6.7.—Changes in the Willamette River channel, 1854–1967. Arrows show the locations of Harrisburg (top) and the McKenzie River tributary (bottom) in each panel.

habitat in the reach. These habitats are extensively used by rearing chinook salmon fry, and may be important for other species.

Geomorphic processes act over both time and space. If watershed erosion increases, for example, the "new" sediments may persist for long periods as they move through the system under the influence of different streamflows and form alluvial fans, bars, low terraces, sediment wedges behind stable woody debris, or even the streambank or floodplain itself (Hogan 1986). Hence, modifications to a stream system introduced in the early road-building phases of forest harvesting may have to be dealt with decades later when planning begins for the harvest of second-growth timber.

The interactions of hydraulic force, sediment, and channel structure result in geomorphic forms and features that, to a salmonid, are its habitat. We can interpret the hydraulic geometry of stream channels within the framework of fish habitat preferences (Sullivan 1986), and it is useful to focus on these habitat elements as a means of clarifying forest management influences on streams.

Channel Forms and Fish Habitat

Biologists describe stream habitats in terms such as pools, riffles, spawning gravel, obstructions, and side channels, and many classifications are available (e.g., Bisson et al. 1982; Helm 1985). These terms are also geomorphic entities, derived from the processes described above, and they are selectively influenced by different harvesting activities. We will briefly discuss these five habitat elements in the context of the geomorphic processes that control them.

Pools.—Pools are the result of local scour or impoundment induced by structural controls in the channel or streambank. Pools are areas of high water velocity during peak flows, but at low flow their depth creates a depositional environment for fine sediment. Hence, if timber harvesting increases the supply of fine sediments, these sediments settle preferentially in pools, which become less useful to fish. Similarly, if the structural element causing the pool to exist (such as a log or tree root) is removed, the pool will disappear after the next flood flow. Pools are thus very susceptible to falling and yarding operations that influence the availability of large woody debris in or near the channel margins.

Riffles.—Riffles are bars (sediment deposits) with water flowing over them. Because riffles represent the first material deposited after high flows, they usually contain larger particles (gravel, cobbles, and boulders) than are found elsewhere in the stream. Aggrading streams have more depositional areas, and hence have more riffles. Riffles are food-producing areas, but offer few habitats to small fish. Harvesting activities that increase sediment supplies increase the extent and number of riffles. Removal of instream woody structure steepens the stream gradient and hence increases the average size of particles in the substrate.

Spawning gravel.—Spawning gravel is the sorted product of bed scour and redeposition from which sand and finer material has been removed and transported downstream. The maintenance of good spawning gravel requires that the stream's normal sediment supply contain relatively low amounts of fine material, and that flows be sufficiently high to "sort" out the fines that do accumulate. These conditions are often associated with the hydraulic transition zones between pools and riffles; the more transition zones, the more spawning gravels there will be. Hence, harvesting activities that maximize the number of pool-forming structural elements and minimize the influx of fine sediments will favor the maintenance of spawning gravel.

Obstructions.—Obstructions, or barriers to fish migration, are more often associated with road engineering than with timber harvest alone. Culverts or bridges, for example, can cause water velocities to be greater than the swimming ability of small fish (Dane 1978a; Bjornn and Reiser 1991). Excessive debris accumulation, if plugged by sediment, can also block fish passage. Channel aggradation worsens the problem at low flows because water may move entirely below the surface, preventing fish from passing the affected reach. Natural barriers often reflect regional geologic history—resistant rock strata, volcanic intrusions, faults, former sea levels—and may control the distribution of anadromous species over a broad region.

Side channels.—Side channels occur in the stream's margin, or where water is forced out of the channel into the floodplain. Side channels are alternative channel locations, and will remain stable only if their structural controls (usually tree root systems) remain intact. They are vulnerable to timber harvesting in the riparian zone unless harvesting is done with the greatest of care, and they can easily be isolated by dyking or dredging for flood protection, or by road construction without adequate culverts. Side channels have a direct hydrologic relationship to runoff from the valley walls and to the valley groundwater table, and hence may be influenced by many forest management activities.

Summary of Harvesting Influences on Channels

Four major timber management effects can modify a stream's geomorphic process and forms.

- Substantial increases in peak flows or the frequency of channel-modifying flows from increased snowmelt or rain-on-snow events can increase bed scour or accelerate bank erosion. Quantitative assessments of channel stability (Pfankuch 1975) and an analysis of flows at which normal channel changes begin will help determine whether flow increases may be important.
- Substantial increases in sediment supply from mass movements or surface erosion, bank destabilization, or instream storage losses can cause aggradation, pool filling, and a reduction in gravel quality (Madej 1982). Assessments of initial habitat condition (Binns and Eiserman 1979; Bisson et al. 1982; de Leeuw 1982) and estimates of the natural variability in sediment regime (Swanson et al. 1982b) will assist in determining whether sediment-supply increases will be meaningful.
- Streambank destabilization from vegetation removal, physical breakdown, or channel aggradation adds to sediment supply and generally results in a loss of the channel structures that confine flow and promote the habitat diversity required by fish populations (Forward [Harris] 1984; Scrivener 1988a).
- Loss of stable instream woody debris by direct removal, debris torrents, or gradual attrition as streamside forests are converted to managed stands of smaller trees will contribute to loss of sediment storage sites, fewer and shallower scour pools, and less effective cover for rearing fish.

Cumulative Effects of Forest Harvesting

In earlier sections, we described how hydrologic, sediment, and channel processes can be changed by timber-harvesting activities, and hence can affect salmonid habitats. These processes operate over varying time scales, ranging from a few hours for coastal streamflow response to decades or centuries for geomorphic channel change and hill-slope evolution. They are also distributed spatially over the landscape, progressively influencing more land area as timber management extends within watersheds and across regions. The consideration of how harvesting influences the landscape and fish habitats through space and time is the subject of this section.

Identifying Cumulative Effects

Observing and identifying cumulative effects of timber management on biophysical processes or fish habitats are difficult, not only because of technical complexity but also because few research efforts have been sustained or focused over the necessary time periods. Nevertheless, some studies now help demonstrate cumulative effects of logging on streams, beginning with the Wagon Wheel Gap (Colorado) snow accumulation and melt experiment, started in 1910 (Holscher 1967), and carrying through to the Carnation Creek watershed study (Hartman 1988), which began in 1972.

In addition to long-term watershed studies, several specific research techniques exist to provide information about cumulative effects. These include time-series analysis of historical aerial photography, tree ring and similar vegetative dating techniques, and standard geologic techniques applied to recent sediment deposits. Systematic review of historical media and file reports has helped define management treatments and effects on Pacific Northwest river systems (Sedell and Froggatt 1984).

Finally, synoptic survey designs such as those of the Fish/Forestry Interaction Program in the Queen Charlotte Islands (Poulin 1984) can address cumulative effects by simultaneously examining many watersheds in various stages of forest-harvesting development. A synoptic survey design has been proposed for a major new research initiative by the Pacific Biological Station of the Canada Department of Fisheries and Oceans (I. Williams, personal communication).

Management actions to deal with cumulative effects suffer from constraints other than lack of information. The costs of dealing with cumulative effects often are large and must be borne by agencies or jurisdictions that may be reluctant to act because of poor short-term benefit:cost ratios. The question of future accountability for historical effects on public or private fisheries resources is increasingly important as management responsibility shifts to or from the private sector. The long-term effects of changes in physical processes on fish habitats are also confounded by various intensities of use by commercial and sport fisheries, and by urban and industrial development.

Despite these qualifications, the previous discussions of processes linking timber harvest and silviculture to salmonid habitats suggest five main categories of cumulative effects:

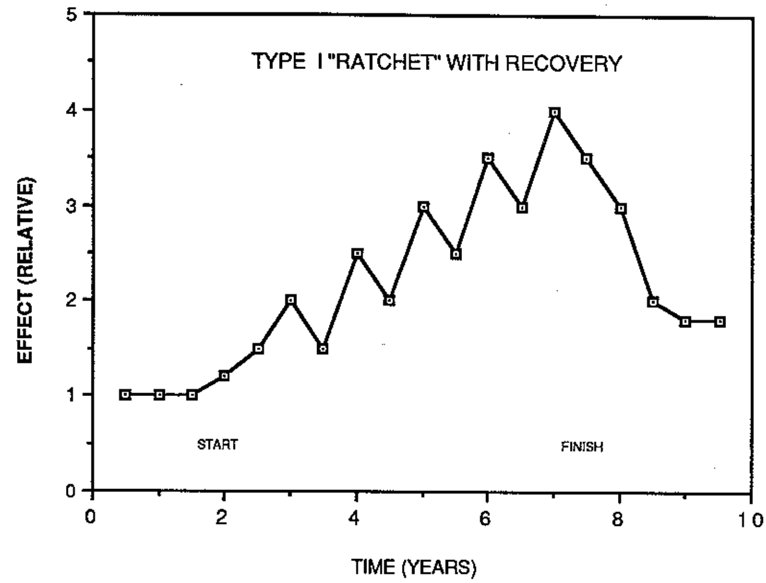
- changes in timing or magnitude of small or large runoff events;
- changes in the stability of stream banks;
- changes in the supply of sediment to channels;
- changes in sediment storage and structure in channels, especially those involving large woody debris; and
- changes in energy relationships involving water temperature, snowmelt, and freezing.

The time frame of these changes, especially in the context of normal forest management planning, defines their "cumulative" nature. The persistence of and recovery from changes in the stream ecosystem form a useful analytical framework for examining typical cases of cumulative effects.

Persistence and Recovery

Although change is a normal feature of stream ecosystems, the amount of change tends to vary within limits that are characteristic of a given stream when flow regime, sediment supply, and channel structure are not perturbed. Biological systems have analogous properties, both in individuals and in populations (i.e., homeostasis).

Geologists have also discovered that some rivers have historically undergone episodes of major sedimentation and erosion associated with hundred- or thousand-year events (Benda et al. 1987) and recent climate changes may have profound effects on channel equilibrium. However, we will consider cumulative effects in the time scale of forest-harvesting activity.



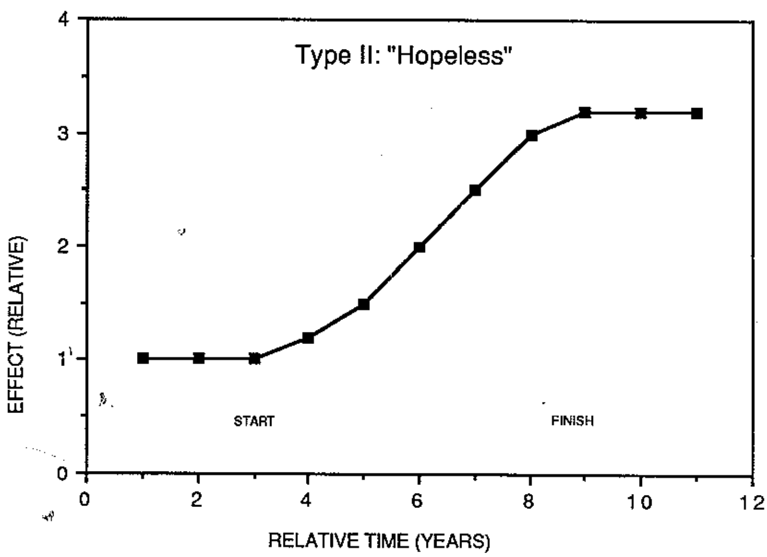


FIGURE 6.8.—Cumulative effects of forest harvesting on streams: type 1 (incremental but reversible changes) and type 2 (irreversible changes).

Foresters and other land managers purposefully impose changes on the ecosystem. We will examine four cases that illustrate different degrees of persistence and recovery. For convenience, they can be classified into two general types, those from which some degree of recovery is possible (type 1) and those from which it is not (type 2). Figure 6.8 graphically illustrates the two types.

Type-1 cumulative effect: incremental change.—Type-1 situations involve management-induced effects that individually are not overwhelming, but that, if compounded, will continue to force the stream into new configurations to the detriment of fish habitat. Type-1 cumulative effects can be corrected if necessary management actions are taken.

The history of forest harvesting in the South Fork Salmon River, Idaho (Platts and Megahan 1975), illustrates a case in which successful recovery occurred. An analysis of this system's history by Sullivan et al. (1987) identified accelerated sedimentation caused by logging and road construction as the source of sediments that gradually inundated pools and degraded gravel quality for spawning fish.

A 12-year moratorium on logging activity, together with watershed rehabilitation measures, allowed the stream to export the fine material; gravel quality was restored and pools were reestablished. The stream was able to recover because sediment input was controlled and because the riparian vegetation remained intact, preserving the structural framework required for normal pool formation and gravel-sorting processes.

Another case of incremental degradation is illustrated by Carnation Creek, British Columbia. In this experimental basin, treatments involving riparian-zone timber removal over a 5-year period, compounded by an upstream debris torrent, have destabilized extensive sections of stream bank and channel (C. D. Harris 1988). In these sections, the channel has widened and established side channels through the adjacent floodplain (Powell 1988). Gravel quality in lower sections of the stream has progressively deteriorated and shows no signs of recovery (Scrivener and Brownlee 1982, 1989).

Recovery in Carnation Creek may be possible if stabilizing riparian vegetation is reestablished, large woody debris is reintroduced to the channel, and sufficient time is allowed to flush accumulated fines from the stream gravels. Time required for recovery will depend on the extent of purposive management actions.

Type-2 cumulative effect: irreversible change.—Type-2 cumulative effects involve changes to the basic watershed processes from which recovery is not possible because of very long time requirements, permanent shifts in social and economic objectives that preclude the required management action, or both.

Sedell and Froggatt (1984) documented the history of the Willamette River from 1854 to 1967. To facilitate transportation and log driving, the river was gradually cleaned of log jams, debris, and streamside trees. Most side channels were logged and cut off, resulting in a much-simplified channel (Figure 6.7) that has vastly reduced shoreline and off-channel habitats. These habitats will never be recovered due to the necessities of navigation and alternative uses of the floodplain land. In some large rivers, however—such as the lower Fraser River, British Columbia—habitat creation to offset industrial alienation may be possible.

A second case of type-2 cumulative effects is illustrated by old-growth timber harvesting adjacent to medium-size or large rivers that depend on very large woody debris. Rivers such as the Yakoun in the Queen Charlotte Islands, British Columbia, depend on spruce log debris 1–3 m in diameter for channel structure and bank stability (de Leeuw 1988). As a result of their loss, the channel is widening and redistributing stored gravel through processes similar to those in Carnation Creek.

Current forest management practices in the Queen Charlotte Islands call for a managed second-growth stand of 80- to 120-year-old trees, so a permanent shift in the size distribution of available large woody debris seems inevitable. This condition is replicated throughout the Pacific Northwest as industrial forests are converted from old-growth to managed stands. Its reversal will depend on implementing a rotation age of 300 or more years for riparian stands. Other resource values (e.g., wildlife and recreation) may contribute to the feasibility of this option in some areas.

These examples clearly show that forest-harvesting activity can have lasting and cumulative effects on fish habitat. Whether the effects can be overcome depends partly on the degree to which stream processes are distorted but more importantly, on the management time scale within which action can be taken. The next section discusses management options to minimize undesirable and maximize desirable effects on salmonid habitat.

Conclusions and Management Options

Management options for ensuring productive fish habitat have evolved considerably over the last 10–20 years as we have learned more about how stream and forest ecosystems function. In this section, we briefly examine the evolution of some successful logging guidelines and then some new directions that may be available for habitat management over the next few decades.

Evolution of Logging Guidelines

In the 1950s and 1960s, planning for fish habitat management took place, if at all, on the streambank, with a biologist and a forester examining the site and using experience and persuasion to arrive at an acceptable management plan. Decisions were usually made with the information at hand for that site, and supported by the policies of the agency with controlling jurisdiction. Much was assumed about what was good or bad, and stream "cleaning" was very popular. Many wheels, both round and square, were reinvented at each "on-site" inspection.

During the 1970s, fisheries agencies became increasingly involved in forest harvest planning and assessment, and began to consolidate and codify approaches to habitat management under various regulatory bodies for forest practices. This had the advantage of encouraging consistency, but the disadvantage of inflexibility with respect to differences among sites and processes. The early "P" or protection clauses used in British Columbia are good examples of guidelines that became inflexible rules (Brownlee and Morrison 1983).

Early guidelines tended to focus on practices that influenced water quality because research on large woody debris and channel geomorphology was not well established. There was considerable reliance on the mitigating influence of the streamside "buffer zone" of arbitrary width (e.g., 10 chains), without reference to the biophysical processes it was influencing. Stream classes, when used (Oregon, Washington), were based on relatively simple criteria such as species "significance" or stream width (e.g., 1 m, 10 m).

During this era, numerous guidelines were developed that specialized in regionally limited procedures for particular forest practices. For example, Packer and Christensen's (1964) classic handbook on how to retard the surface transport of sediment on interior slopes still has application, and many jurisdictions (e.g., Toews and Wilford 1978; Harr 1981) proposed total cut limitations (percentage of watershed) to minimize runoff impacts.

However well intentioned, guidelines tended to become rules and even were incorporated into law in many states. Yet, as we have seen in this chapter, the quality and care with which logging is carried out can have much more bearing on fish habitat quality than what is specified in a planning document. In the 1980s, this knowledge was explicitly recognized in the development of the British Columbia Coastal Fishery/Forestry Guidelines (B.C. Ministry of Forests et al. 1988). In this document, state-of-the-art guidelines are presented as possible means to achieve various defined levels of habitat protection, which, in turn, are related to a stream reach's fishery value. However, the opportunity to devise better ways of meeting these levels of protection is left open to the initiative of the industries and agencies involved. This management philosophy is consistent with two important factors in guideline evolution today: the increasing "privatization" of public resource management responsibilities, and the increasing use of detailed site information and models instead of generalized guidelines.

New Directions

The challenge of resource managers today is both to understand the watershed processes that are important for a given decision and to have enough site information about that watershed to apply the knowledge. This book contributes to the first objective and suggests important types of data that should be gathered, but neither will be of much value without a management framework in which they can be used.

Two important ingredients are necessary to take advantage of *both* knowledge and data. The first is a cooperative attitude between agencies and industry, predicated on commitment from the most senior political levels to the integrated and sustainable management of resources. The second is the development of a technical information infrastructure that makes possible the sharing of knowledge and information *in the planning and decision-making environment*. Both the political and the technical support legs must be in place to move beyond the "spearchucking" days of the not-too-distant past.

Several extensions of current forestry and fisheries management policies have been suggested in previous sections. We list here some that we feel may contribute to the new directions we are seeking.

- *Long rotations.* Large trees have been proven necessary for the maintenance of channel integrity and productivity in most forest streams (Kaufmann 1987). They also contribute to many nonfisheries values. Yet rotation ages of more than 120 years (and much less on high-site land) seem absent from harvesting plans, despite their technical and economic feasibility. Urgent reevaluation of management strategies for remnant old-growth and older second-growth forests seems warranted.
- *Clear-cut stability modeling.* Distributed small-patch cuts have important advantages in some ecosystems, especially where snowpack manipulation is a priority. However, their universal application as a magical panacea, as with "leave strips" (strips of uncut trees between patches) is inappropriate in unstable or windthrow-prone terrain where road construction and edge effects should be minimized. In either situation, stability modeling would provide important

management direction, and is almost universally lacking in normal harvest planning.

• *Privatization.* Some policy analysts (e.g., Pearse 1988) have suggested that the private sector (forest industry) should be given an increased role in the management of fish habitats in exchange for options on the forest resource. This approach offers savings of public management moneys in appropriate jurisdictions, but will be effective only if desired habitat and fishery values can be identified (Platts 1974; Paustian et al. 1983) and an effective performance audit is supported.

• *Regional index streams.* Very little stream assessment occurs after timber is harvested. Yet the postlogging condition of habitats is the best indicator of adequate harvesting practices. Index streams, considered typical of a regional situation, could be monitored as a check on policy similar to the ambient water quality monitoring of many states and provinces, and the index survey streams of the U.S. Geological Survey. When carried out over several decades (as, for example, in Carnation Creek), these assessments would focus discussion on processes and practices amenable (or not) to change. Without such assessments, most discussions about cumulative effects will remain academic.

• *Accelerated habitat restoration and enhancement.* In addition to maintaining existing stream habitats, managers need to identify opportunities to restore degraded streams to productive capacity. In most streams this means recreating geomorphic structures and sediment-storage opportunities through techniques such as placing logs or boulders, augmenting off-channel habitats, restoring riparian vegetation, and rebuilding fisheries stocks. A combination of restoration and enhancement measures may be necessary. For detailed examples see Canada Department of Fisheries and Oceans and B.C. Ministry of the Environment (1980) and Bustard (1984).

We conclude this section on management options, as we began the chapter, by suggesting that both timber and fish can be successfully managed in a watershed if timber and fishery managers communicate their needs and coordinate their activities. The technical knowledge base is more than sufficient if the necessary policies and attitudes are in place to support its use.

Logging Impacts of the 1970's vs. the 1990's in the Caspar Creek Watershed[1]

Peter H. Cafferata[2] and Thomas E. Spittler[3]

***Abstract*:** *The Caspar Creek watershed study provides resource professionals with information regarding the impacts of timber operations conducted under varying forest practices on sensitive aquatic habitats. In the South Fork watershed, roads were constructed near watercourse channels in the 1960's, and the watershed was selectively logged using tractors during the early 1970's. Subwatersheds in the North Fork were clearcut from 1985 to 1991 using predominantly cable yarding and roads located high on ridges. Numerous landslides were documented after road construction and logging in the South Fork owing to inadequate road, skid trail, and landing design, placement, and construction. In contrast, the size and number of landslides after timber operations in the North Fork to date have been similar in logged and unlogged units. Considerably more hillslope erosion and sediment yield have also been documented after logging operations in the South Fork, when compared to the North Fork. An analysis of the storm events associated with documented landslides showed that high 3-day or 10-day precipitation totals in combination with moderately high 1-day amounts have been more important than very high 1-day totals alone in triggering debris sliding at Caspar Creek. Storm sequences meeting the criteria required for causing documented landslides were found to have occurred in all phases of the 36-year study, with the greatest number occurring in water year 1998. Numerous large landslides associated with the road system in the South Fork occurred in early 1998, indicating that "legacy" roads continue to be significant sources of sediment decades after they were constructed.*

The impacts of harvesting and road construction in a second-growth redwood/Douglas-fir forest have been studied for 36 years in the Caspar Creek watershed. This allows us to compare the impacts from the first phase of the project, completed before the implementation of the modern Forest Practice Rules in California, with those associated with considerably improved forestry practices. Specifically, in the South Fork, roads were constructed in 1967, and the entire basin was selectively harvested from 1971 to 1973, before the enactment of the Z'Berg Nejedly Forest Practice Act of 1973. Approximately 6.8 km (4.2 mi) of road were built low on the slopes in the watershed, much of it adjacent to the South Fork channel, and tractors were used to skid logs to low-slope landings. Some of the skid trails were built in small stream channels. In contrast, 47.8 percent of the North Fork, within 10 nested subwatersheds, was clearcut from 1985 to 1992 using 11.4 km (7.1 mi) of existing roads and 8.4 km (5.2 mi) of new roads located high on the ridges (Preface, fig. 2, these proceedings). The steeper slopes were cable yarded. This long-term instream monitoring study provides resource professionals in California with information regarding the impacts of timber operations with varying forest practices on sensitive aquatic habitats.

In this paper, we present a discussion of the geology and geomorphology present in the Caspar Creek drainage, as well as a summary of the major erosional sources which have followed logging in the gaged portions of each tributary. Additionally, we compare and contrast rainfall and runoff events that occurred during both phases of the study. A summary of the sediment yields documented during the life of the study is presented, and changes in sediment generation attributable to improved forest practices are discussed. Finally, recommendations are offered to forest managers regarding the applicability of Caspar Creek results to other California watersheds.

Geology and Geomorphology

Physiography

The North Fork of Caspar Creek above its weir drains a watershed of 473 ha (1,169 ac), in northern California, whereas the area above the South Fork weir is approximately 424 ha (1,047 ac). These small watersheds, located inland from the central Mendocino County coast, are about 11 km (7 mi) southeast of Fort Bragg. The low point of each experimental watershed is at its weir, 85 m (275 ft) for the North Fork and 50 m (160 ft) for the South Fork, with the high points 310 m (1,020 ft) and 320 m (1,057 ft), respectively.

Geology

Both watersheds are underlain by the Coastal Belt of the Franciscan Complex (Kilbourne 1982, 1983; Kilbourne and Mata-Sol 1983). Well-consolidated marine sedimentary sandstone with intergranular clay and silt (graywacke) and feldspatic sandstone, with lesser amounts of siltstone, mudstone, and conglomerate, are the dominant rock types. The sandstones are poorly bedded to massive, and moderately well consolidated. Individual exposures range from coarsely jointed sandstone that is moderately hard and strong to highly fractured to sheared rock that exhibits low strength.

Alluvium of Holocene age is locally present in both watersheds. A significant accumulation of this material is present in the upper portion of the North Fork watershed. The alluvium consists of loose to somewhat indurated, poorly sorted sands, gravels, and silts that were deposited behind barriers or locally along low-gradient segments of the stream channels. An extensive area of alluvium was deposited behind an ancient landslide dam in the North Fork. The

[1] An abbreviated version of this paper was presented at the Conference on Coastal Watersheds: The Caspar Creek Story, May 6, 1998, Ukiah, California.

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remnant of the landslide dam is an inclined (depositional inclination) deposit of poorly sorted sand, silt, and gravel that forms a perched bench on the north bank. The geometry of the remaining sediment indicates that this old landslide transported on the order of 1 to 5 million cubic meters of material.

Geomorphology

Both the North Fork and South Fork of Caspar Creek flow in relatively narrow, bedrock-controlled stream channels. The upper portion of the North Fork of Caspar Creek flows in a narrow gully through Holocene alluvium that was deposited behind the landslide dam. Carbon-14 (^{14}C) dating of carbonaceous material, found by Dr. Stephen Reneau (1989) in the alluvium upstream from the landslide dam, indicates that this material slowly filled the valley beginning about 7,000 years before present (BP) through at least the late Holocene. Deposition stopped only recently, after which renewed downcutting occurred. This indicates to us that the broad (approximately 100 m [330 ft] wide) debris flow deposits formed a long-lasting dam that resisted downcutting for thousands of years.

Logging of the old-growth forest in the Caspar Creek watershed occurred between the 1860's and 1904. All large, accessible trees that would generate high-quality forest products were felled, topped, and limbed on the slopes. Included in the harvest were the trees that lined the stream channels and all partially buried logs and other structural features that could impede the transport of logs down the main channels during future floods. After the limbs and tops had dried (and the logs had lost a great deal of water weight), the watersheds were burned to clear away the logging slash. The resistance to fire of the old-growth redwood protected the logs from immolation. Fire scars are still visible on old-growth trees that were not harvested. After burning, the bucked logs were skidded to the main stream channels by oxen (a small percentage of the North Fork watershed was logged later starting about 1900 with steam donkeys). Small stream channels were used as skid roads, with corduroy logs half buried, heavily greased, and evenly spaced at intervals equal to the stride of the oxen that were teamed to pull the log trains.

During this time dams were constructed across steep, narrow reaches in the headwaters of the two forks of Caspar Creek. Remnants of the splash dam in the North Fork are visible along the steep, narrow channel at the old landslide dam. During the winter, when the streams were at flood stage and the reservoirs behind the dams were full, the gates would be opened and the resulting flood would raft the accumulated logs to the mill at the mouth of Caspar Creek. The transport of logs down the streams continued for roughly 25 years until most of the watershed had been logged (Wurm 1986).

It is clear that the artificial floods and the removal of the riparian vegetation and in-channel large woody debris had a profound impact on channel geometry, stream bank stability, and sediment discharge. Napolitano (1996) documents that the depletion of large woody debris from the stream channel, caused by the clearing and flooding, was so severe that the streams have not yet recovered. However, the consequences of the logging on the hillslopes did not persist. Other than a few remnants of corduroy skid roads in small channels, little evidence of the original logging is still present. In fact, in the North Fork watershed, the majority of which was never logged with tractors, the well-preserved details of the surface morphology allow the geomorphic mapping of debris slides, debris flows, rotational landslides, disrupted, hummocky ground, and inner gorges.

The geomorphic mapping of the North Fork took place at intervals between 1986 and 1994. The upper portion of the watershed was field mapped using the published topographic map that was photographically enlarged to a scale of 1:6,000. The remainder of the North Fork was mapped using aerial photographs and limited field reconnaissance on a 1:12,000-scale, photographically enlarged topographic base. The quality of ground surface exposure in the North Fork allowed the landslides to be subdivided into five relative age classes (Spittler and McKittrick 1995): (1) fresh-appearing landslides that were most recently active within the past 20 years; (2) landslides that have affected the second-growth trees but have recovered to some degree—estimated to be from 20 to 120 years old; (3) landslides that have affected the old-growth trees or stumps, but have not affected the second-growth trees—estimated to be from 100 to 1,000 years old; (4) landslides that have not affected the old-growth trees or stumps but have well-defined surficial morphologies—estimated to be from 500 to several thousands of years old; and (5) geomorphic features with morphologies suggestive of landsliding but that are highly modified. These last features may be related to differential erosion of inhomogeneous bedrock, perched ancient erosional surfaces, or ancient landsliding.

In contrast to the logging history of the North Fork, the logging history of the South Fork of Caspar Creek has resulted in a significant impairment in our ability to map geomorphic features. Mapping was conducted by using aerial photographs taken for this project, and photos taken in 1975 after the more recent logging was completed. Field work was impeded by ground surface modifications and dense regeneration. The South Fork watershed was affected by a high degree of ground disturbance that occurred as a result of road construction in 1967 and tractor logging between 1971 and 1973, before the implementation of modern forest practices. This disturbance has modified surface features to the extent that only larger landslides, and those that occurred following the timber harvesting, are well defined. The landslide incidence and sediment yield data suggest that the ground disturbances affected more than our ability to map landslides. Although the South Fork was selectively logged, the persistence of the surface disruption during the 25 years since the logging was completed suggests to us that the recovery is very slow.

Within the North Fork watershed, only one small landslide[4] was observed in the clearcut units of the North Fork of Caspar Creek between the beginning of logging in 1985 and the end of the geologic study in late 1994. This was a failure from a yarder landing in

[4] Landslides are defined here as those greater than or equal to 76 m^3 (100 yd^3). Additional smaller features were also recorded.

subwatershed G that occurred during an unseasonable storm event in late May 1990. Of the 15 other landslides in the watershed that were fresh appearing, seven are associated with the existing roads across the upper slopes and eight occurred in areas not adjacent to roads. Other than the subwatershed G feature, landslides in the harvested blocks appear to predate the timber harvesting on the basis of the age of vegetation growing on the scars. Eight of the fresh-appearing landslides are larger than 0.2 ha (0.5 ac) (*fig. 1*). Of these, all but one are associated with the older roads. After the completion of the geologic study, a debris flow in the YZ subwatershed transported about 3,600 m^3 (4,700 yd^3) in January 1995. Seven other small failures have been documented in the North Fork since the start of 1995. Three small failures occurred in both clearcut blocks and areas outside of harvest units.[5] In addition, one fill failure occurred that was associated with an existing road in a clearcut subwatershed.

Unlike the North Fork watershed, where only one small landslide was related to roads, skid trails, or landings constructed for the recent predominantly cable clearcut logging, almost all of the smaller, more recent landslides in the selectively cut South Fork watershed are associated with these types of disturbance features. Interpretation of aerial photographs of the South Fork of Caspar Creek from 1975 revealed 66 recently active landslides, all of which appear to be debris slides or debris flows. Of these, 35 are associated with roads, 12 with landings, and 16 with skid trails, with three not associated with ground disturbances and not within the area that was selectively logged. Seventeen of the post-logging landslides were larger than 0.2 ha (0.5 ac). Of these larger landslides, six are associated with roads, seven with landings, and three with skid trails; one is not associated with the timber operations (*figs. 2, 3*).

The number and relative sizes of post-harvesting landslides differ substantially between the North Fork and South Fork of Caspar Creek. Within the North Fork, 10 landslides have been reported to have occurred since the beginning of operations in 1985, including two in 1998. Of these, six are associated with logged units and four are in unlogged portions of the watershed. Only one landslide failed in the North Fork that exceeded 0.2 ha (0.5 ac). In contrast, during the first 8 years after the initiation of road construction within the South Fork watershed, 66 landslides, 17 of which are larger than 0.2 ha, failed.

During the El Niño storm year of 1997-1998, which is the winter of record for precipitation during the life of the Caspar Creek watershed study (greater than 2,030 mm [80 in.] total precipitation), only one small landslide was reported for recently logged or roaded units in the North Fork watershed. This slide feature is approximately 76 m^3 (100 yd^3) and occurred along the North Fork in a unit clearcut in 1990; it is a located immediately above the streambank and is actually a reactivated slide mapped earlier in the study. Another slide of similar size took place in uncut tributary H (*table 1*). In contrast, landslides of 1376, 149, 68, 57, and 25 m^3 (1800, 195, 89, 74, and 33 yd^3) have been documented in

[5] Data from North Fork Caspar Creek Large Event Inventory, supplied by Elizabeth Keppeler, USDA Forest Service, Pacific Southwest Research Station, Fort Bragg, California.

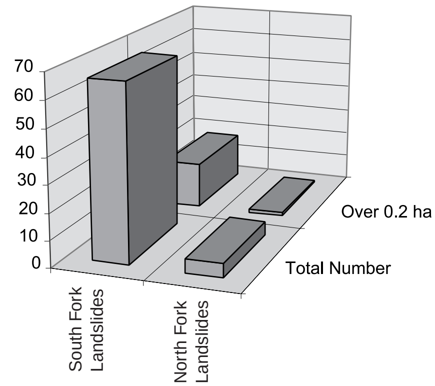

Figure 1—Comparison of the total number of landslides and number of landslides over 0.2 ha after timber harvesting activities in the North Fork (until 1996) and South Fork (until 1975) of Caspar Creek.

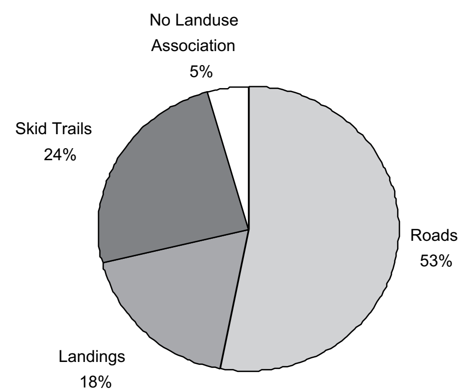

Figure 2—Distribution of all landslides after 1967 road construction and 1971-1973 logging as interpreted from 1975 aerial photographs, South Fork Caspar Creek.

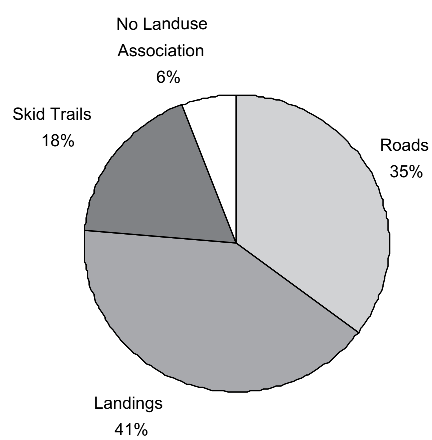

Figure 3—Distribution of landslides over 0.2 ha after 1967 road construction and 1971-1973 logging as interpreted from 1975 aerial photographs, South Fork Caspar Creek.

the South Fork watershed that were related to the old road system and have discharged substantial amounts of sediment into the stream. In addition, a 420-m^3 (550-yd^3) feature associated with the old skid trail network and an inner gorge slide of 168 m^3 (220 yd^3) not related to old roads or skid trails occurred (Keppeler 1998).

Hydrologic Considerations

Rainfall and Hillslope Failures

Rainfall characteristics are well-correlated with landslide initiation. Deep-seated failures are heavily influenced by seasonal precipitation amounts (Sidle and others 1985). In contrast, shallow rapid failures, such as the debris avalanches and debris slides that have occurred in the Caspar Creek watershed, are generally triggered by a critical combination of rainfall intensity and duration (Cannon and Ellen 1985). Intense rainfall can exceed the rate of hillslope drainage, causing the piezometric surface to rise and generating positive pore water pressures within the soil that can ultimately cause slope failure (Campbell 1975).

Timber operations can alter hillslope drainage patterns. Harvesting timber in a small swale in the North Fork of Caspar Creek elevated pore water pressures from 9 to 35 percent above background levels during the first 4 years after logging, but did not initiate slope failure. On slopes with known or suspected stability problems, however, additional pore water pressure generated by timber harvest may increase the risk of landsliding (Keppeler and others 1994). Mid-swale road construction along with timber harvest in the North Fork produced dramatic increases in pore water pressures in and up-slope of the road prism (Keppeler and Brown, these proceedings). LaHusen (1984) reported similar results in Redwood National Park. He documented greatly increased pore water pressures within roadfill material, with a corresponding two- to five-fold decrease in hydraulic conductivity, and concluded that intense rainfall events can create pronounced "groundwater mounds" in road prisms that eventually culminate in debris flow initiation.

Campbell (1975) and Wieczorek and Sarmiento (1983) found that 25 to 38 cm (10 to 15 in.) of antecedent seasonal precipitation can ready a hillslope for debris slides. Once field capacity of the soil mantle has been reached, a storm with extreme 12- to 24-hour precipitation can cause shallow debris avalanche failure. For 24-hour storm duration, a failure threshold has been shown to occur at a rainfall intensity of 0.7 cm hr^{-1} (0.3 in. hr^{-1}) in the part of the San Francisco Bay area with mean annual precipitation of more than 66 cm (26 in.) (Cannon and Ellen 1985), and a failure threshold of 0.43 cm hr^{-1} (0.17 in. hr^{-1}) has been reported by Caine (1980) using data in the worldwide literature. Caine (1980) used published records of rainfall intensities and durations associated with shallow landsliding to develop a rainfall-debris flow threshold equation for durations from 10 minutes to 10 days.

Rainfall Records for Caspar Creek

Before we can draw valid conclusions regarding the relationship between landsliding and sediment generation with forestry practices, we must assess the number and relative sizes of stressing storms that occurred throughout the calibration period for both watersheds (water years 1963-1967), the South Fork road construction, logging, and recovery period (1968-1978), and the North Fork road construction, logging, and recovery period (1986-1998).

Daily precipitation values are available for the South Fork of Caspar Creek for hydrologic years 1963 through 1998.[6] Goodridge (1997) provides rainfall depth duration frequency data for the South Fork Caspar Creek 620 station.[7] From this information, we plotted

Table 1*—Precipitation amounts from storm periods associated with known landslides greater than 76 m^3 (100 yd^3) in the North Fork of Caspar Creek (exact dates of landslides during storm events are assumed in some cases).*

Slide date	Subwatershed	Slide Vol (m^3)	Slide Vol (yd^3)	1-Day Total (cm)	3-Day Total (cm)	5-Day Total (cm)	10-Day Total (cm)	API (cm)
March 31, 1974[1]	L-not logged	3306	4324	6.17	11.79	17.63	20.09	17.30
Feb 16, 1986	L-not logged	1262	1650	4.95	11.94	15.95	16.13	16.92
May 27, 1990	G-logged	283	370	4.88	12.40	12.40	23.77	17.81
Jan 9, 1995	YZ-logged	3606	4715	5.97	15.62	19.71	23.67	20.80
March 14, 1995	A-not logged	306	400	5.11	11.38	15.98	25.58	20.07
March 14, 1995	G-logged	76	100	5.11	11.38	15.98	25.58	20.07
March 14, 1995	C-logged	130	170	5.11	11.38	15.98	25.58	20.07
Jan 24, 1996	E-logged	84	110	5.13	7.42	10.85	23.37	16.56
Dec 31, 1996	H-not logged	122	160	8.59	17.60	19.18	23.90	23.55
February 1998[2]	L-logged	76	100	—	—	—	—	
March 22, 1998	H-not logged	103	135	5.08	12.57	12.57	12.62	15.16
Mean				5.74	12.59	15.53	21.14	18.52

[1]The date of this slide feature was assumed based on large amounts of precipitation at the end of the month. Slide volume is from Rice and others (1979).
[2]The date of this feature is unknown, preventing the association of rainfall amounts with the landslide feature.

[6] The hydrologic year is defined as beginning on August 1st for the Caspar Creek watershed. Daily precipitation values are determined from midnight to midnight.

[7] Data missing from Goodridge (1997) were obtained from the USDA Forest Service's Pacific Southwest Research Station's Internet site (http://www.rsl.psw.fs.fed.us/projects/water/caspar.html).

the 1-, 3-, 5-, and 10-day annual maximum rainfall totals to determine the frequency and size of stressing storm events over the life of the study. For example, the 1-day annual maximums are displayed in *figure 4*.

In Goodridge's (1997) analysis, 11.07 cm (4.36 in.) of precipitation over a 1-day period constitutes a 5-year return period event, 12.88 cm (5.07 in.) represents a 10-year 1-day event, and 16.59 cm (6.53 in.) is the 50-year 1-day event. Before any modern disturbances in either the South Fork or the North Fork, 5-year rainfall events occurred in water years 1964, 1965, and 1966. After road construction in the South Fork, another 5-year stressing storm occurred in 1969. An event of approximately this magnitude occurred in 1985, before the start of logging in the North Fork. The highest 1-day precipitation total during the period of the study is 12.75 cm (5.02 in.) and occurred in water year 1998, approximately 7 years after completion of logging in the North Fork. This is only slightly less than a 10-year event. Therefore, it appears that only one 1-day duration storm total approached the 10-year recurrence interval during the 36-year study record. Five-year return period rainfall events occurred in all phases of the study except the period immediately after logging in the South Fork, but were most frequent before road construction or logging occurred in either watershed.

The 10-day precipitation totals for the South Fork Caspar Creek station tell a similar story (*fig. 5)*. Goodridge (1997) reported the 5-year return interval for this duration as 28.42 cm (11.19 in.). Storms in hydrologic years 1965, 1966, 1969, 1988, 1995, and 1998 exceeded this amount. The 1995 10-day total of 33.25 cm (13.09 in.) is second only to that of 1965, with 34.26 cm (13.49 in.), and both of these exceed the 10-day, 10-year return interval storm of 32.56 cm (12.82 in.) reported by Goodridge (1997). Therefore, these longer-duration storms are distributed more evenly throughout the 36-year rainfall record than the 1-day storm events, and have occurred in all phases of the study except the period immediately after logging in the South Fork.

Durgin and others (1989) state that Caine's (1980) data are most relevant to harvest area related landslides, but a similar threshold may apply to road-related failures. Rice and others (1985) reported that Caine's 1-day threshold has about a 4-year return period on the North Coast of California. In Caspar Creek, Goodridge (1997) reported the 5-year 1-day storm as 11.07 cm (4.36 in.), which is slightly above Caine's 24-hour threshold of 10.4 cm (4.1 in.). Therefore, to determine whether forest practices are sufficient to prevent landsliding above background rates with Caine's threshold, it is necessary for logging units and new roads to be tested by a 4-year return interval storm event after seasonal precipitation amounts that produce saturated mantle conditions. Events of this magnitude occurred after logging impacts in the South Fork in water years 1969, 1985, and 1998, and in the North Fork in 1998.

Caine's index includes durations up to 10 days. Using his equation, minimum landslide-triggering rainfall amounts for 3-, 5-, and 10-day durations are 20.12 cm (7.92 in.), 27.50 cm (10.83 in.), and 41.95 cm (16.52 in.), respectively. These 3- and 5-day totals are between 10- and 25-year return interval events, and the 10-day total is between a 50- and 100-year event for Caspar Creek, based on Goodridge's (1997) data. Highest recorded totals for these durations at Caspar Creek are all less than Caine's thresholds.

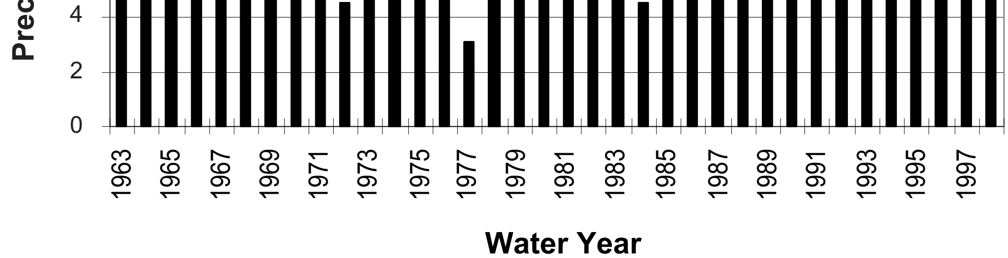

Figure 4—Annual maximum 1-day rainfall totals for the South Fork Caspar Creek 620 raingage.

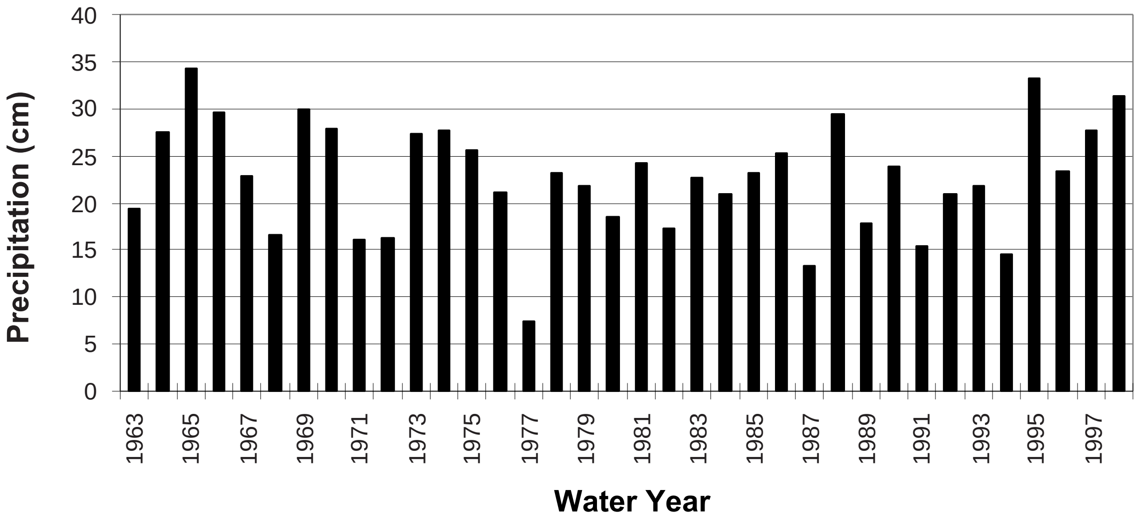

Figure 5—Annual maximum 10-day rainfall totals for the South Fork Caspar Creek 620 raingage.

Rainfall Records and Large Landslides in the North Fork

Eleven large (i.e., greater than or equal to 76.5 m^3 or 100 yd^3) landslides have been documented in the North Fork watershed during the life of the study[8] (*table 1*). Five features are located in unharvested areas (i.e., mature second-growth forest), and six are in clearcut blocks. Three of these landslides were much larger than the others, exceeding 1,000 m^3 (1,308 yd^3). The first of these features occurred during March 1974 and was a shallow debris slide of approximately 3,306 m^3 (4,324 yd^3) that directly entered the North Fork of Caspar Creek (Rice and others 1979). The second recorded large feature was a shallow debris slide that occurred during February 1986, near the old splash dam site. Both of these landslides occurred before the most recent logging and were not associated with the clearcut logging or new road construction. The largest landslide in the study period was a 3,606-m^3 (4,715-yd^3) debris flow that came from a steep hollow high on a hillslope in a clearcut unit in January 1995.

The rainfall amounts for storm periods associated with the landslides larger than 76 m^3 are displayed in *table 1*. The totals for 1-, 3-, 5-, and 10-day period preceding the event are generally similar for logged area and uncut area landslides. The mean precipitation totals for 1-, 3-, 5-, and 10-day durations are all below 5-year recurrence interval amounts. For individual storms, rainfall amounts for either 3-day or 10-day durations were at or over 2-year return intervals, whereas all but one of the 1-day totals were under this return frequency. This suggests that it is critical to have significant amounts of precipitation for long durations to generate large landslides in Caspar Creek.

On the basis of the largest landslides documented in the North Fork during the life of the study, it appears that Caine's (1980) 1-day threshold of 10.4 cm (4.1 in.) is less important for slide initiation than 3-, 5-, or 10-day totals below Caine's thresholds. None of the recorded large landslides in the North Fork occurred when 1-day precipitation totals exceeded Caine's threshold, and all of the landslides occurred with precipitation totals that were less than Caine's thresholds for 3-, 5-, and 10-day totals.

Therefore, we defined a potential minimum threshold for stressing storm events on the basis of the rainfall amounts associated with known landslides in the North Fork. We screened the entire rainfall data set from August 1962 to April 1998 and determined the storm events that met minimum standards of 4.88 cm (1.92 in.) of precipitation in 1 day and either 11.94 cm (4.70 in.) in 3 days or 20.09 cm (7.91 in.) in 10 days. These rainfall amounts were based on the values triggering the landslides shown in *table 1*. Additionally, an antecedent precipitation index (API) was calculated for all daily rainfall totals in the life of the study.[9] We found that 41 days and 34 unique storm sequences met the screening criteria (*fig. 6* and *appendix 1*). Three storm events occurred during the calibration period for both watersheds, five events took place after road construction in the South Fork, five events occurred during logging and recovery in the South Fork, six events occurred during the early 1980's before logging the North Fork, and 15 events occurred during the logging, road construction, and recovery period in the North Fork. These data suggest that storm events of magnitude similar to those known to have created landslides in the North Fork were reasonably well distributed over

[8] The Critical Sites Erosion Study (Rice and Lewis 1991) used 189 m^3ha^{-1} (100 yd^3ac^{-1}) as the definition of a large erosion site (either landslide or large gully). An inventory of erosion events greater than 7.6 m^3 (10 yd^3) was begun in 1986 for the North Fork.

[9] $API = Ppt_i + 0.9(API)_{i-1}$

all phases of the study. The greatest number of storm events occurred during hydrologic year 1998.

Based on this interpretation of the landslide data and the rainfall record for the basin, we conclude that: (1) Caine's thresholds did not predict the conditions leading to landslides at Caspar Creek, (2) the approximate magnitude of stressing storms that have triggered failures in the North Fork watershed are equal to or greater than 4.88 cm (1.92 in.) of precipitation in 1 day and either 11.94 cm (4.70 in.) in 3 days or 20.09 cm (7.91 in.) in 10 days, (3) sufficient numbers of stressing rainstorms or combinations of storm events have tested the practices implemented on the landscape in both the South and North Fork, and (4) an extreme precipitation event, such as a 50-year return period storm of any duration, has not occurred in either watershed. When such an event occurs, we will be able to further evaluate the impacts of logging and road construction in Caspar Creek.

Streamflow Discharge Records

Streamflow has been measured at both the South Fork and North Fork weirs since hydrologic year 1963.[10] Instantaneous annual peak discharges for the North Fork are displayed in *figure 7*. Peak discharges for 240 separate storm events were used for a partial duration flood series to plot a flood frequency analysis for both watersheds (R. Ziemer, USFS-PSW, written communication).[11] Flood events with return periods of 5 or more years are shown in *table 2*. These data illustrate that for discharge, flood events with 5-year or greater return frequency occurred before and after logging for the South Fork phase, as well as before and after logging in the North Fork phase. The primary difference, however, is that the largest flood events for the South Fork phase had return frequencies of approximately 20 years, whereas the return frequencies for the North Fork phase were between 5 and 10 years.

Estimates of historic flooding can also be made for the Caspar Creek watershed. The December 1955 discharge measured at the USGS's Noyo River gaging station[12] was similar in size to that measured for the January 1993 storm. It is likely that this was also the case at Caspar Creek, because the distribution of major flood peaks is similar in both basins from 1963 to the present. This flood event was likely to have been about a 10-year return interval event at Caspar Creek. According to regional records, the only other large flood that is likely to have taken place in the 1900's may have occurred in 1937 (Janda and others 1975).

One possible reason why the return frequencies for rainfall and runoff differ relates to antecedent moisture conditions. If a watershed is fairly dry before a large precipitation event, considerably less runoff will occur when compared to a fully

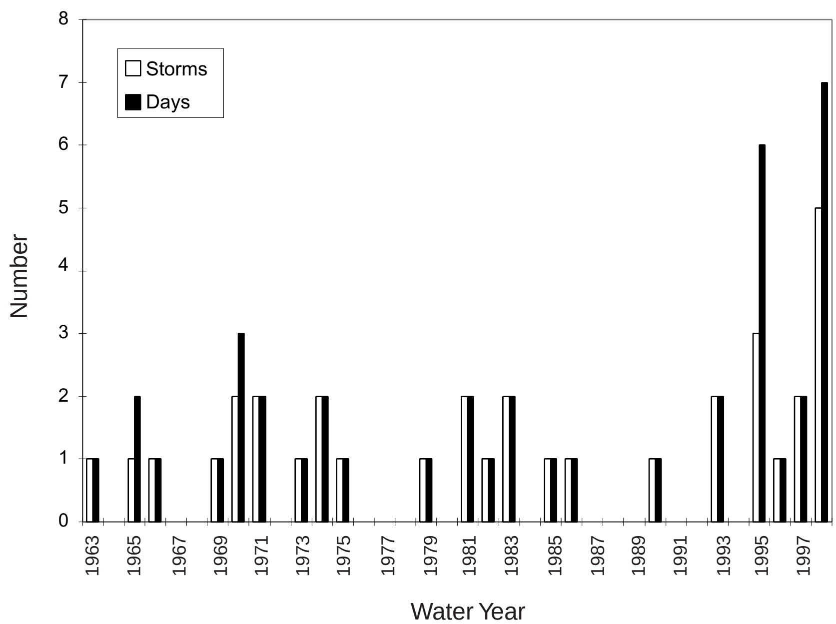

Figure 6—Number of days and unique storm sequences above the minimum threshold estimated to have the potential to produce landslides in the Caspar Creek watershed for the study period.

[10] Data for hydrologic year 1977 is missing, but this was the driest year of record and no large peaks occurred that winter.

[11] A storm for this analysis was defined as having a stage of at least 0.6 m (2 ft) at the South Fork weir (or a discharge of 0.7 m^3s^{-1} [24.5 ft^3s^{-1}]).

[12] USGS No. 11468500; records at this station began in 1952. The station is located approximately 6.4 km (4 mi) to the north of the Caspar Creek watershed.

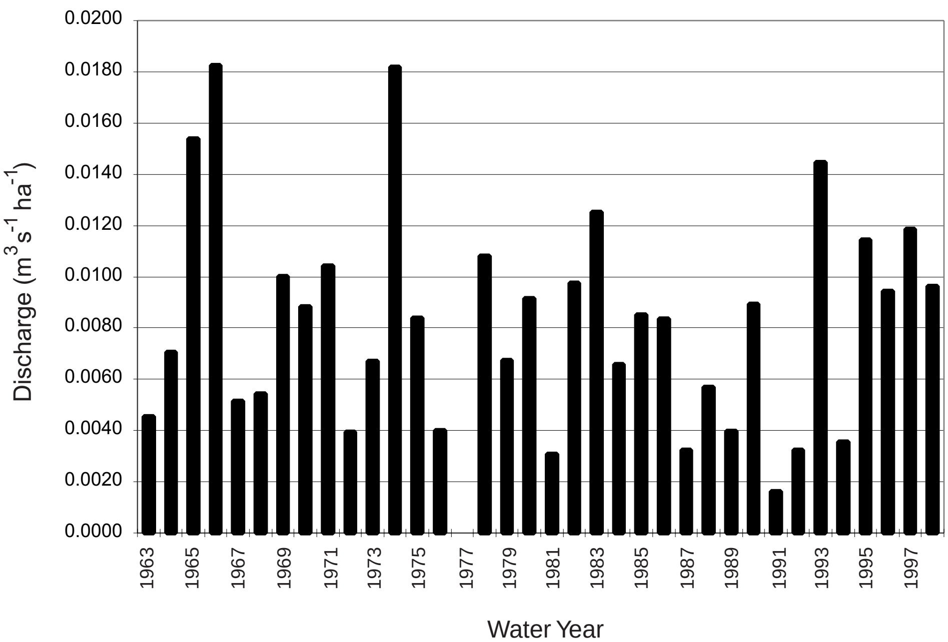

Figure 7—Maximum annual instantaneous peak discharges for the North Fork of Caspar Creek.

***Table 2**—North Fork Caspar Creek peak discharges with return intervals of 5 years or more.*

Date	Discharge (m^3s^{-1})	Discharge ($m^3s^{-1}ha^{-1}$)	Discharge (ft^3s^{-1})	Return Interval (yr)
01-04-1966	8.6	0.0182	304.9	21
01-16-1974	8.6	0.0182	304.3	21
03-29-1974	7.5	0.0159	263.8	12
12-21-1964	7.3	0.0154	257.5	10
01-20-1993	6.8	0.0145	241.6	8
12-21-1982	5.9	0.0125	209.6	6
12-09-1997	5.6	0.0118	198.0	5
03-14-1995	5.4	0.0115	191.5	5

saturated wet mantle condition. Antecedent wetness has been shown to be an important variable explaining runoff differences at Caspar Creek (Ziemer, these proceedings). Additionally, instantaneous peak discharges vary considerably depending on storm intensity and duration. Long-duration, low-intensity storm events can generate high rainfall amounts and high total storm flow volumes, but relatively low instantaneous peak discharges at Caspar Creek, when compared to shorter-duration storms with higher intensities.

Most sediment movement during an average hydrologic year occurs during a few, very large runoff events. Rice and others (1979) reported that about 80 percent of suspended sediment measured during the South Fork phase was transported by flows exceeding 1.13 $m^3 s^{-1}$ (40 $ft^3 s^{-1}$). Discharges of this magnitude occur about one percent of the time as shown by flow duration curves developed for both the North and South Forks of Caspar Creek.

Hillslope Erosion and Sediment Delivery Data

Hillslope erosion was measured for both the North and South Fork phases of the study. In the South Fork, estimates were obtained from seven plots distributed throughout the watershed. Plots were rectangular, approximately 200 m (656 ft) wide and 200 m to 320 m (1,050 ft) long. Gullies greater than 0.09 m^2 (1 ft^2) in cross section and mass movements displacing more than 0.76 m^3 (1 yd^3) were measured. Rice and others (1979) concluded that logging resulted in 81.1 $m^3 ha^{-1}$ (42.9 $yd^3 ac^{-1}$) of hillslope erosion above background rates. Only 3 percent of the total erosion was rill erosion; the remainder occurred as landslides or large gullies (Rice and others 1979). Sheet erosion was not included in the erosion estimate.

Rice (1996) also completed a sediment delivery study in the North Fork of Caspar Creek. Comparable types of hillslope erosion measurements were made, but the sampling scheme differed considerably, with measurements made on smaller randomly located plots. Circular 0.08-ha (0.20-ac) plots were installed on harvested or forested areas, and road plots consisted of 1.5-m (5-ft) segments of road prism normal to the road centerline (plus erosion to the nearest drainage structure). Rice (1996) concluded that the average hillslope erosion rate above background levels for the North Fork was 45.5 $m^3 ha^{-1}$ (24.1 $yd^3 ac^{-1}$), or roughly half that measured in the earlier South Fork study. We updated Rice's (1996) estimate with data through water year 1998 and revised Rice's earlier estimate to 47.6 $m^3 ha^{-1}$ (25.2 $yd^3 ac^{-1}$).

These erosion rates are generally similar to those reported earlier in the literature. For example, Dodge and others (1976) found hillslope erosion rates on California's North Coast of 106, 77, 195, and 346 $m^3 ha^{-1}$ (56, 41, 103, and 183 $yd^3 ac^{-1}$) on slopes of 0-30, 31-50, 51-70, and > 70 percent, respectively, from timber harvesting conducted before the implementation of the modern forest practice rules. The Critical Sites Erosion Study (Rice and Lewis 1991) compared hillslope erosion on 0.81-ha (2-ac) sites having large erosion events greater than 189 $m^3 ha^{-1}$ (100 $yd^3 ac^{-1}$) to randomly selected control sites and found an average of 19.1 $m^3 ha^{-1}$ (10.1 $yd^3 ac^{-1}$) for roads and harvest areas with logging that was completed under the modern Forest Practice Rules (1978-1979). In the North Fork of Caspar Creek, using the landslides in the harvested units listed in *table 1*, we found that the comparable amount is 18.8 $m^3 ha^{-1}$ (10.0 $yd^3 ac^{-1}$).

Rice and others (1979) reported that 22.4 percent of the measured hillslope erosion was delivered as sediment at the South Fork weir during the South Fork phase of the study. In contrast, for the North Fork logging, Rice (1996) calculated a sediment delivery of 11.3 percent at the North Fork weir. Therefore, Rice (1996) concluded that the North Fork logging resulted in approximately half as much erosion and a sediment delivery ratio that was similarly about half of the estimate for the South Fork logging. This indicates that the volume of sediment delivered to the stream channel in the North Fork was approximately one-quarter of that delivered to the South Fork.

Sediment Yields

Sediment sampling at Caspar Creek has been accomplished with several different techniques, reflecting changing technology and attempts to improve data quality. For most of the initial South Fork phase, suspended sediment yield was estimated with rising stage samplers mounted on the weirs. These devices, used from 1962 to 1975, are mounted at a specified stage and collect a sample only when the streamflow is rising. Some measurements were made with DH-48 depth-integrated hand-held samplers, but the majority of the data was from the mounted bottles. In 1975, PS-69 automatic pumping samplers were installed at both the North and South Forks. During spring 1976, frequency-controlling devices were added to these pumping stations, which provided for more intensive sampling during higher flows. This, however, was at the very end of the South Fork phase and, for all practical purposes, did not heavily influence the study results. Suspended sediment yields collected during the South Fork phase were generated from sediment rating curves. Thomas (1990) reported that rating curve estimates of sediment yields are biased and depend systematically on sampling protocols.

Sediment measurement methods were substantially improved for the entire North Fork phase (1985-1995), which used SALT (Selection At List Time) sampling at both the North and South Fork weirs and also at 13 gaging stations located above the North Fork weir. This newer suspended sediment sampling technique yields unbiased estimates of sediment discharge. At Caspar Creek, the probability of taking a sample is based on stage height. This provides unbiased estimates of total suspended sediment yield while causing more sampling to occur at higher flows (Thomas 1985).

Bedload transport has been estimated with annual surveys of weir pond sedimentation. Sediment samples analyzed to determine the percentage of particles ≥ 2 mm have been used to estimate the percentage of the material that settled behind the weirs and that can be considered bedload (material ≥ 1.4 mm), using a correction factor to account for the percentage of material from 1.4 mm to 2 mm (Napolitano 1996). Napolitano (1996) found that approximately 85 percent of the sediment produced at Caspar Creek can be considered suspended sediment. Material surveyed in the weir ponds averages about 35 percent of the total sediment load, which indicates that about 20 percent of the fine sediment (< 1.4 mm) settles out in the ponds (Napolitano 1996). Lewis (these proceedings) has completed a similar data analysis and concluded that approximately 40 percent of the suspended sediment yield settles out in the weir ponds. With this analysis, approximately 30 percent of the total sediment yield can be considered bedload and 70 percent suspended sediment. This latter estimate is likely to more accurate, since Napolitano (1996) used annual loads based on the fixed-stage samplers.

Annual sediment totals for suspended sediment and bedload measured at the weir ponds and estimated from samples taken at the weir outlets are shown in *figure 8* for both the North and South Forks of Caspar Creek. Mean sediment yields in the North and South Forks from 1963 to 1995 are 1,895 and 2,018 $kg\ ha^{-1} yr^{-1}$ (465 and 495 $yd^3 mi^{-2} yr^{-1}$), respectively.[13]

Lewis (these proceedings) has used a log-log model to compare suspended sediment and total sediment yields for both the North and South Forks for the before and after road construction and logging periods. For the South Fork, an increase of approximately 212 percent

[13] Note that the sediment estimates made from 1963 to 1975 were made without SALT sampling and are likely to overestimate true suspended sediment loads, since rating curves were generally based on rising limb sediment collection.

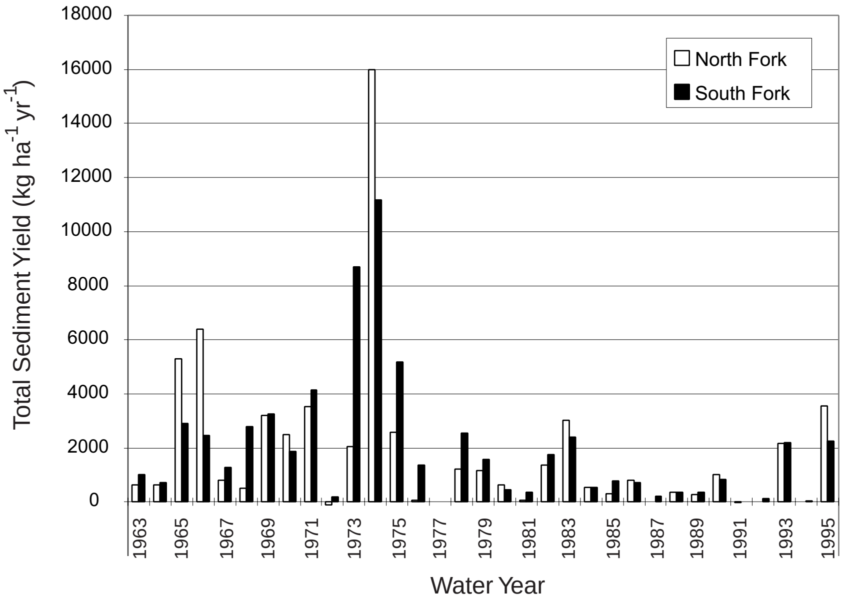

Figure 8—Total annual sediment yield for both the North and South Forks of Caspar Creek.

(2,510 kg ha^{-1} yr^{1}) over background levels has been calculated for suspended sediment loads for the first 6 years after the start of logging.[14] Total sediment load was 184 percent (2,763 kg ha^{-1} yr^{-1}) above expected levels. Lewis (these proceedings) has determined that because of the data collection techniques used for the South Fork phase, the original sediment volume estimate may be too high by a factor of 2 or 3 for suspended sediment measured in outflow from the weir, which would be approximately 1.7 to 2.4 times for the total volume of suspended sediment and bedload material. Sediment loads appeared to have returned to pre-logging levels in the 6th or 7th winter after the completion of logging (Thomas 1990). However, on the basis of widespread failures noted along South Fork roads during winter 1998, this conclusion may have been premature.

Lewis similarly calculated suspended sediment and total sediment yield increases above expected levels at the North Fork weir for logging and road construction during hydrologic years 1990 through 1995 and found no significant increase over background levels for either parameter. Lewis (these proceedings) then used a more sensitive analysis based on sediment measured at the North Fork weir for individual storm events compared to uncut tributaries upstream in the North Fork. This analysis indicated a significant, estimated increase of 89 percent (188 kg ha^{-1} yr^{-1}) in suspended sediment over that predicted for undisturbed conditions.

Lewis (personal communication) has concluded that percent increase is a more accurate representation of increased sediment yields than sediment volume, because of the problems associated with measurement techniques used in the South Fork phase of the study. Comparing the 89 percent increase for the North Fork with the 212 percent South Fork increase suggests that suspended sediment yields were 2.4 times greater following logging in the South Fork. This estimate is based on the unadjusted sediment measured in the North Fork during 1974, when a large landslide (see *table 1*) in the uncut basin produced more sediment than any of the post-disturbance years documented during the life of the study through water year 1995 (*fig. 8*).

Conclusions

After timber harvesting activities, both the North Fork and the South Fork of Caspar Creek were subjected to stressing storms that triggered landslide activity. The data indicate that high 3-day or 10-day precipitation totals in combination with moderately high 1-day amounts have been more important for initiating shallow landsliding than very high 1-day totals alone. Storm events with the potential to produce landslides, based on the minimum threshold we defined in this paper, have occurred at least 13 times since the completion of logging and road construction in the North Fork (*appendix 1*). Therefore, we conclude that stressing storm events with return intervals of up to 10 years have adequately tested the forestry practices implemented in the North Fork.

The frequency of landslides greater than 76 m^3 (100 yd^3) to date has not been substantially different between the clearcut units and the uncut control subwatersheds for the North Fork of Caspar Creek. Additionally, the volume of sediment discharged by landslides from the uncut and cut units to date has been

[14] Data for hydrologic year 1977 is missing.

approximately the same: 21 $m^3 ha^{-1}$ (11 $yd^3 ac^{-1}$) from the uncut units and 19 $m^3 ha^{-1}$ (10 $yd^3 ac^{-1}$) from the harvested areas. Long-term monitoring will inform us if these trends continue with much larger stressing storm events. For perspective on the magnitude of past events, the largest landslide mapped in the North Fork watershed, the debris flow that dammed the creek for thousands of years, was on the order of 1,000,000 to 5,000,000 m^3. This is more than three orders of magnitude larger than the largest landslide observed during the study.

Road, landing, and skid trail design, placement, and construction are the dominant controls on the number and locations of shallow landslides. As observed in the monitored part of the South Fork of Caspar Creek, land use practices, specifically tractor operations on steep slopes, can obscure and overwhelm intrinsic properties for shallow landslides. In contrast, in the watershed of the North Fork of Caspar Creek, where cable yarding was conducted on steeper slopes, the rate of landsliding is substantially lower, and there does not appear to be a significant increase in post-logging landsliding. Roads, landings, and skid trails that were constructed prior to the implementation of the Forest Practice Act have resulted in a legacy that continues to affect the watershed of the South Fork of Caspar Creek 30 years after operations began.

Results similar to those reported in this paper have been found elsewhere in northwestern California. Rice (1998) compared logging-related road erosion on industrial timberland in the middle portion of the Redwood Creek watershed before the implementation of the modern Forest Practice Rules with erosion rates associated with roads used in the 1990's. The estimated erosion rate under the modern Forest Practice Rules was about one-tenth of that estimated for an adjacent tributary of Redwood Creek as a result of timber operations utilized before 1976 (Best and others 1995).

Recommendations for Forest Managers Based on Caspar Creek Results

The lessons that have been learned at the Caspar Creek watershed may be applied to many North Coast watersheds. Numerous issues on Timber Harvesting Plans, such as hillslope erosion rates, sediment yields, and changes in peak flows, have already been addressed with data generated from this study.[15] The value of having research-level, long-term monitoring data from various types of logging operations is significant in today's arena of listed species, Total Maximum Daily Load (TMDL) allocations, and Habitat Conservation Plans. The data provided by this project has, and will continue to be, used to estimate the true impacts of modern logging operations. Long-term monitoring should continue at Caspar Creek as further operations are completed in the basin in the future decades, and as the watersheds are subjected to large, long-return frequency storms and floods.

Clearly, forest managers in other North Coast watersheds should take home the message that old roads built with practices prevalent in the 1950's, 1960's, and early to mid-1970's are still significant sources of erosion. Dr. William Weaver, Pacific Watershed Associates, has often referred to perched fill and poor watercourse crossings associated with old roads as "loaded guns" waiting to fail with strong stressing storm events. It is imperative that forest managers develop long-term *road management plans* that inventory these source areas and quickly reduce their numbers with an organized schedule based on watershed sensitivity and vulnerability of downstream beneficial uses.

One of the most important components of a comprehensive road management plan is the determination of which high-risk roads should be properly abandoned. Under the current California Forest Practice Rules, this means leaving a logging road in a condition that provides for long-term functioning of erosion controls with little or no continuing maintenance. Proper road abandonment usually involves removing watercourse crossing fills, removing unstable road and landing fills, and providing for erosion-resistant drainage (Weaver and Hagans 1994). During summer 1998, most of the old road system built in the South Fork watershed in 1967 will be properly abandoned. Proper abandonment of old roads and removal of high-risk sites on roads which will be part of the permanent transportation network are examples of the level of commitment from resource managers that is needed to substantially reduce the impacts from practices that were implemented on the landscape before the mid-1970's. Continued monitoring of the effects of various road abandonment techniques in Caspar Creek will aid forest managers elsewhere in developing proper abandonment practices.

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***Appendix 1**—Dates and amounts of precipitation (cm) above the defined threshold based on actual landslides and associated Antecedent Precipitation Index (API) levels for the entire study period . Abbreviations under Study Period are South Fork (SF), North Fork (NF), and Subwatershed YZ (YZ).*

Date	Study Period	Hydrologic Yr	1-day	3-day	10-day	API
10-11-62	Calibration	1963	7.49	12.80	14.38	13.36
12-21-64	Calibration	1965	11.28	17.78	19.58	20.09
12-22-64	Calibration	1965	5.41	18.06	24.99	23.50
01-04-66	Calibration	1966	11.89	18.39	26.97	23.29
01-11-69	Post SF road	1969	12.42	12.65	12.78	16.38
12-12-69	Post SF road	1970	9.14	13.97	15.88	14.76
01-23-70	Post SF road	1970	6.10	13.72	27.79	24.36
01-26-70	Post SF road	1970	5.08	6.40	27.84	23.93
12-03-70	Post SF road	1971	7.29	11.68	22.94	17.70
01-16-71	Post SF road	1971	5.84	13.77	20.90	19.02
01-11-73	Logging SF	1973	8.97	14.66	17.60	17.40
01-15-74	Post SF log, road	1974	6.10	15.11	16.43	17.45
03-31-74	Post SF log, road	1974	6.17	11.79	20.09	17.30
03-21-75	Post SF log, road	1975	7.11	7.37	20.75	18.64
02-13-79	Post SF log, road	1979	5.72	12.22	13.23	13.08
12-03-80	Pre NF log, road	1981	7.52	13.23	14.63	14.35
01-27-81	Pre NF log, road	1981	6.73	8.64	20.98	16.18
02-15-82	Pre NF log, road	1982	7.62	14.48	14.48	14.61
12-21-82	Pre NF log, road	1983	8.00	10.92	20.14	17.93
01-26-83	Pre NF log, road	1983	8.08	12.14	23.44	18.47
11-11-84	Pre NF log, road	1985	10.62	14.81	18.77	18.97
02-16-86	Log, road NF (YZ)	1986	4.95	11.94	16.13	16.92
05-27-90	Log, road main NF	1990	4.88	12.40	23.77	17.81
12-31-92	Post NF log, road	1993	8.28	16.74	12.65	17.42
01-20-93	Post NF log, road	1993	6.05	10.90	20.98	20.17
01-08-95	Post NF log, road	1995	7.54	12.42	17.86	16.48
01-09-95	Post NF log, road	1995	5.97	15.62	23.67	20.80
01-13-95	Post NF log, road	1995	6.40	11.35	33.25	25.17
03-13-95	Post NF log, road	1995	5.44	8.10	20.47	16.61
03-14-95	Post NF log, road	1995	5.11	11.38	25.58	20.07
03-20-95	Post NF log, road	1995	5.23	7.54	21.11	18.08
01-24-96	Post NF log, road	1996	5.13	7.42	23.37	16.56
12-09-96	Post NF log, road	1997	8.31	12.52	25.12	20.60
12-31-96	Post NF log, road	1997	8.59	17.60	23.90	23.55
11-26-97	Post NF log, road	1998	12.75	13.54	19.02	19.15
01-12-98	Post NF log, road	1998	5.00	11.94	17.48	15.65
01-14-98	Post NF log, road	1998	5.00	11.56	20.17	19.08
01-18-98	Post NF log, road	1998	5.00	9.68	30.43	22.96
01-26-98	Post NF log, road	1998	7.67	11.15	21.26	22.40
02-21-98	Post NF log, road	1998	4.98	11.73	22.83	22.89
03-22-98	Post NF log, road	1998	5.08	12.57	12.62	15.16

Low thermal tolerances of stream amphibians in the Pacific Northwest: Implications for riparian and forest management

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Abstract. Temperature has a profound effect on survival and ecology of amphibians. In the Pacific Northwest, timber harvest is known to increase peak stream temperatures to 24°C or higher, which has potential to negatively impact cold-water stream amphibians. I determined the Critical Thermal Maxima (CT_{max}) for two salamanders that are endemic to the Pacific Northwest. *Rhyacotriton variegatus* larvae acclimated at 10°C had mean CT_{max} of 26.7 ± 0.7 SD°C and adults acclimated at 11°C had mean CT_{max} of 27.9 ± 1.1°C. These were among the lowest known values for any amphibian. Values were significantly higher for larval *Dicamptodon tenebrosus* acclimated at 14°C ($\bar{x} = 29.1 \pm 0.2$°C). Although the smallest *R. variegatus* had some of the lowest values, size of larvae and adults did not influence CT_{max} in this species. Current forest practices retain riparian buffers along larger fish-bearing streams; however, such buffers along smaller headwaters and non-fish bearing streams may provide favorable habitat conditions for coldwater-associated species in the Pacific Northwest. The current study lends further evidence to the need for protection of Northwest stream amphibians from environmental perturbations. Forest guidelines that include riparian buffer zones and configurations of upland stands should be developed, while monitoring amphibian responses to determine their success.

Key words: Critical Thermal Maximum; *Dicamptodon*; headwater; *Rhyacotriton*; salamander; temperature.

Introduction

The Pacific Northwest of North America is home to three endemic families of amphibian that are restricted to cool-water streams: tailed frogs (Ascaphidae); torrent salamanders (Rhyacotritonidae); and Pacific giant salamanders (Dicamptodontidae). *Ascaphus* are adapted to fast, rocky streams (e.g., the larvae have a large suctorial mouth to attach to rocks) (Dupuis and Steventon, 1999; Jones et al., 2005). *Rhyacotriton* are small (few > 100 mm total length), "brook-type" salamanders that live in seeps, splash zones, waterfalls and headwaters (Nussbaum and Tait, 1977). *Dicamptodon* frequent creeks and larger streams with some in pools, and may reach large size of up to 350 mm total length (Nussbaum et al., 1983; Jones and Welsh, 2005). *Rhyacotriton* and *Ascaphus* reach their highest numbers in older forests (Welsh, 1990; Adams and Bury, 2002) while *Dicamptodon* may also occur in more open or logged habitats (Bury and Corn, 1988; Wilkins and Peterson, 2000).

Of all the physical parameters in the aquatic environment, temperature is perhaps the most dramatic in its effect on the physiology, ecology, and behavior of anuran larvae (Ultsch et al., 1999). One of the most common responses used to quantify temperature tolerance is the Critical Thermal Maximum (CT_{max}). This index represents the temperature at which an animal loses its righting ability and would perish quickly if not removed to cooler conditions (see Hutchinson and Dupré, 1992). The CT_{max} for most anuran larvae is between 38°C and 42°C (Ultsch et al., 1999). The thermal tolerances of *Ascaphus* and *Rhyacotriton* are among the lowest known for amphibians, but data are limited in geographic scope or available for few of the species (table 1). For example, Brattstrom (1963) reported *Rhyacotriton* had a low CT_{max} ($\bar{x} = 28.3$°C) but there was no mention if these were larvae or adults. He also did not state the collection site, which is important as there are now four species in the group (Good and Wake, 1992). Welsh and Lind (1996) briefly mentioned that CT_{max} tests for *R. variegatus* indicated thermal stress at 17.2°C, but they provided no other test information. I was not able to find published CT_{max} data for *Dicamptodon*.

Stebbins (1951) found field body temperatures of California *Rhyacotriton* (=*R. variegatus*) to be 5.9-9.6°C ($N = 25$). Field data of *R. variegatus* in northern California suggest that they only occur in streams where stream tempera-

Table 1. Critical Thermal Maximum (CT_{max}) of stream amphibians in the Pacific Northwest.

Species stage	N	Acclimation		CT_{max} (°C)	Source
		°C	$\bar{x}$	Range	
Ascaphus montanus					
Larvae	24	10	ca. 29		Metter (1966)
Adults	12	10	ca. 28		”
	8	0	27.6		Claussen (1973)
	8	10	29.6		”
	6	20	ca. 29		”
Ascaphus truei					
Larvae	8	5	ca. 29.6	28.9-30.1	de Vlaming and Bury (1970)
Rhyacotriton sp.					
Larvae	8	13-14	ca. 28.3	27.8-29.0	Brattstrom (1963)
Rhyacotriton variegatus					
Larvae	7	10	26.7	25.6-27.4	This study
Adults	8	11	27.9	26.3-29.3	”
Dicamptodon tenebrosus					
Larvae	12	11	29.1	28.7-29.3	”

tures are $\leqslant$ 15.0°C in summer (Diller and Wallace, 1996; Welsh et al., 2001; Welsh and Karraker, 2005). Brattstrom (1944) reported body temperatures of larval *Dicamptodon* (species unknown) from 12.0-16.2°C ($N = 6$), and summarized field body temperatures: *Rhyacotriton*, $\bar{x} = 8.7$°C ($N = 28$); and *Dicamptodon*, $\bar{x} = 13.1$°C ($N = 12$).

Objectives of this study are to provide estimates of CT_{max} for single populations of larval and adult *R. variegatus* and larval *D. tenebrosus* from Oregon. I also summarize what is known of the thermal ecology of these two species and *Ascaphus* in relation to forestry practices that can elevate stream temperatures. Elevation of stream temperatures may pose a particular threat to *R. variegatus*, which was petitioned for listing as a Federal threatened species (U.S. Fish and Wildlife Service, 1995). Although not found to qualify for listing at that time, concerns remain about losses of *Rhyacotriton* and *Ascaphus* populations due to management practices such as timber harvest, road construction and prescribed fires.

Materials and Methods

Study animals and acclimation periods

I obtained *R. variegatus* at Parker Creek (elevation = ca. 700 m) on Mary's Peak, ca. 25 km WSW of Corvallis, Benton Co., Oregon. Measurements of snout-vent length (SVL; mm) and mass (g) of seven tested larvae were: 22 mm (0.30 g), 23 (0.38), 26 (0.55), 27 (0.54), 27 (0.61), 28 (0.57) and 32 (0.91). Adults ($N = 8$) were: male — 40 mm (1.97 g); females — 32 mm (1.06 g), 33 (1.10), 34 (1.25), 37 (1.94), 38 (1.79), 42 (1.97) and 43 (0.98). Larvae were acclimated in the laboratory at 10°C for 2-3 weeks; adults were held at 8-9°C in a refrigerator and then one week at 11°C. These slight variations in average acclimation temperatures were due to changes in water inflow to the laboratory (from wells) and seasonal shifts in air temperatues.

Samples of *D. tenebrosus* larvae ($N = 12$) were from Withrow Creek (elevation ca. 790 m) near Glide, Douglas Co., Oregon, in the foothills of the Cascade Mountains. Lengths and masses were: 30 mm (1.82 g), 37 (2.92), 38 (3.15), 39 (2.95), 43 (3.52), 43(3.90), 45 (3.77), 45 (4.25 g), 47 (4.58), 47 (5.72), 48 (5.89) and 48 (6.53). They were acclimated at 14°C (ambient water temperature) for 3 weeks.

Water for acclimation and testing was from a well near the Willamette River at Corvallis, Oregon. Photoperiod was 14 h light: 10 h dark and light intensity was $\sim$150-200 lumens m^{-2} at the water surface. Animals were not fed during acclimation periods (2-3 weeks) because of the short time in laboratory and lack of access to natural live food sources.

Critical Thermal Maxima (CT_{max}*)*

I defined CT_{max} of larvae as the temperature at which they exhibit uncontrolled, jerky swimming behavior or if they rolled over (belly-up) on the bottom of the beaker. Animals were observed constantly and, once the response was clear, I poured cool water (ca. 20°C) into the beaker. This resulted in recovery with no apparent ill effects 1 week after experiments.

To prevent thermal shock, each individual was held at 20°C (1000 mL of well water in a 1-L beaker) for 10 min before each test. For larvae, water was heated with a 50-W glass-aquarium heater at an average rate of 0.6°C min^{-1} from 20°C to the CT_{max}. Water was aerated and mixed to eliminate supersaturated gas conditions during heating. I recorded water temperature at 1-min intervals during exposure to determine the CT_{max} value. I ran control exposures under the same time and test conditions but with no heater.

I tested individual adults in an aerated 1-L beaker (with 900 mL well water), but with a screen floor (5-mm mesh; stainless steel) suspended 10 mm above the water level so that animals were in an air space above the water. A plastic Petri dish covered the beaker but a cut area allowed space for the heater, two thermometers, and a glass aeration line to enter through the top. Water was heated (and the air space above the water and screen floor) with a 50-W aquarium heater for about 20 min exposure. The air coming into the beaker through the aeration line (ca. 3 bubbles s^{-1}) was heated in a water bath to 30-35°C. Air temperature was measured at the screen floor and at the Petri dish lid, and the two values were averaged. Control exposures were conducted with the adults in the same conditions except air and water were not heated. The adults were quickly removed from the beaker when they displayed erratic behavior (e.g., twisting of body) or inability to right themselves. Animals were quickly removed and placed in cooler water (ca. 20°C) for several minutes, and then returned to holding tanks. No animals died during testing or in the control groups.

Statistical analyses

I employed a Likelihood Ratio Test (LRT) to compare the distribution of each group's scores for mass and CT_{max} (Ramsey and Schafer, 1996). The trend lines fit to each group produced a slope and y-intercept. The likelihood of parameters in a general regression model (y-intercept and slope of each group) and a reduced model (y-intercept of each group and just the larval *Rhyacotriton* data slope) were evaluated at their maximum. A P-value of <0.05 indicates a significant difference between the groups. I express CT_{max} as mean $\pm$ Standard Deviation.

Results

All tested animals responded similarly after ca. 15-20 min exposure to heating. Larval *R. variegatus* reached CT_{max} at 25.6-27.4°C ($\bar{x} = 26.7 \pm 0.7$°C; fig. 1). The two control tests ($N = 1$ each) showed no effects (animals remained motionless or moved little). For adult *R. variegatus*, CT_{max} values were 26.3-29.3°C ($\bar{x} =$

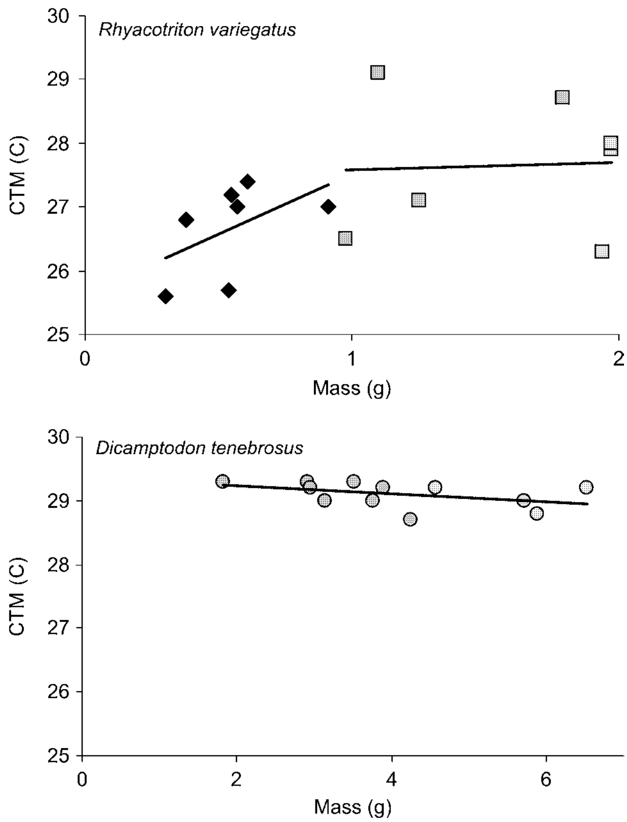

Figure 1. Critical Thermal Maxima (CT_{max}; °C) as a function of body mass (g). Larval southern torrent salamanders (*Rhyacotriton variegatus*) are represented by black diamonds, and adults by grey squares. Larval coastal giant salamanders (*Dicamptodon tenebrosus*) are grey circles in the lower graph.

27.9 ± 1.1°C). Three control tests ($N = 1$ each) showed no effects (animals were quiet or seldom moved).

Larval *D. tenebrosus* had statistically higher CT_{max} ($x = 29.1 \pm 0.2$°C) than either group (larvae or adults) or combined data (larvae + adults) for *R. variegatus* (fig. 2). No effects occurred in two controls ($N = 1$ each).

Size of *Rhyacotriton* did not influence the CT_{max} value (fig. 1). Larval *R. variegatus* were smaller in SVL ($\bar{x} = 26.43 \pm 2.43$ mm) and mass ($\bar{x} = 0.55 \pm 0.44$ g) than adult *R. variegatus* SVL ($\bar{x} = 37.38 \pm 4.14$ mm) and mass ($\bar{x} = 1.63 \pm 0.43$ g). Further, both were less than *Dicamptodon* in SVL ($\bar{x} = 42.5 \pm 5.50$ mm) and mass ($\bar{x} = 4.08 \pm 1.39$ g). Although the lowest CT_{max} values were the smallest *R. variegatus*, size of animals did not influence the CT_{max} for adult *R. variegatus* or larval *D. tenebrosus* (fig. 1). A comparison of a general regression model and a reduced model showed no significant difference between the slopes of the three groups (Likelihood Ratio Test; $P = 0.24$).

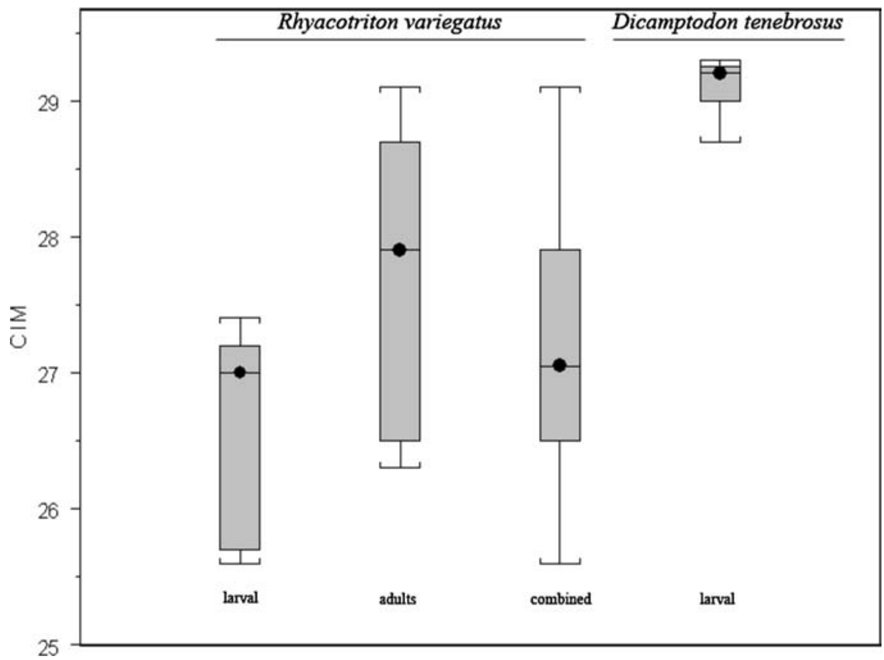

Figure 2. Critical Thermal Maximum (CT_{max}) ranges recorded for larval, adult and combined data for the southern torrent salamander (*Rhyacotriton variegatus*) and larval coastal giant salamander (*Dicamptodon tenebrosus).* Data medians for each group are represented by the black circles with bar inside each box. Boxes represent the 95% confidence interval of the data, and the associated whiskers represent the maximum and minimum data for each.

Discussion

The CT_{max} of Oregon *R. variegatus* are lower for larvae ($\bar{x} = 26.7$°C) and adults ($\bar{x} = 27.9$°C) than reported for any other salamander, and these values are among the lowest of any amphibian species (Brattstrom, 1963; Ultsch et al., 1999). The only lower records (CT_{max}) were for two high altitude species of Australian frogs acclimated at 5°C (Brattstrom, 1970): *Philoria frosti*, 27.1°C; and *P. sphagnicola*, 25.8°C. Thermal tolerances of *D. tenebrosus* were higher than for *R. variegatus* (table 1). Although *D. tenebrosus* is relatively widespread in distribution and habitat, it still appears sensitive to elevated water temperatures (e.g., CT_{max} near 30°C).

The samples were progressively larger (in SVL), heavier (in g) and, apparently, higher CT_{max} from larval to adult *Rhyacotriton*, and then to larval *Dicamptodon*. However, a comparison of mass and CT_{max} for all data (no separation by species or stage) showed no statistical significance. Rather, *R. variegatus* (CT_{max} of larvae and adults combined) were different from *D. tenebrosus* (fig. 2). Within *Rhyacotriton*, most larvae had lower CT_{max} than adults but the differences were statistically insignificant. Also, greater range in mean values for adult *R. variegatus* compared to larvae may reflect the varied area within the air space for adult testing, which allowed adults to climb on the screen and up the sides of the beaker. Although Dupré and Hutchinson (1985) reported selection of higher mean temperatures during ontogenetic shifts in larval amphibians (three anurans, one salamander), Miller and Packard (1977) reported that most evidence suggests that there was no correlation between heat resistance and body size in amphibians. I found differences between the two species of stream salamanders, but further research is required to tease out details of variation related to body size.

Alternatively, the larval *D. tenebrosus* may have had higher values because they were held at 14°C compared to *R. variegatus* larvae at 10°C and adults at 11°C. Acclimation temperatures influence CT_{max} in other amphibians, but changes are modest (range 0.7-1.8°C difference) for a 10°C change in acclimation temperature (Rome et al., 1992). Hutchison et al. (1973) found no statistical differences in CT_{max} of the stream-dwelling *Cryptobranchus alleganiensis* acclimated at 5 or 15°C but differences occurred between groups acclimated at 15°C (CT_{max} $\bar{x} = 32.99 \pm 0.40$°C) and 25°C ($CT_{max} \bar{x} = 36.57 \pm 0.46$°C), which is a change of 0.36°C in CT_{max} for degree^{-1} acclimation increase. This type of response to acclimation temperature may account for the slight difference (1.2°C in CT_{max}) between the acclimation difference (3°C) between adult *R. variegatus* and larval *D. tenebrosus*.

In another cold-adapted species, Claussen (1973) reported that adult *Ascaphus* acclimated near zero temperatures had lower CT_{max} ($\bar{x}$ = 27.6°C) than frogs at either 10°C or 20°C acclimation (CT_{max} $\bar{x}$ = ca. 29.6°C). This was an average increase of 0.2°C in CT_{max} degree^{-1} acclimation temperature from near zero to 10°C but no difference above 10°C. There are no equivalent data for any species of *Dicamptodon* or *Rhyacotriton*. Even accounting for possible influence from slightly higher acclimation temperatures, *D. tenebrosus* had a higher thermal tolerance than *R. variegatus*.

Management implications

Rhyacotriton experience the largest losses of any stream amphibian in the Pacific Northwest following clear-cut logging (Corn and Bury, 1989; Welsh and Karraker, 2005). One explanation may be absence or reduction of forest canopy after logging that result in increased stream temperatures, which may be stressful or lethal to *Rhyacotriton*. Effects appear to be ameliorated in areas with a coastal marine climate (see Diller and Wallace, 1996; Russell et al., 2004) or in small streams that have cool groundwater flows (Steele et al., 2003). In Oregon, Everest et al. (1985) stated that small streams are more subject to temperature changes (i.e., increases) than large streams.

Rome et al. (1992) reported that there is little evidence that amphibians ever experience temperatures that approach CT_{max} in nature. For *R. variegatus*, stream temperatures that are lethal (CT_{max} = 26.7-27.6°C) may be uncommon or not experienced in forests or logged areas. Following logging or other perturbations, however, stream temperatures in summer may reach levels that are physiologically stressful to *Rhyacotriton* and *Ascaphus*. Adams and Frissell (2001) suggested that *A. montanus* may migrate to avoid warmer parts of streams. They may be able to escape in subterranean retreats or cold seeps/springs, if available. However, stream amphibians cannot move into adjacent forests in summer or the non-rainy season as temperatures are much higher in adjacent uplands than in flowing waters or riparian zones (Chen et al., 1999).

Although CT_{max} is a standardized test of thermal tolerance by a species, it is measured with rapid increase in temperature for a short time. Summer stream temperatures at harvest units or other areas of perturbation may reach 24°C but these are usually only for 1 or more hr day^{-1}. Thus, testing species of stream amphibians under more natural patterns with fluctuating conditions (e.g., peaks at 22-24°C for 1-2 hrs day^{-1} followed by cooling down periods to 15°C or less) would likely be more relevant to field situations.

Response to warm water for extended periods may be lethal. For example, another coldwater genus in the Pacific Northwest is *Ascaphus*. In constant water temperatures of 22°C, its larvae began to die after 24 hrs and 75% were dead after 48 hrs whereas all adults ($N = 12$) died between 18 and 30 hrs (Metter, 1966). A similar response is expected for *Rhyacotriton* because its CT_{max} is 1-3°C less than that reported for *Ascaphus*. However, such periods of extended warm water (e.g., 22°C for 12 hrs day^{-1}) are seldom encountered in the wild.

Both *Rhyacotriton* and *Ascaphus* face risk where there are elevated stream temperatures. In the Oregon Coast Range, one small stream in summer rose from 14° to 22°C at mid-day following clear-cutting of the drainage, with a peak in a pool at 30°C (Brown and Krygier, 1970). In the Oregon Cascades, Johnson and Jones (2000) reported maximum water temperatures of 23.9°C in two streams flowing through a clear-cut in a small watershed and in a stand with three small patch-cuts plus construction of logging roads. Both logged areas were burned post-harvest, which is a common forestry practice in the region. Streams in nearby mature forests did not have temperatures exceeding 19°C ($\bar{x} = 16.7$°C) in summer. Temperatures in streams in logged plots did not return to the pre-harvest levels until ca. 15 yr later, coinciding with return of the riparian zone and canopy closure.

Moreover, eggs of *A. truei* die in water > 18.5°C (Brown, 1975). *Ascaphus* deposits its eggs in mid-summer in the warmest part of the year and, thus, face stress immediately. *Rhyacotriton* appears to deposit eggs in spring and early summer, and it may take 200 d for hatching (Nussbaum and Tait, 1977; Nussbaum et al., 1983). Currently, we lack any data on the thermal tolerance of the eggs of *Rhyacotriton*.

Perturbations caused by natural (e.g., wildfires) and human (e.g., timber harvest) events in the Pacific Northwest may cause elevated stream temperatures to levels of 24°C or more in summer. These have potential to stress or harm cold-adapted species such as stream amphibians. Similar to salmonid fishes of the Pacific Northwest (Carline and Hachung, 2001), *Ascaphus* or *Rhyacotriton* rarely occur in streams that have water temperatures > 16°C (Welsh, 1990; Diller and Wallace, 1996; Welsh et al., 2001). In Oregon streams, Huff et al. (2005) reported that stream amphibians were consistently found in streams with low temperatures (averages): larval *Dicamptodon* (12.0-14.3°C) and *Ascaphus* (11.7-15.3°C). Some *Ascaphus* have been found in streams with water temperatures up to 21°C where groundwater seeps create cold pockets and spatially complex thermal environments (Adams and Frissell, 2001). Recently, Dunham et al. (2007) report *A. montanus* occurring in streams with a maximum daily peak in summer up to 26°C but most waters (54%) were cooler (<20°C). Sites in burned, reorganized stream beds had a high probability (>0.75) of exceeding 20°C whereas streams in unburned areas were low (<0.25). Although stream amphibians may occur in relatively warm waters for brief periods (e.g., >24°C for 1-2 h), the animals may be compromised (e.g., have reduced agility, feeding, and growth rates). However, these potential sublethal effects of thermal stress on these amphibians remain unstudied.

To provide suitable habitat conditions for coldwater species, several authors (see Vesely and McComb, 2002; Bury, 2004; Sarr et al., 2005; Olson et al., 2007) recommend buffer zones along headwaters and around seeps to provide shade and reduce sedimentation from management activities. These are now prescribed to protect fish habitat on larger streams (Beschta et al., 1987; Hawkins et al., 1983; Sedell and Swanson, 1984), but are inconsistently applied across geographic regions (Olson et al., 2007) or rare on non-fish bearing streams (see Sheridan and Olson, 2003). Current forest practices increasingly recommend or require riparian buffers along headwaters and small streams (see Bury, 1994; de Maynadier and Hunter, 1995; Diller and Wallace, 1996). These are critical steps toward maintenance of stream conditions and adjacent riparian habitat favorable to amphibians and other forest wildlife. Moreover, many forests have multiple perturbations and landscape effects such as salvage logging occurring after wild fires (Pilliod et al., 2003; Odion and Sarr, 2007) or are a mosaic such as logged habitat upstream of old-growth forests (Corn and Bury, 1989; Biek et al., 2002). It will be important to understand the responses of amphibians and other forest wildlife to these complex interactions. Headwaters, streams and riparian buffers (e.g., 10-25 m wide) should only cover a fraction (e.g., 5-10%) of the land area in most Pacific Northwest forests (Bury and Corn, 1988). However, we lack information on the most effective buffer widths to protect these sensitive riparian zones, which are critical to the survival of cold-adapted species such as *Rhyacotriton* and *Ascaphus*, and salmonid fishes.

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Some Effects of Logging and Associated Road Construction on Northern California Streams[1]

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ABSTRACT

The effects of logging and associated road construction on four California trout and salmon streams were investigated from 1966 through 1969. This study included measurements of streambed sedimentation, water quality, fish food abundance, and stream nursery capacity. Logging was found to be compatible with anadromous fish production when adequate attention was given to stream protection and channel clearance. The carrying capacities for juvenile salmonids of some stream sections were increased when high temperatures, low dissolved oxygen concentrations, and adverse sedimentation did not accompany the logging. Extensive use of bulldozers on steep slopes for road building and in stream channels during debris removal caused excessive streambed sedimentation in narrow streams. Sustained logging prolonged adverse conditions in one stream and delayed stream recovery. Other aspects of logging on anadromous fish production on the Pacific. Coast are discussed.

INTRODUCTION

A major concern of resource managers on the Pacific Coast of the United States and British Columbia has been the effect of timber harvest and associated road construction on salmon and trout. At first interest focused on log jams blocking salmon *(Oncorhynchus* spp.) and steelhead trout (Salmo *gairdneri)* from their spawning grounds. California resolved this problem by a law requiring clear passage for fish and by a log jam removal program (Mongold, 1964) . Then interest shifted to damage caused by bulldozers working in streambeds and along stream banks (Calhoun, 1962, 1966) and to erosion resulting from improper road and skid trail construction on steep terrains (Cordone and Kelley, 1961; Calhoun, 1967) . In July 1966, the California Department of Fish and Game initiated a study in northern California watersheds to determine the effects of logging and associated road building on stream salmonids. This report describes the study from 1966 through 1969 and summarizes the resulting conclusions about streambed sedimentation, water quality, fish food abundance, and stream nursery capacity.

[1]This study was performed as part of Dingell-Johnson California Project F-10-R, "Salmonid Stream. Study," supported by Federal Aid to Fish Restoration funds.

STUDY AREA

Four small streams on the northern California coast were chosen for study (Figure 1): Bummer Lake Creek, South Fork Yager Creek, Little North Fork Noyo River, and South Fork Caspar Creek. They are located within 40 km of the ocean and drain watersheds ranging from 425 to 2,514 ha (Table 1). The watersheds are relatively steep, with canyon sides having mean slopes ranging from 36 to 49%. The coastal climate is characterized. by heavy winter rainfall and dry summers. Mean annual precipitation varies from

TABLE 1.-*Characteristics of the streams and watersheds*

	Bummer Lake Creek	South Fork Yager Creek	North Fork Noyo River	South Fork Caspar Creek
Drainage area of watershed (ha)	1,400	2,514	989	425
Average canyon slope in study section (%)	45	38	36	49
Stream distance from study section to ocean (km)	28.5	40.0	16.0	11.2
Study section length (m)	1,524	1,119	1,530	3,093
Average stream gradient: in study section (%)	5	4	3	3
Average stream width in study section (m)[1]	4.9	5.2	1.5	1.8
Major materials composing streambed surface in study section[2]	cobble & boulder	cobble & boulder	pebble	pebble
Soil series in watershed[3]	Melbourne	Hugo	Hugo	Hugo
Mean annual precipitation (cm)[4]	203	102	127	127
Annual streamflow range liters/sec[5]	14.2-1,416	8.5-934	2.3-396	1.7-255

[1] Measured during minimum flow in summer.
[2] Wentworth's classification (Welch, 1948).
[3] Storie and Weir (1953).
[4] Durenberger (1960).
[5] Range observed during water quality sampling in 1968-69 (Kopperdahl, Burns, and Smith, 1971). Only South Fork Caspar Creek had a streamflow gage and its range exceeded that observed during water qulity sampling. South Fork Caspar. Creek generally reaches a maximum flow of about 1,189 liters/see in the winter (Ziemer, Kojan, Thomas and Muller, 1966).

102 to 317 cm. Air temperatures are cooled by dense, recurrent fogs, with the mean maximum temperature in July being about 21 C (Durenberger, 1960). Soils in these drainages are predominately loam and moderately erodible. The combination of steep slopes, heavy rainfall, and erodible soils renders these watersheds unstable. The watersheds are forested with redwood *(Sequoia sempervirens)* and Douglas fir *(Pseudotsuga menziesii).*

Streamflows fluctuate seasonally, with freshets occurring from November to March, and intermittent flows are common in the headwaters during the summer. Minimum streamflows range from 1.7 to 14.2 liters/sec. These small streams are important spawning and nursery areas for coho (silver) salmon *(Oncorhynchus kisutch)*, steelhead trout, and cutthroat trout *(Salmo clarki).* Sculpins *(Cottus* spp.) and threespine stickleback *(Gasterosteus aculeadus)* inhabit some of the streams. Coho salmon spawn from November through January. Young salmon emerge from the gravels from February through May and usually spend a year in the stream before emigrating to the ocean. Steelhead trout spawn from December to May. Their young emerge from April to June and remain in the stream from 1 to 4 years before emigrating to the ocean (Shapovalov and Taft, 1954). Cutthroat trout form both resident and anadromous populations in California streams from the Eel River north. Cutthroat trout spawn from October to May, and their young follow a stream life similar to that of rainbow trout (DeWitt, 1954.) Fishing pressure on juvenile salmonids is negligible in these streams.

METHODS

Each stream was studied for three summers before, during, and after either logging or road building. This season was selected because it is a critical period for the survival of stream-dwelling salmonids. Living space is limited by streamflow, water temperatures are highest, and most logging occurs in the summer.

Streamflow and stream dimensions were measured systematically within each study stream section (Burns, 1971) . Water quality was monitored periodically after logging, while stream temperatures were recorded each summer (Kopperdahl, Burns, and Smith, 1971). Spawning bed sedimentation was measured (Burns, 1970), using techniques similar to those of McNeil and Ahnell (1964). The

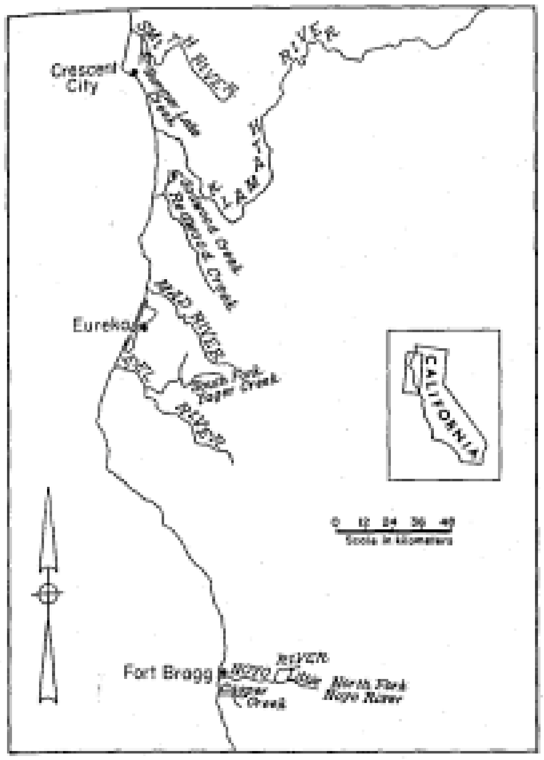

FIGURE 1.-Map of the northern California coast showing the location of study streams.

abundance of juvenile salmonids was estimated at selected times in the summers of 1966 to 1969 (Burns, 1971) . Fish were captured with a battery-powered, DC, back-pack shocker and populations estimated by the Petersen single-census, mark-and-recovery method (Davis, 1964) or by the two-catch, removal method (Seber and LeCren, 1967) . The abundance of fish food and the summer food habits of salmonids in South Fork Caspar Creek were determined (James Burns, John Brode, and Gary Smith, MS; Hess, 1969).

EFFECTS OF LOGGING AND ASSOCIATED ROAD BUILDING ON FOUR CALIFORNIA STREAMS

Bummer Lake Creek

Bummer Lake Creek flows through private lands into the Smith River system, near the California-Oregon border (Figure 1) . A portion of its old growth forest was clear cut in alternate blocks on the southwest slope of the stream in the summer of 1968 (Figure 2) ; 58,000 m^3 of redwood and Douglas fir timber were harvested from 110 ha. Logs were yarded by highlead away from the stream up to the road and by bulldozer above the road. The average horizontal distance between the stream and the road was 120 m, and there were no stream crossings. A bulldozer was operated in the streambed for the removal of logs and other debris from the 1,524-m study section.

Streamflow ranged from 14.8 to 34.5 liters/sec and stream surface area from 0.557 to 0.730 ha during the September surveys (Table 2) . Water temperatures remained cool after the logging and never exceeded 18.3 C. In the clear cut blocks bordering the stream, water temperatures were 4.4 C higher than they were in the uncut section upstream. In the cut sections, stream temperatures increased 1.0 C/100 m. In the uncut block between the two cut blocks, stream temperatures cooled 0.5 C/100 m; thus, beyond this shaded area, temperatures were 2.2 C cooler. Water quality remained within limits tolerated by salmonids. No abnormal concentrations of oxygen, carbon dioxide, pH, alkalinity, chloride, sulfate, nitrate, phosphate, or tannin and lignin were: detected (Kopperdahl *et al.,* 1971).

The mean percentage of spawning bed sediments smaller than 0.8 mm diameter increased from 10.2 to 13.3% after the logging (Burns, 1970), but the increase was not statistically significant at the 5% level (Student's t-test). The bulldozing of logging debris from the streambed did not fill in pools, erode the stream banks, or cause any adverse conditions. The slight increase in fine materials was probably due to erosion on the cut slopes. Apparently the wide stream channel and boulder-and-cobble bottom prevented the bulldozers from gouging the stream channel.

Fish populations were not adversely affected by the logging. The biomass of salmonids was slightly lower during the logging and increased after it (Table 3). The 19% increase in salmonid biomass to 49.2 kg/ha, however, was within the range of natural variation in unlogged California streams (Burns, 1971). Yearling and older trout were fewer after the logging, but young-of-the-year were more abundant. (Steelhead and cutthroat trout pop-

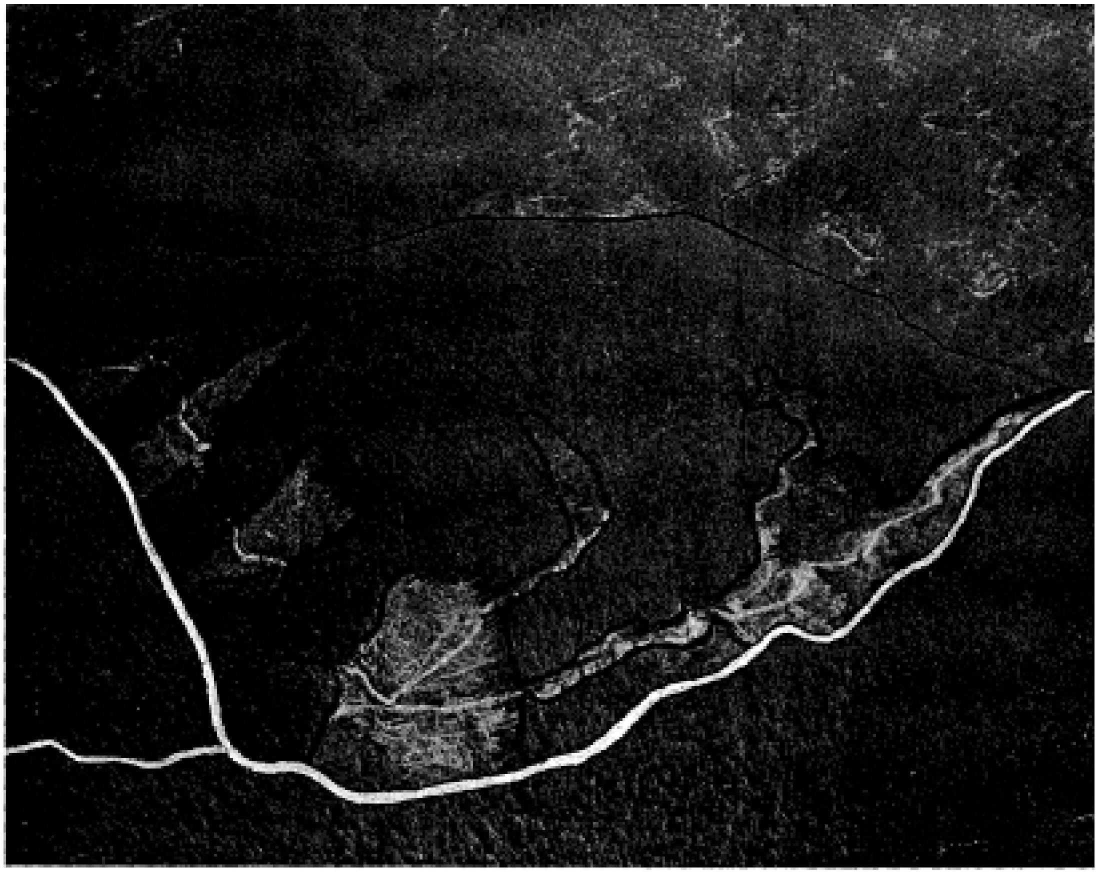

FIGURE 2.--Aerial view of a portion of the Bummer Lake Creek watershed. Light patches are clear cut blocks, while darker patches are uncut blocks. The white line represents the stream and its forks flowing west. The study section extends from the forks 1,524 m downstream through the second cut block.

TABLE 2. *Dimensions of the Bummer Lake Creek study section during the September surveys*

Year	Condition	Streamflow (liters/sec)	Length (km)	Pool surface area (ha)	Riffle surface area (ha)	Volume (m^3)
1967	Unlogged	14.8	1.524	0.459	0.270	1,444
1968	Two months after logging	34.5	1.524	0.511	0.219	1,025
1969	Fourteen months after logging and 11 months after stream cleanup	17.0	1.524	0.340	0.217	764

TABLE 3.-*Population densities, mean fork lengths, and absolute numbers of salmonids in Bummer Lake Creek*

	Steethead rainbow and coast cutthroat trout								
	Young-of-the-year			Yearling and older			Coho salmon		
Survey date	No./m^2 (kg/ha)	Mean fork length (mm)[1]	Number[1]	No./m^2 (kg/ha)	Mean fork length (mm)[1]	Number[1]	No./m^2 (kg/ha)	Mean fork length (mm)'	Number[1]
September 1967	0.62 (14.20)	55(54-56)	4,509 (4109-4909)	0.14 (25.42)	112(111-114)	1,003 (797-1209)	0.04 (1.54)	69(67-71)	279 (221-336)
September 1968	0.39 (13.25)	64 (63-66)	2,916 (2698-3134)	0.10 (18.39)	113 (112-114)	720 (606-834)	0.15 (4.85)	63 (62-64)	1,124 (938-1323)
September 1969	0.93 (28.76)	62 (61-63)	5,175 (5018-5332)	0.08 (19.58)	1.30 (125-135)	430 (335-525)	0.02 (0.87)	70 (68-72)	111 (98-124)

[1] 95% confidence intervals in parentheses.

FIGURE 3.-Aerial view of a portion of the South Fork Yager Creek watershed. Center white line represents the stream flowing north to main Yager Creek. Black lines outline area of forest which was selectively logged. A buffer strip was left along the stream.

ulations were pooled because of the difficulty in identifying juveniles in the field; of the older fish that could be identified, about 75% were cutthroat.) All age groups of trout had longer mean lengths after the logging. Coho salmon formed a marginal population in this stream and showed considerable population fluctuation. Their mean length did not change significantly after the logging. Sculpins also increased in biomass after the logging, from 1.3 kg/ha in 1967 to 6.2 in 1968 and 21.8 in 1969.

South Fork Yager Creek

South Fork Yager Creek flows through private lands into the Van Duzen-Eel River System, south of Eureka, California (Figure 1) . Old growth timber was cut selectively in the summer of 1968 from the mouth of South Fork Yager Creek upstream 560 m and over an area of 305 m on each side of the stream (Figure 3). Eighty percent of the timber volume was cut from the original volume of 34.4 m^3/ha. Yarding was done with bull-

TABLE 4. *Dimensions of the South Fork Yager Creek study section during the August surveys*

Year	Condition	Streamflow (liters /sec)	Length (km)	Pool surface area (ha)	Riffle surface area (ha)	Volume (m^3)
1967	Unlogged	16.9	0.566	0.211	0.099	378
1968	Immediately after logging	14.9	0.566	0.152	0.119	372
1969	Twelve months after logging	20.4	0.566	0.251	0.075	446

TABLE 5.-*Population densities, mean fork lengths, and absolute numbers of steelhead rainbow trout in South Fork Yager Creek*

Survey date	Young-of-the-year Number/m² (kg/ha)	Young-of-the-year Mean fork length (mm)[1]	Young-of-the-year Number[1]	Yearling and older Number/m² (kg/ha)	Yearling and older Mean fork length (mm)[1]	Yearling and older Numbers
August 1967	1.22 (13.38)	42(41-43)	3,781 (3522-4040)	0.02 (4.08)	115(101-129)	65 (25-103)
August 1968	1.08 (21.61)	58 (57-59)	2,932 (2838-3026)	0.04 (8.02)	118 (114-123)	116 (71-161)
August 1969	1.74 (22.60)	48 (47-49)	5,668 (5449-5886)	0.07 (13.46)	123 (120-126)	212 (157-267)

[1] 95% confidence intervals in parentheses.

dozers. Great care was taken to protect the stream during the logging. Riparian vegetation, including merchantable redwood and Douglas fir trees leaning toward the stream, was not cut, and heavy equipment did not enter the stream. Roads and landings were built away from the stream on low gradients.

Streamflow ranged from 14.9 to 20.4 liters/sec and stream surface area from 0.271 to 0.326 ha during the August surveys (Table 4).

Water temperatures did not increase after the logging. Temperatures were high in all years, usually reaching 21.5 C in the summer. The protection of riparian vegetation along the stream prevented stream temperatures from increasing to lethal levels after the logging. No abnormalities in water quality were detected after the logging (Kopperdahl *et al.*, 1971).

The mean percentage of spawning bed sedi-

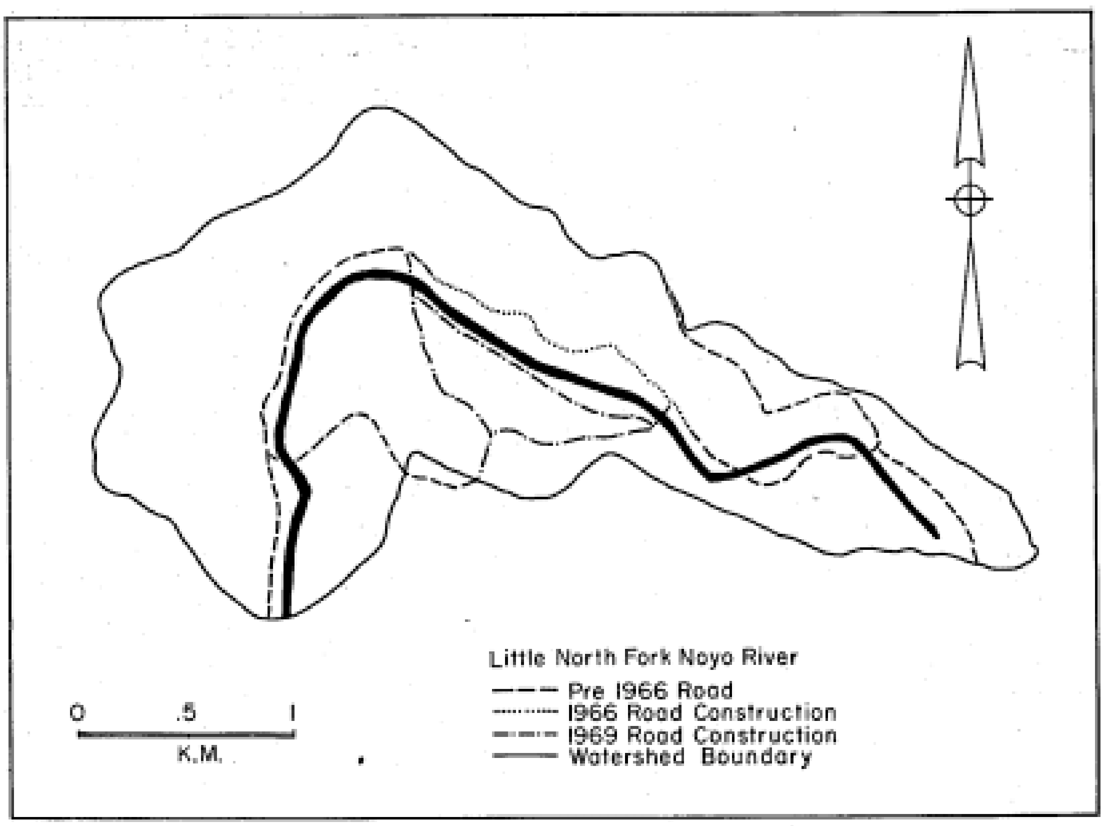

FIGURE 4.—Diagram of Little North Fork Noyo River road construction.

FIGURE 5.-Little North Fork Noyo River received considerable alteration during road construction. The road on the right of the photograph was built in 1966, while the road at the top was built in 1969.

ments smaller than 0.8 mm diameter increased from 16.4 to 22.1% after the logging; however, this increase was not due to the logging but was due to the release of sediments from the collapse of a tree jam-and-rock barrier upstream from the study section (Burns, 1970) .

Fish populations increased after the logging (Table 5). The biomass of all age groups of steelhead trout increased and their mean lengths were longer after the logging. The 106% increase in biomass of steelhead trout to 36.1 kg/ha was greater than the natural range of fluctuation in unlogged streams. However, data from an upstream area indicates that this stream was not at carrying capacity during the prelogging population census (Burns, 1971) and, therefore, the entire increase cannot be attributed to the logging. Sculpin and stickleback populations decreased after the logging from 2.3 kg/ha in 1967 to 0.9 in 1968 and 0.4 in 1969.

Little North Fork Noyo River

Little North Fork Noyo River flows through private lands into the Noyo River, near Fort Bragg, California (Figure 1). Its second-growth (logged 100 years ago) redwood and Douglas fir forest has been subjected to selective logging since 1964 (Figure 4). Thirty percent of the timber volume has been removed from 542 ha of watershed since 1966. A bulldozer worked in or near the 1,530-m study section during road construction and right-of-way logging in the fall of 1966 and in the spring of 1969 (Figure 5). Yarding was done with bulldozers. Average distance from the road to the stream was 23 m. There was one bridge crossing at the upper end of the study section.

Streamflow ranged from 2.2 to 7.3 liters/sec and stream surface area from 0.609 to 0.998 ha during the October surveys (Table

TABLE 6.- *Combined dimensions of tire four study areas in Little Fork Noyo River daring the October surveys*

Year	Condition	Streamflow (liters/sec)	Length (km)	Pool surface area (ha)	Riffle surface area (ha)	Volwne (m^3)
1966	Pre-road construction	2.2	0.399	0.414	0.195	93
1968	Twenty-four months after initial road construction and 12 months after gully logging	5.4	0.399	0.619	0.379	122
1969	Immediately after second road con struction	7.3	0.424	0.547	0.447	91

6) . The selective removal of timber along the stream opened the forest canopy and undoubtedly increased stream temperatures; however, instrument damage and malfunctions prevented collection of stream temperature data before the road construction. Temperatures after the road construction and logging, however, did not exceed 21.1 C. No abnormalities in water quality were detected after the logging (Kopperdahl *et al*., 1971) .

Bulldozer activities increased stream turbidity and spawning bed sediments. After a light rain in November 1969, turbidity reached 53 J. T. U. (Kopperdahl *et al*., 1971). Two years after the construction of an all-weather road adjacent to the stream, the mean percentage of sediments smaller than 0.8 mm had increased from 20.0 to 31.0% (Burns, 1970). After a second road had been constructed on the other side of the stream and the streamside selectively logged, sediments smaller than 0.8 mm increased to 33.3%. These increases were statistically significant at the 5% significance level (Student's t-test). The pebbles and small gravel composing the narrow stream channel were easily gouged, leaving a heavily silted streambed, with the stream flowing along two bulldozer tracks (Figure 6) .

Fish populations decreased as watershed and stream disturbances progressed on Little North Fork Noyo River (Table 7) . Steelhead trout numbers remained about the same, but the trout were smaller after the logging, and consequently their biomass decreased 42%. The numbers and biomass of coho salmon decreased more markedly. Biomass decreased 65%, even though the average weight of coho salmon increased as population densities decreased. The total biomass of salmonids decreased 62% to 9.3 kg/ha and this decrease was greater than that of unlogged streams (Burns, 1971). Sculpin abundance decreased each time the streambed became heavily silted, but the sculpins were quick to recover. Before the road construction there were 1.6 kg/ha, and 24 months after road construction there were 11.6 kg/ha. Immediately after the 1969 stream disturbances, the sculpin population was down to 0.4 kg/ha.

South Fork Caspar Creek

Caspar Creek, which flows through Jackson State Forest just south of Fort Bragg, received

FIGURE 6.-After a bulldozer operated in the streambed of Little North Fork Noyo River, the streamflow split into two separate channels, each formed by bulldozer tracks.

TABLE 7. *Population densities, mean weights, and absolute numbers of salmonids in Little North Fork Noyo River*

	Steelhead rainbow trout			Coho salmon		
Survey date	Number/m2 (kg/ha)	Mean weight (g)[1]	Number[1]	Number/m2 (kg/ha)	Mean weight (g)[1]	Number[1]
October 1966	0.03 (3.66)	11.8	19 (11-27)	1.15 (20.70)	1.8	698 (672-724)
October 1968	0.03 (1.73)	6.0 (4.8-7.2)	29 (23-35)	0.40 (9.66)	22.4 (2.2-2.6)	403 (390-416)
October 1969	0.03 (2.14)	8.6 (5.6-11.5)	25 (24-26)	0.26 (7.15)	2.8 (2.5-3.0)	255 (238-272)

[1] 95% confidence intervals in parentheses.

more attention than the other study streams since there was an interagency program (U.S. Forest Service, California Division of Forestry, California Department of Water Resources, Humboldt State College, and California Department of Fish and Game) to determine the effects of road construction on streamflow, sedimentation, fish life, and fish habitat (U.S. Forest Service, 1965) .

The South Fork's second-growth forest of redwood and Douglas fir was disturbed by 6.0 km of road construction in the summer of 1967 (Figure 7) . Nineteen thousand-four hundred m of sawlogs were harvested and 18,800 m3 of road materials moved during the road right-of-way construction. The road was built adjacent to the stream, ranging from four bridge crossings (Figure 8) to 76 m at the furthest point from the stream. Road materials were side-cast into a portion of the stream and 79m of the stream were relocated.

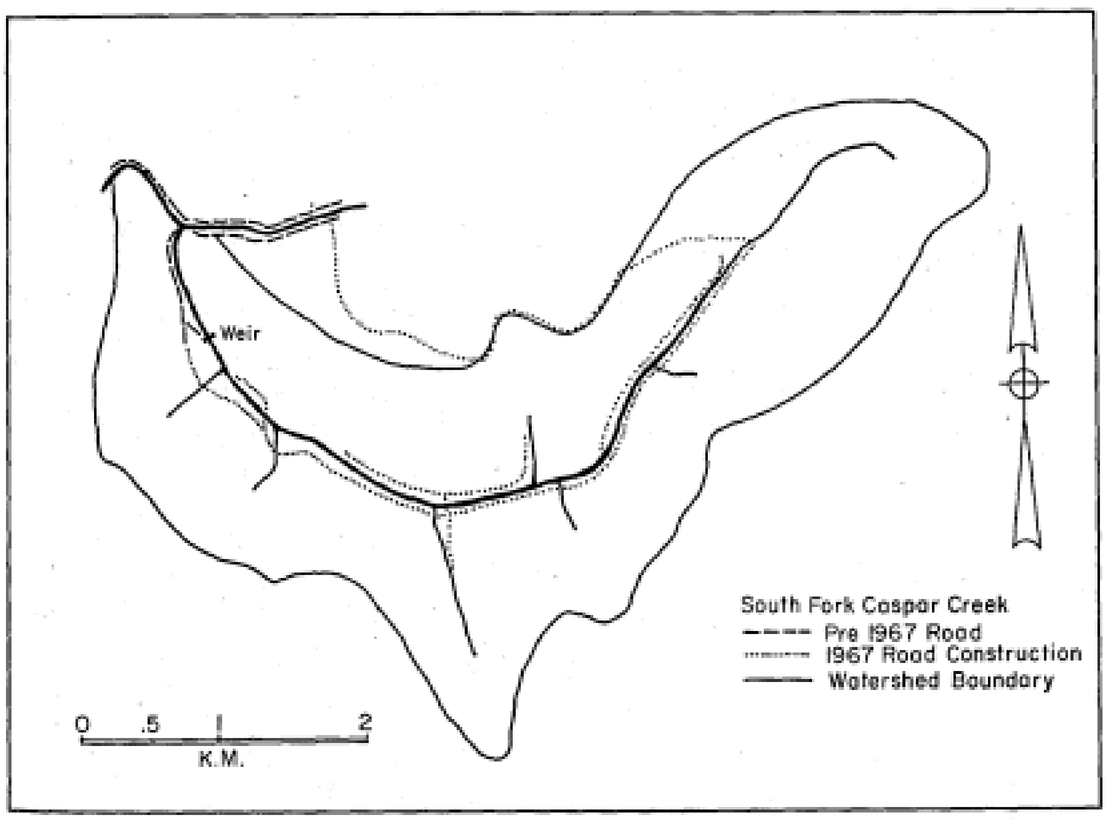

FIGURE 7.—Diagram of South Fork Caspar Creek road construction.

FIGURE 8.-*South Fork Caspar Creek, showing one of the four bridge crossings and the stream disturbance resulting from* bridge construction.'

A bulldozer operated through 41% of the 3,093-m study section during the yarding of logs and the removal of debris. Most of the fill slopes, secondary roads, and streambank were fertilized with urea and seeded with annual rye grass (*Elymus* sp.) at a rate of 56 kg/ha. No logging trucks used the road after the first summer. Road slides occurred during

TABLE 8.-*Dimensions of the South Fork Caspar Creek study section during the June and October surveys*

Year	Condition	Streamflow (liters/sec)	Length[1] (km)	Pool surface area (ha)	Riffle surface area (ha)	Volume (m^3)
June 1967	Pre-road construction	12.7	3.091	0.295	0.307	367
October 1967	Immediately after road construction	4.6	2.896	0.193	0.301	226
June 1968	Eight months after road completion	8.0	3.039	0.290	0.210	305
October 1968	Twelve montbs after road completion	2.3	2.723	0.216	0.191	186
June 1969	Twenty-two months after road completion	12.8	3.031	0.367	0.289	400
October 1969	Twenty-three months after road completion	5.1	2.887	0.234	0.162	234

[1] Variable length due to intermittent flow drying some stream sections.

the winter and road repair was necessary in the springs of 1968 and 1969.

Streamflow ranged from 12.8 to 2.3 liters/sec and stream surface area from 0.656 to 0.396 ha during the June and October surveys (Table 8) . Prior to the road construclion, the amount of solar radiation received at some stations along the stream was less than 5% of the total available at that latitude. Even on clear days, about half of the stream received less than 10% of the available radiation because of the dense streamside vegetative canopy. Absolute values ranged from 7 to 276 langleys/day (DeWitt, 1968). After the road construction some stations along the South Fork received 140% more radiation than they had previously. The absolute average increased 98 langleys/day. Increased solar radiation increased stream temperatures. Water temperatures increased as much as 11.1 C during the road construction at some South Fork stations (Hess, 1969) . The maximum observed temperature was 25.3C (DeWitt, 1968). The maximum preconstruction temperature at the downstream end of the study section was 13.9 C (Kabel and German, 1967), while the postconstruction maximum for this station was 21.1 C.

During the logging, dissolved oxygen dropped to 5 ppm in some isolated pools with decaying slash, while undisturbed stream sections had 10 ppm (Richard Brandon, Humboldt State College, pers. comm.). In August 1968, 11 months after cessation of the right-of-way logging, the concentration of carbon dioxide was 8 ppm (Kopperdahl *et al*., 1971) . However, this level of carbon dioxide is not lethal to salmonids (McKee and Wolf, 1963) . The increase in carbon dioxide probably resulted from decomposition of logging slash in the streambed. Unlogged streams on the coast had concentrations of carbon dioxide of less than 2 ppm during this same period (Kopperdahl *et al*., 1971) . Other concentrations of chemicals within South Fork Caspar Creek were generally normal.

High turbidities were localized in areas where a bulldozer was working in the stream during bridge construction. Silt extended only a short distance downstream from the disturbance and the stream cleared quickly upon cessation of bulldozer activities. During a moderately heavy rainfall in the first winter after road construction, erosion and slippage of the road caused turbidities of 3,000 ppm and deposition of as much as 0.6 m of sediment in the stream (Hess, 1969) . The volume of sediments smaller than 0.8rnm increased from 20.6 to 34.2% immediately after road construction. The next summer this class of sediments returned to the predisturbance level. Twenty-two months later, however, this class of sediments was up to 28.5%. These changes in streambed composition were statistically significant at the 5% significance level. The initial increase in 1967 followed extensive use of a bulldozer to clear the stream of logging debris (Figure 9) . The narrow streambed composed of small materials was particularly susceptible to degradation. Erosion was lessened the first winter and spring by planting annual, rye grass on the stream banks, fill slopes, and skid trails. Without excessive erosion, accumulated sediments were scoured from the riffles by the summer 1968. The increase in 1969 resulted from erosion of the streambank, side casts, and slides.

The road construction and right-of-way logging were immediately detrimental to most aquatic invertebrates in South Fork Caspar Creek, although conditions favored Diptera and Plecoptera (J. W. Burns, J. M. Brode, and G. E. Smith, MS). Increases in these two orders offset the losses in other invertebrates, causing an increase in benthos from 286 mg/m^2 to 634 mg/m^2 (120% increase) immediately after the road construction and fertilization with 817 kg urea. North Fork Caspar Creek (an unlogged, second-growth stream used as one of the controls for this study) also showed a 120% increase in benthos; there fore, the immediate increase in the South Fork cannot be interpreted as being caused by the road construction and fertilization (J. W. Burns, J. M. Brode, and G. E. Smith, MS). Recolonization of the South Fork was rapid and, within two years, the South Fork's benthos increased 370% over the preroad construction biomass. The North Fork's benthos increased only 64% during the same period (J. W. Burns, J. M. Brode, and G. E. Smith, MS). Ephemeroptera took longer to recover

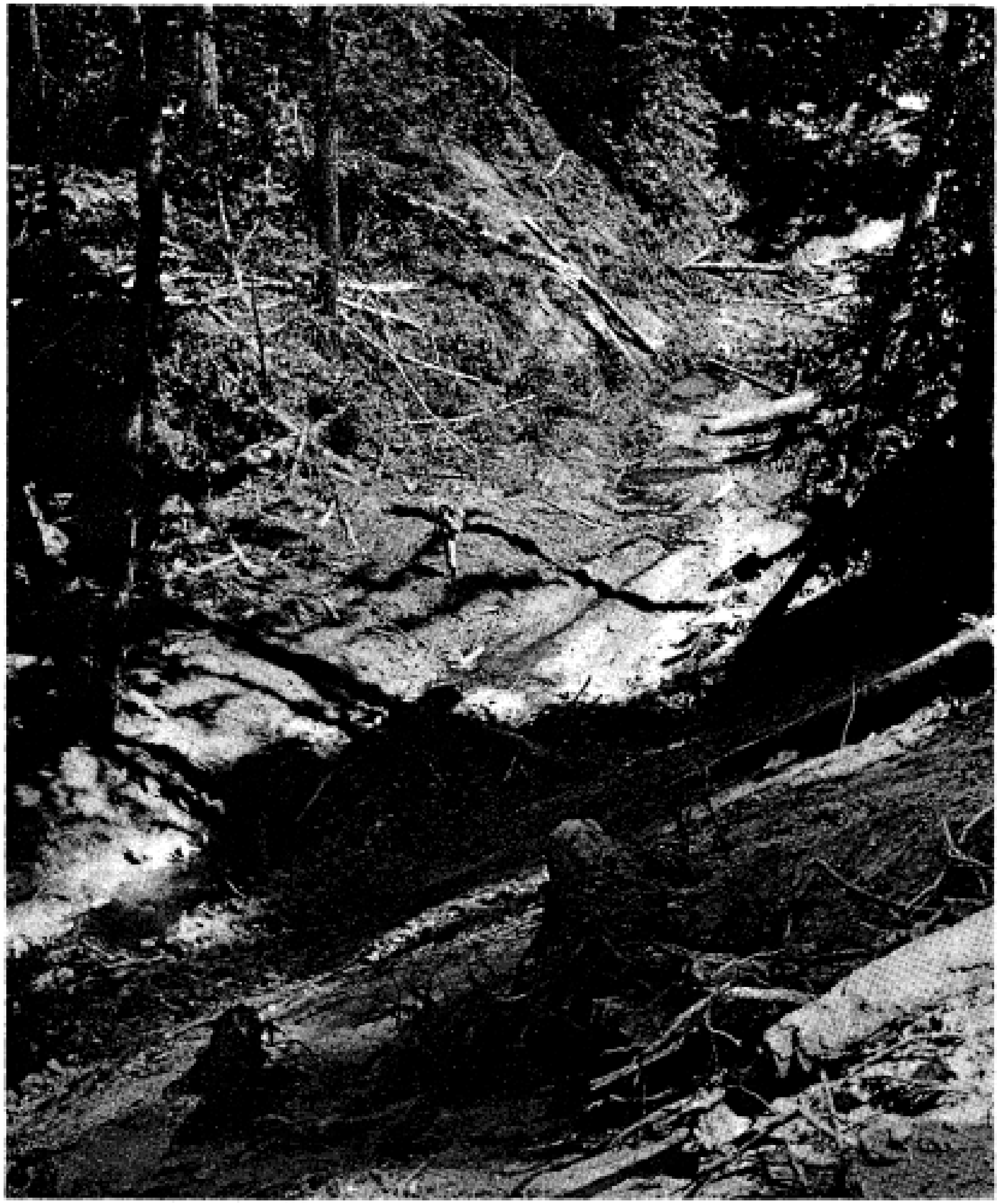

FIGURE 9.-South Fork Caspar Creek was grossly altered by road construction. The primary cause of damage was the operation of bulldozers in the stream channel.

than did most other insect orders. Trichoptera recovered rapidly and along with Plecoptera and Diptera made up the majority of the South Fork's benthos. Trichoptera drift increased more than drift of other orders after the road construction. The occurrence of terrestrial organisms in the drift appeared to be influenced only slightly by the disturbance. The total drift increased 47% by the first spring after the road construction and 100%

TABLE 9.- *Population densities, mean fork lengths, and absolute numbers of salmonids in South Fork Caspar Creek*

	Steelhead rainbow trout								
	Young-of-the-year			Yearling and older			Coho salmon		
survey date	No./m^2 (kg/ha)	Mean fork length (mm)[1]	Number[1]	No./m^2 (kg/ha)	Mean fork length (mm)[1]	Number[1]	No./m^2 (kg/ha)	Mean fork length (mm)[1]	Number'
June 1967	1.69 (11.81)	37 (36-38)	10,183 (9507-10859)	0.11 (10.26)	86 (78-93)	673 (362-984)	1.00 (15.90)	47 (45-48)	6,001 (5613-6389)
October 1967	0.29 (4.94)	50 (48-52)	1,436 (1313-1559)	0.02 (4.53)	124 (112-135)	106 (95-117)	0.21 (5.45)	58 (56-60)	1,038 (962-1114)
June 1968	1.32 (10.51)	43 (42-44)	6,580 (6473-6687)	0.04 (3.65)	95 (92-99)	176 (141-211)	0.50 (7.42)	49 (48-50)	2,510 (2452-2568)
October 1968	0.58 (12.77)	58 (57-59)	2,363 (2307-2419)	0.01 (2.13)	115 (108-122)	51 (33-69)	0.32 (5.78)	54 (53--55)	1,283 (1244-1322)
June 1969	1.45 (17.38)	47 (46-48)	9,512 (9238-9786)	0.06 (5.70)	92 (89-94)	407 (303-511)	0.77 (11.62)	51 (50-52)	5,036 (4833-5239)
October 1969	0.81 (17.07)	57 (56-58)	3,224 (3153-3295)	0.04 (5.23)	111 (107-113)	141 (91-191)	0.48 (8.08)	54 (53-55)	1,885 (1849-1921)

[1] 95% confidence intervals in parentheses.

by the second spring. The weight of insects dropping into the South Fork doubled over the preroad construction values (Hess, 1969) . The greatest increase was in those having aquatic immature stages, the increase being exceptional in Trichoptera and Diptera. Aquatic organisms were more important in the diets of steelhead trout and coho salmon than were terrestrial organisms. Diptera became more important in the diets of South Fork salmonids after the road construction.

Salmonid populations decreased immediately after the road construction (Table 9) . Recovery began the following spring and by the second spring the salmonid biomass was only 20% lower than the predisturbance biomass of 38.0 kg/ha. All age groups of salmonids had greater mean lengths after the road construction. The road construction may have reduced the total yield of coho salmon and steelhead trout smolts in 1968 and 1969, because of high mortality of both species and the premature emigration ofyearling and older steelhead trout in 1968 (Graves and Burns, 1970) . Population changes in the summer (June to October) were highest in 1967, when the road was built into the South Fork's watershed. The population of young-of-the-year (Age +) steelhead trout decreased 85%, older (Age 1+) steelhead trout decreased 84%, and coho salmon decreased 82%. These rates were much higher than the average decreases of 65% for Age + steelhead trout, 68% for Age 1+ steelhead trout, and 55% for coho salmon in 1968 and 1969. The oversummer loss of South Fork salmonids was also higher than the decrease observed for the same period in North Fork Caspar Creek. In the summer of 1967, these were 69, 25, and 16% for the respective species and age groups in the North Fork (Burns, 1971). Some of the decrease in the South.Fork in 1967 may have resulted from emigration of some large steelhead trout from the study area to the pools formed behind the streamflow gaging and fish trapping facilities. Downstream migrant census data for the spring of 1968 suggest that these pools provided refuge for a few fish during road construction (Graves and Burns, 1970) . In the strict sense, then, not all of the decrease was mortality. In 1968 downstream migrants in both the North and South Forks were monitored from June to October. Only young-of-the-year fish entered the traps, with 6 steelhead trout .and three coho salmon trapped in both forks. These data suggest that few fish normally migrate downstream in Caspar. Creek during the summer. The combined smolt yield of steelhead trout and coho salmon in the South Fork for the spring of 1968 was 20% lower than the preroad construction smolt yield, but was within the range of other California streams (Graves and Burns, 1970) . Stickleback biomass fluctuated widely during the study, showing an overall increase after the road construction. June and October biomasses were: 0.2 to 0.7 kg/ha in 1967, 6.6 to 2.5 in 1968, and 1.1 to 5.0 in 1969.

DISCUSSION

Studies of the effects of logging reported from California, Alaska (Sheridan and McNeil, 1968; Meehan, Farr, Bishop, and Patric, 1969), arid Oregon (Hall and Lantz, 1969) suggest that logging is compatible with anadromous fish production if adequate attention is given to stream and watershed protection and channel clearance. Under special circumstances, stream salmonid production can even be enhanced by logging. Cold streams, shaded by dense forest canopies are not optimum trout habitats (White and Brynildson, 1967) . Thinning the riparian canopy allows a greater total solar radiation to reach the stream, rais ing temperatures a few degrees (viz., Bummer Lake Creek), and increasing the production of bacteria, algae, and the insects upon which fish feed (viz., South Fork Caspar Creek). Salmonid biomass increased in two California streams (Bummer Lake Creek and South Fork Yager Creek) after the streams were carefully logged.

Temperature increases can be predicted and modified by leaving shade along the stream (Brown, 1969). A dense understory or buffer strip (e.g., South Fork Yager Creek) can effectively keep temperatures cool. Alternating cut and uncut sections (e.g., Bummer Lake Creek) can be used to control temperatures. Increases are at least partially reversible if the warmed water passes through shaded areas.

Logging often results in higher summer streamflow (Rothacher, 1965; Hibbert, 1967), providing more living area for juvenile salmonids and thereby increasing the fish rear ing capacity of the stream. If protective logging is compatible with fish production, then what logging activities are incompatible or need special attention? Chapman (1962) reviewed many of the effects of logging on fishery resources; many of his points are reviewed here and others observed in this study and in Oregon (Hall and Lantz, 1969) are discussed.

Removing too much of the forest canopy, such as cutting all or a major portion of a watershed, can have drastic results for salmonids. Warmed waters entering the main stream from several logged tributaries may increase main stream temperatures beyond those tolerated. by salmonids. Temperatures above 25 C for extended periods are usually lethal to salmonids (Brett, 1952) . Streams can reach lethal temperatures or, more commonly, levels which increasemetabolicrates and maintenance requirements, increase pathogenic activity, and decrease the solubility of oxygen. These dangers are even more critical inland, away from cooling influences of coastal fog. Temperatures of California streams within the coastal fog belt did not exceed 21 C for extended periods. Trout production in some sections of Berry Creek. Oregon, was not increased by removing the forest canopy, even though the amount of solar radiation reaching the cleared sections was triple that reaching the shaded sections. Algal production was much higher in the cleared sections; however, this increase was offset by decreases in terrestrial plant debris available for insect foods (Warren, Wales, Davis, and Doudoroff, 1964). Terrestrial detritus and leaf fragments are apparently more important as food to insects eaten by coho salmon than are aquatic plants (Chapman and Demory. 1963).

Extensive use of bulldozers on steep slopes or in stream channels can cause excessive erosion which can be deleterious to salmonid reproduction. Small streams with narrow channels seem most vulnerable to this type of damage. The mean volumes of streambed sediments smaller than 0.8 mm in Little North Fork Noyo River and South Fork Caspar Creek exceeded 30% during the logging but probably were less during the salmon and steelhead spawning periods (Burns, 1970) . Had fine sediments remained this abundant after the spawning, salmonid survival to emergence would probably have averaged less than 10% (Hall and Lantz, 1969) . Building roads away from the stream (viz., Bummer Lake Creek), or leaving a buffer strip (viz., South Fork Yager Creek) to intercept sediments and slash protects the stream habitat. Seeding the disturbed areas with grass (viz., South Fork Caspar Creek) mitigates the damage. Streambed compaction which prevents the digging of redds or impairs the emergence of fry was not observed in the California streams studied, but has been observed in other California streams.

Excessive erosion from logging frequently fills pools necessary for the rearing of larger salmonids (Fisk, Gerstung, Hansen, and Thomas, 1966). Pools filled with sediment in Little North Fork Noyo River and South Fork Caspar Creek were scoured during each winter after the road construction and logging, thus providing adequate living space the following vear. However, both streams built up numer ous sediment bars, thus forming unstable streambeds with considerable gravel movement during periods of high streamflow. Extensive streambed movement is not unusual for California streams. For example, in unlogged North Fork Caspar Creek as much as 2.3 m^3 of sediment per hectare of watershed has been deposited behind the streamflow gaging weir in a single year (Jay S. Krammes, U. S. Forest Service, pers. comm.). After road construction, the greatest amount of sediment deposited behind the South Fork Caspar Creek weir in one year was 0.7 m^3/ha.

Logging often results in higher peak streamflows and more rapid attainment of peaks (Hibbert, 1967) . High flows accompanying a large deposition of sediments from side slope and streambank erosion will cause a great deal of streambed movement and stream turbidity. Streambed movement can crush and dislodge developing salmonid embryos and fry (James, 1956) . Excessive turbidity is especially condemned by fishermen, since it limits their fishing days.

Another important consideration is the time of year when the logging occurs. Felling trees into the stream when embryos and fry are in the gravel is deleterious, since decaying slash depletes intragravel dissolved oxygen (Hall and Lantz, 1969) or produces copious amounts of slime bacteria (*Sphaerotilus*) which suffocate developing eggs and alevins (Gordon and Martens, 1969) . This emphasizes the importance of keeping excessive amounts of slash out of streams. Because most of this slash was removed after the logging, dissolved oxygen concentrations inthe California streams studied were generally at saturation. The most desirable practice is to keep all timber out of the stream. In my investigations, the major reason bulldozers entered stream channels was to remove logging debris. Another reason for keeping timber out of streams is to prevent the continual formation of log jams. Few loggers remove all trees, limbs, and other debris to above the high water level. Usually streams have to be cleared after each winter, since high water washes materials back into the stream, where they accumulate and form new barriers to fish migration.

Sustained logging and associated road construction over a period of many years do not afford either the stream or the 'fish population a chance to recover. Logging operations on the California streams studied were usually limited to one season and to only a small fraction of the total watershed. Had the watersheds been more extensively logged, changes may have been more severe. Prolonged disturbances (viz., Little North Fork Noyo River) damage stream habitat and fish populations. Logging operations should be completed in the short est time possible and then the watershed left to recover. South Fork Caspar Creek recovered quite rapidly from extensive stream damage, although recovery may have been accelerated by streamside fertilization and seeding and by scheduling the major logging operation after the stream had recovered from the road construction. My studies and those in Oregon demonstrate that coho salmon and steelhead trout are resilient fish, able to compensate for adversities. Generally, the yields of downstream migrants were not drastically reduced and juvenile populations recovered rapidly. In a clear cut operation in a Douglas fir watershed in Oregon, the numbers of juvenile coho emigrating to the ocean during the two years after the logging were within the range of variation recorded before the logging (Hall and Lantz, 1969) . Some fish killed in the summer during the logging would die during this period, anyway. In the summer, when populations are large and mortality is great, the impact of logging is not as severe as it is in the period following population adjustment to stream carrying capacity. In the fall and spring, when smolts are preparing for their downstream migration, the added mortality caused by logging could have serious consequences. The loss of yearling and older fish killed during this period would have a direct effect upon smolt yields. In some of the California streams studied (Bummer Lake Creek and South Fork Caspar Creek), there were fewer yearling and older trout after the logging. The impact of this decrease on smolt yields is not well understood since the loss of older trout may be mitigated by the increased growth and survival of the remaining fish. However, it is known that the larger, older smolts have the highest ocean survival (Shapovalov and Taft, 1954) . The prime objective of protecting anadromous fish streams from adverse watershed use is to maintain the quality and quantity of smolts entering the ocean. A secondary objective is to maintain water clarity, so that anglers can effectively fish for adults.

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SURVEY OF LIVESTOCK INFLUENCES ON STREAM AND RIPARIAN ECOSYSTEMS IN THE WESTERN UNITED STATES

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Interpretive Summary

Livestock grazing has damaged approximately 80% of stream and riparian ecosystems in the western United States. Although these areas compose only 0.5-1.0% of the overall landscape, a disproportionately large percentage (~70-80%) of all desert, shrub, and grassland plants and animals depend on them. The introduction of livestock into these areas 100-200 years ago caused a disturbance with many ripple effects. Livestock seek out water, succulent forage, and shade in riparian areas, leading to trampling and overgrazing of streambanks, soil erosion, loss of streambank stability, declining water quality, and drier, hotter conditions. These changes have reduced habitat for riparian plant species, cold-water fish, and wildlife, thereby causing many native species to decline in number or go locally extinct. Such modifications can lead to large-scale changes in adjacent and downstream ecosystems.

Despite these disturbances, some people support continued grazing. These advocates argue that most of the damage occurred 50-100 years ago; however, recent studies clearly document that livestock continue to degrade western streams and rivers, and that riparian recovery is contingent upon total rest from grazing.

Abstract

This paper summarizes the major effects of livestock grazing on stream and riparian ecosystems in the arid West. We focused primarily on results from peer-reviewed, experimental studies, and secondarily on comparative studies of grazed vs. naturally or historically protected areas. Results were summarized in tabular form. Livestock grazing was found to negatively affect water quality and seasonal quantity, *stream channel morphology, hydrology, riparian zone soils, instream and streambank vegetation, and aquatic and riparian wildlife. No positive environmental impacts were found. Livestock were also found to cause negative impacts at the landscape and regional levels. Although it is sometimes difficult to draw generalizations from the many studies, due in part to differences in methodology and environmental variability among study sites, most recent scientific studies document that livestock grazing continues to be detrimental to stream and riparian ecosystems in the West.*

Introduction

Grazing by livestock has damaged 80% of the streams and riparian ecosystems in arid regions of the western United States (U.S. Department of the Interior (USDI) 1994a). A number of symposia (e.g. Warner and Hendrix 1984, Johnson et al. 1985, Gresswell et al. 1989, Meehan 1991, Clary et al. 1992) and reviews (Platts 1981b, 1982, 1991, Kauffman and Krueger 1984, Skovlin 1984, Chaney et al. 1990, 1993, Armour et al. 1994, Fleischner 1994, Rhodes et al. 1994, USDI 1994a, Kattelmann and Embury 1996, Ohmart 1996) describe this degradation. Livestock grazing affects watershed hydrology, stream channel morphology, soils, vegetation, wildlife, fish and other riparian-dependent species, and water quality at both local and landscape scales. Because riparian and stream ecosystems represent only 0.5-1% of the surface area of arid lands of the eleven western United States (U.S. General Accounting Office (US-GAO) 1988, Chaney et al. 1990, Ohmart 1996), they were historically ignored by land managers. In fact, riparian habitats in the West were viewed until the late 1960s as "sacrifice" areas (e.g., Stoddart and Smith 1955), being dedicated primarily to providing food and water for domestic livestock.

Recently, both critics and advocates of arid-land livestock grazing have focused their attention on western streams and their associated riparian zones, especially those in shrublands, grasslands, and deserts of the Southwest, Great Basin, and Pacific Northwest. Critics of grazing emphasize damage to riparian habitats to illustrate the unsuitability of cattle grazing in the arid West, while advocates of grazing argue that most of the damage to land and streams occurred 50-100 years ago, before modern grazing systems were instituted.

The evidence is undeniable that early grazing practices -- before the Taylor Grazing Act in 1934 established some control over livestock grazing in the public domain -- were highly destructive (Duce 1918, Bryan 1925, Leopold 1946). However, recent studies document that livestock grazing remains a key factor in the continued degradation of riparian habitats (US-GAO 1988, Szaro 1989, Platts 1991, Elmore and Kauffman 1994, Fleischner 1994, McIntosh et al., 1994, USDI 1994a, Ohmart 1996). As recently as 1990, Chaney et al. (1990) wrote in a U.S. Environmental Protection Agency (US-EPA) report on livestock grazing that "extensive field observations in the late 1980s suggest that riparian areas throughout much of the West were in their worst condition in history" (p.5). A joint Bureau of Land Management (BLM) and US Forest Service Report (USDI 1994a) also concludes that "riparian areas have continued to decline [since 1934]" (p.25) and estimates that 20% of the riparian areas managed by BLM are "non-functioning" and 46% are "functioning at risk." Altogether, less than 20% of potential riparian habitat in the western United States still exists (USDI 1994a). This continued decline has been attributed, in part, to increased numbers of cattle in western rangelands (Trimble and Mendel 1995); between 1940 and 1990, the number of cattle in the western United States increased from 25,500,000 to 54,400,000.

Recent scrutiny by scientists reflects a growing recognition by the public, land managers, and scientific community of the importance of streams, rivers, and riparian habitats to western ecosystems. One reason for this interest is the high productivity and biodiversity of riparian systems, which is due, in part, to their high soil moisture and fertility levels (Hubbard 1977, Meehan et al. 1977, Thomas et al. 1979, Knight and Bottorff 1984, Fleischner 1994, Ohmart 1996). Riparian areas in arid and semi-arid regions are composed of complex edaphic and vegetation mosaics because of high variability in landforms, soil types, and location of surface and subsurface water (Thomas et al. 1979, Green and Kauffman 1995, Lee et al. 1989, Gregory et al. 1991). These mosaics, plus extensive borders (ecotones) between moist streamsides and arid uplands, result in high species diversity (Thomas et al. 1979, Lee et al. 1989). An estimated 60-70% of western bird species (Ohmart 1996) and as many as 80% of wildlife species in Arizona and New Mexico (Chaney et al. 1990) and in southeastern Oregon (Thomas et al. 1979) are dependent on riparian habitats. Consequently, riparian ecosystems are considered to be important repositories for biodiversity throughout the West.

Riparian zones provide key services for all ecosystems, but are especially important in dry regions, where they provide the main source of moisture for plants and wildlife, and the main source of water for downstream plant, animal, and human communities (Meehan et al. 1977, Thurow 1991, Armour et al. 1994, among others). These services are highly dependent on streambanks and flood plains being in a vegetated and relatively undisturbed state. Rooted streamside plants retard streambank erosion, filter sediments out of the water, build up and stabilize streambanks and streambeds, and provide shade, food, and nutrients for aquatic and riparian species (Winegar 1977, Thomas et al. 1979, Kauffman and Krueger 1984). The ability of undisturbed plant communities to stabilize banks was notable during extensive floods in eastern Oregon in 1996, when shrubby vegetation in ungrazed sections of the Deschutes River "broke the flood's velocity and combed logs and mud from the river" (Meehan 1996).

Healthy riparian areas also act as giant sponges during flood events, raising water tables and maintaining a source of streamwater during dry seasons. The result is a more stable streamflow throughout the year (US-GAO 1988).

Cattle cause more damage to riparian zones than their often small numbers would suggest. Cattle tend to avoid hot, dry environments and congregate in wet areas for water and forage, which is more succulent and abundant than in uplands. They are also attracted to the shade and lower temperatures near streams, most likely because their species evolved in cool, wet meadows of northern Europe and Asia. In fact, cattle spend 5-30 times as much time in these cool, productive zones than would be predicted from surface area alone (Roath and Krueger 1982, Skovlin 1984). One study found that a riparian zone in eastern Oregon comprised only 1.9% of the grazing allotment by area, but produced 21% of the available forage and 81% of forage consumed by cattle (Roath and Krueger 1982).

Our goal is to summarize along biological and ecological lines the major effects of cattle grazing in stream and riparian ecosystems. We include only those studies that discuss the direct and indirect effects of livestock activities on stream and riparian habitats. We exclude other aspects of livestock production such as conversion of flood plains to cultivated fields for livestock feed, leaching of fertilizer from these fields into streams, and streamwater diversion for crop or pasture irrigation. We also do not include the effects of impounding streamwater for stock ponds or other activities that support livestock production, although these activities contribute significantly to stream degradation.

Methods

We searched the scientific literature for peer-reviewed empirical papers and reviews of the biological and physical effects of livestock on western rivers, streams, and associated riparian areas. Because of the extensive literature on the subject, not all papers could be reviewed or cited. In choosing the papers to be included, we gave highest priority to recent papers in refereed journals presenting experimental manipulations such as paired samples from grazed vs. ungrazed areas or from heavily grazed vs. lightly grazed pastures (when ungrazed controls were not included in the experimental design). Many of these studies used sites recently protected from grazing as controls (e.g., Kauffman at al. 1983a, Schulz and Leininger 1990), but a few used previously ungrazed areas to which livestock were newly introduced (e.g., Sedgwick and Knopf 1987, Samson et al. 1988). Secondary priority was given to descriptive or comparative studies of grazed vs. naturally or historically protected areas where similarity of initial conditions could be inferred. Where there was a paucity of data, we also used non-peer-reviewed reports, usually from government documents or symposia. In no case were our general conclusions drawn from unrefereed reports or from studies showing anomalous results. Instead, we based our conclusions on what seemed to be the consensus of experts in the field.

We also identified and listed comprehensive review papers on each topic. Environmental impacts were defined as environmental changes that were significant at the $P < 0.1$ level (e.g., Peterman 1990) (discussed below) or those effects deemed significant by the authors.

Results

Damage caused by cattle to riparian and stream habitats in the arid and semi-arid West can be separated into two broad categories: impacts that occur at the local level (Table 1) and those that occur at landscape and regional levels (Table 2). Local impacts can be further segregated by their effects on water quality and seasonal quantity, stream channel morphology, hydrology, riparian-zone soils, instream and streambank vegetation, aquatic biota, and terrestrial wildlife (Table 1). Local impacts have been investigated in a large number of studies, but landscape-level impacts have received less attention.

Our search uncovered no systematic investigations showing positive impacts or ecological benefits that could be attributed to livestock activities when grazed areas were compared to protected areas (see also Bock et al. 1993, Ohmart 1996). Thus, we mostly present negative environmental impacts. In general, there was little debate about the effects of livestock grazing. Most authors tended to agree that livestock damage stream and riparian ecosystems.

Discussion

In the following, we discuss pertinent topics that have not been addressed in depth in recent reviews. These reviews, which are listed after each major category in Tables 1 and 2, should be consulted for additional discussion of other topics.

Positive and neutral effects of cattle grazing on riparian zones. An extensive literature search did not locate peer-reviewed, empirical papers reporting a positive impact of cattle on riparian areas when those areas were compared to ungrazed controls, but some studies reported no statistically significant effects due to riparian grazing (e.g., Buckhouse and Gifford 1976, Samson et al. 1988). The authors of these papers usually explained this absence of statistically significant impacts as being due to stochastic or design problems associated with their research, rather than to grazing having no effect on vegetation, fish, or stream hydrology. They described such problems as (1) high variability among treatment plots, which masked treatment effects (e.g., Tiedemann and Higgins 1989, Shaw 1992), (2) insufficient recovery periods after protection from grazing (e.g., Hubert et al. 1985, Sedgwick and Knopf 1991, Shaw 1992, Sarr et al. 1996), (3) heavy browsing and grazing by native herbivores (or trespassing cattle) on supposedly ungrazed control plots (e.g., Shaw 1992, Clary et al. 1996), (4) unplanned disturbances such as flooding (e.g., Sedgwick and Knopf 1991, Clary et al. 1996, Myers and Swanson 1996a), and (5) the unknown effects of a prior history of heavy grazing, which may have permanently altered stream function and prevented recovery of control plots (e.g., Tiedemann and Higgins 1989).

The absence of significant effects may also be due to investigators setting statistical significance at arbitrarily low levels (i.e., at $P<0.05$). Peterman (1990) argues that many studies, such as those with few treatment replications or high spatial variability, have low power (i.e. poor ability) to detect environmental change. Because of the possibility that already depleted fish stocks could become endangered or important habitats become permanently altered, he argues that higher probability levels (i.e., $P<0.1$) are appropriate to test significance of hypotheses.

Authors have also attributed non-significant results to supplemental feeding of livestock (e.g., Sedgwick and Knopf 1991), which resulted in lower forage consumption levels than originally prescribed, and to high recreational fishing, which obscured the negative effects of grazing on fish populations (e.g., Hubert et al. 1985). Finally, severe environmental damage such as loss of native species or channel downcutting cannot be reversed in just a few years of protection. Streams may recover slowly or only over geological time scales (Sarr et al. 1996). Together, these circumstances have caused some (e.g., Platts 1982) to question the ability of many experimental techniques to adequately assess livestock impacts. Others (e.g. Peterman 1990) also question the statistical power of many experiments to accept or reject hypotheses.

Several recent papers (e.g., Clary and Webster 1989, Elmore and Kauffman 1994, Burton and Kozel 1996, Weller 1996) describe the benefits of reduced cattle stocking rates and newer grazing systems, such as seasonal grazing, rest-rotation, and deferred grazing. The authors also discuss examples of grazed riparian zones regaining their herbaceous and woody cover and water quality. These studies, however, only contrasted newer grazing systems with more traditional and destructive systems, such as year-long grazing and high stocking rates. They did not contrast these systems with no-grazing. The only conclusion that could be fairly drawn from these studies is that newer grazing systems improve streamside conditions relative to other grazing systems, not that cattle grazing truly benefit riparian zones. In fact, Meehan and Platts (1978) and Platts and Wagstaff (1984) found no grazing system that was compatible with healthy aquatic ecosystems.

In mid-western prairies, livestock have been reported to be useful at breaking up dense, rank vegetation near wetlands (Weller 1996). However, in the Intermountain West, where low densities of native grazers provided only light grazing and trampling disturbances during the last 10,000 years, riparian species have inherently lower tolerances for livestock disturbances (Mack and Thompson 1982). It is doubtful that grazing or trampling by cattle in this region would do more good than harm.

Problems in drawing generalizations from riparian studies. Although most research has shown grazing in streams and riparian zones to be deleterious, results have been variable (Platts 1982, Trimble and Mendel 1995). This has caused riparian specialists problems in drawing broad generalizations about the effects of cattle grazing. These problems can be attributed to several issues:

1. *Inadequacy of study design.* Most watershed-scale riparian management plans were not designed as experiments with the idea of researchers evaluating them years later.

2. *Inherent variability found between and within watersheds*. Streams are unique, having their own combination of channel morphology, soils, climate, riparian species, geology, and hydrology (Elmore and Beschta 1987, Myers and Swanson 1991, Trimble and Mendel 1995). One management strategy may have a particular effect in one area, but a greater or lesser effect elsewhere.

3. *Insufficient study replication*. Lack of adequate replication of experimental treatments make data interpretation difficult (Matthews 1996).

4. *Ambiguities or differences in study design* (Platts 1982, Rinne 1985). In some cases, terms such as "heavy" and "light" grazing to describe grazing treatments are subjective, making comparisons within and between experimental studies difficult (Fleischner 1994, Trimble and Mendel 1995). In other cases, differences in research methodologies make comparisons unreliable (Trimble and Mendel 1995).

5. *Grazing inside exclosures by small mammals and invertebrates*. Small animals often congregate inside exclosures where food and cover are abundant. Increases in grazing inside exclosures by grasshoppers, rabbits, and rodents may reduce differences between treatments, thus masking the effects of cattle grazing outside the exclosure.

6. *Prior grazing history*. Many pastures now protected within exclosures were grazed at some time in the past and thus do not accurately fall within a truly ungrazed (i.e. "pristine") landscape. In fact, many older exclosures were purposely erected in severely overgrazed and eroded areas in order for investigators to monitor recovery and successional processes. Since many of these protected stream segments may have been deeply downcut previously, their recovery may take hundreds to thousands of years. These exclosure studies, therefore, may underestimate the true extent of livestock damage because they fail to take into account the damage that occurred before the exclosures were erected (Fleischner 1994).

7. *Variable time lags.* Recovery of different ecological, hydrological, and geomorphologic processes require different amounts of time, often longer than the average research grant and sometimes longer than the life-span of the researcher. Recovery of herbaceous and woody vegetation along stream sides may begin immediately after grazing is terminated, while the recovery of channel form may take hundreds of years (Kattelmann and Embury 1996, Trimble and Mendel 1995, Clary et al. 1996).

8. *Influences from outside the study area*. Stream channel morphology and aquatic organisms respond not only to factors occurring inside the study area, but to those occurring outside as well (Rinne 1985). Soil compaction and reduced infiltration of rainwater due to cattle trampling on slopes above riparian exclosures may increase the volume of water flowing over soil surfaces and into protected research sites. In addition, grazed streambanks upstream from exclosures may fail, releasing sediments into protected segments. Together, these factors may contribute large amounts of sediment to the stream system, inhibiting stream recovery (Kondolf 1993). Similarly, water flowing out of exclosures may be cleaner, cooler, and produce better spawning habitat downstream than that inside the exclosures (Duff 1977, Rinne 1985). Conditions over the larger landscape, therefore, minimize differences in grazed/ungrazed comparative studies.

In spite of numerous problems in experimental design and difficulties in interpreting earlier studies, Platts (1982) concluded that livestock grazing was the major cause of degraded stream and riparian environments and reduced fish populations throughout the arid West. In an extensive review of the literature, he found that 85% of the studies demonstrated that livestock negatively impacted riparian and stream ecosystems, which he concluded was a sufficiently powerful statistic to override inadequacies in individual experimental design.

Effects of riparian grazing on channel morphology and water tables. Plants on undisturbed uplands and streamsides slow the downhill flow of rainwater, promoting its infiltration into soils. Water that percolates into the ground moves downhill through the sub-soil and seeps into stream channels throughout the year, creating perennial flows. But as upland and riparian vegetation is removed by livestock and as hillsides and streambanks are compacted by their hooves, less rainwater enters the soil and more flows overland into streams, creating larger peak flows. This was illustrated in a simulation by Trimble and Mendel (1995), who estimated that peak storm runoff from a 120 ha basin in Arizona would be 2-3 times greater when "heavily" grazed than when "lightly" grazed. Moderate and high rainfall events in grazed sites are, therefore, more likely to result in high energy and erosive floods, which deepen and reshape stream channels (Fig. 1, USDI 1994a).

Where streams flow over deep soils or unconsolidated substrates, the erosive energy of floods cause channel downcutting, or incision (Fig. 1). As the channel deepens, water drains from the flood plain into the channel, causing a lowering (subsidence) of the water table. The roots of riparian plants are left suspended in drier soils. Eventually, riparian plants and their associated wildlife species are replaced by upland species such as sagebrush (*Artemesia* spp.) and juniper (*Juniperus* spp.), which can tolerate these drier soils. Additionally, with less water entering upslope and riparian soils, less is available to provide late-season flows. Consequently, the high intensity floods of the spring and early summer are often followed by low and no flow in late summer and fall.

Effects of riparian grazing on biodiversity. Most studies comparing grazed and protected riparian areas show that some plant and animal species decrease in abundance or productivity in grazed sites while other species increase. Plant species that commonly decline with livestock grazing are either damaged by removal of their photosynthetic and reproductive organs, or are unable to tolerate trampling or the drier conditions caused by lowered water tables. Plant species that commonly increase with livestock grazing are usually weedy exotics that benefit from disturbed conditions, upland species that prefer the drier conditions created by grazing, or sub-dominant species that are released from competition when taller neighbors are grazed down (Kauffman and Krueger 1984, Schulz and Leininger 1991, Stacy 1995, Green and Kauffman 1995, Ohmart 1996, Sarr et al. 1996).

Neotropical migratory birds (Bock et al. 1993, Saab et al. 1995) and prairie waterbirds (Weller 1996) are also variously affected by livestock grazing. After reviewing a large number of relevant studies, Saab et al. (1995) concluded that livestock grazing in the West led to a decline in abundance of 46% of the 68 neotropical migrant landbirds that utilize riparian habitat, an increase in 29% of the migrants, and no clear response in 25%. Those species that are grounded nesting or forage in riparian areas with heavy shrub or ground cover tended to decrease in abundance with grazing, while species that prefer open habitats, are ground foragers, or are attracted to livestock (i.e., cowbirds (*Molothrus* spp.), tended to increase in abundance in grazed riparian habitats (Bock et al. 1993, Saab et al. 1995). Cavity and canopy nesters were least affected. After a thorough analysis, Bock et al. (1993) concluded that few neotropical bird species actually "benefited from [cattle] grazing in riparian habitats, and that those that do are not restricted to riparian communities" (p.302). In other words, species that benefit from grazing are already widely distributed over the landscape and gain no extra benefit from additional habitat. Conversely, those species that are harmed by grazing are usually restricted to riparian habitats. Riparian grazing, therefore, makes them vulnerable to local extinction.

Fish populations are also differentially affected by livestock grazing. As stream waters become warmer and more sediment-laden due to streamside grazing (Table 1), trout, salmon, and other cold-water species decline in number and biomass. They are often replaced by less valued and more tolerant species. For example, Stuber (1985) found a higher biomass of game fish (predominantly brown trout (*Salmo trutta*)) in protected stream segments in Colorado, but a higher biomass of non-game species (predominantly longnose sucker (*Catostomys catastomus*)) in grazed segments. Similarly, Marcuson (1977) found that trout (*Salmo* spp.) were more abundant in an ungrazed stream segment in the Beartooth Mountains while mountain whitefish (*Prosopium williamsoni*) were more abundant in a grazed segment.

Changes in species composition due to cattle grazing should not be evaluated in conventional species-diversity terms, since even an influx of exotic weeds will increase species richness and diversity. These weeds may increase diversity, but they also alter wildlife habitat and ecosystem processes (i.e. erosion rates, seasonal flows) to which native species are adapted. Of greater importance than species diversity is whether grazing reduces the abundance or diversity of native species and riparian specialists, and whether these species are being replaced by introduced or upland species. In both cases, such changes lead to a reduction in native biological diversity, homogenization of the biotic landscape, and loss of high-value wildlife (i.e. game) species (Stuber 1985, Bock et al. 1993). Reductions in number, size, and productivity of native riparian or aquatic species are nearly always viewed as negative or as representing declining ecosystem health (Ohmart 1996).

Cattle grazing has converted many of the riparian habitats in the arid West into communities dominated by habitat generalists and weedy species such as dandelions (*Taraxacum officionale*), cheatgrass (*Bromus tectorum*), cowbirds, and small-mouth bass (*Micropterus dolomieu*), and by upland or abundant species such as sagebrush, juniper, and speckled dace (*Rhinichthys osculus*). As a result, both habitat quality and native species diversity have been severely reduced (Marcuson 1977, US-GAO 1988, Armour et al. 1994, Popolizia et al. 1994, Green and Kauffman 1995, Sarr et al. 1996). Consequently, a recent Forest Service report found livestock grazing to be the fourth major cause of species endangerment in the United States and the second major cause of endangerment of plant species (Flather et al. 1994). Within certain regions (i.e. Arizona Basin and Colorado/Green River Plateau), livestock grazing was listed as the #1 cause of species being federally listed as threatened or endangered.

Effects of riparian grazing on water quality. Bacterial contamination of drinking and surface water by domestic livestock is a significant non-point source of water pollution (George 1996). Although usually not considered pathogenic (Gary et al. 1983), fecal coliform (e.g., *Escherichia coli*), and enterococci bacteria are regularly monitored in surface waters because they are indicators of fecal contamination that may include pathogenic organisms such as *Cryptosporidium, Giardia*, *Salmonella*, *Shigella* and enteric viruses (Bohn and Buckhouse 1985b, George 1996). Because these organisms are carried by cattle and because fecal bacteria levels tend to increase with increasing grazing pressure (Gary et al. 1983, Owens et al. 1989, George 1996), the probability of disease-causing organisms contaminating swimming areas and entering human water supplies increases with intensity of cattle use.

Another concern is that nutrients found in animal wastes stimulate algal and aquatic plant growth when they are deposited directly or washed into streamwater. If resulting plant growth is moderate, it may provide a food base for the aquatic community. If excessive, these nutrients stimulate algal blooms. Subsequent decomposition of the algae leads to low dissolved oxygen concentrations (US-EPA 1995), which endangers aquatic organisms.

Landscape and regional effects of riparian grazing. The impacts of grazing on local riparian and stream environments and on stream morphology may be acute, but they also often extend far beyond their immediate surroundings (Table 2). Streams connect uplands to lowlands, terrestrial ecosystems to aquatic, and arid ecosystems to moist (Gregory et al. 1991, Knopf and Samson 1994). They act as corridors for migrating animals, provide moisture for aquatic, riparian, and upland species, and distribute sediments and nutrients downstream (Table 2; Thomas et al. 1979, Lee et al. 1989). In the case of anadromous fish, nutrients that are consumed in the ocean are brought inland, where they are distributed throughout the landscape as the fish are consumed by predators or decompose along streambanks after spawning.

By degrading water supplies and reducing the area of healthy riparian habitat, livestock fragment these landscape-level connections. They also damage the connection between natural and human communities, since degraded streams reduce the potential for recreational fishing and swimming, degrade municipal water supplies, provide less water for reservoirs, and damage coastal commercial fishing.

Neither are streams isolated from their adjacent uplands. Heavy grazing on upland communities impacts riparian areas primarily by increasing runoff and erosion. Blackburn (1984) and Trimble and Mendel (1995) summarized the negative impacts of heavy grazing on watersheds. They listed the erosive force of raindrops on denuded surfaces, the shearing force of hooves on slopes, decreased soil organic matter, and increased soil compaction as primary impacts. Together, these lead to reduced water infiltration and increased runoff, soil bulk density, erosion, and sediment delivery to streams. In addition, cattle form trails and terracettes (Trimble and Mendel 1995) (also called bovine terraces), which are also subject to erosion (Rostagno 1989).

Other factors contributing to riparian degradation. Cattle grazing is not the only factor damaging stream and riparian habitats in the arid West. Urban development, mining, damming for hydroelectric power, road construction, local eradication of beaver, logging, agricultural activities, and water diversions for industry, irrigation, and municipal water supplies have also exacted heavy tolls on riparian and aquatic ecosystems (Skovlin 1984, Szaro 1989, USDI 1994b). These factors acting alone and in combination have caused devastating cumulative impacts on western streams (Lee et al. 1989). Despite this, livestock grazing is still considered to be the most pervasive source of upland and riparian habitat degradation in the arid West (Elmore and Kauffman 1994, USDI 1994a, Ohmart 1996, among others).

Effects of riparian grazing in humid environments. Most investigations of the effects of livestock grazing on streams, rivers, and riparian zones have been located in arid regions. Although empirical studies from more humid (mesic) regions, such as western Oregon and Washington, the mid-West, and the eastern United States, are not as numerous (Trimble and Mendel 1995), available evidence suggests that environmental impacts of grazing in these regions are similar to those in drier areas. In all environments, cattle consume streamside vegetation, disturb soils, destabilize streambanks, deposit manure and urine, and churn up channel sediments (Trimble 1994, Armour et al. 1994, Trimble and Mendel 1995). Similar to arid areas, cattle were found to reduce overhead cover, herbaceous cover on banks, and woody vegetation in western Washington and Wisconsin (Chapman and Knudsen 1980, White and Brynildson 1967). Livestock also increased concentrations of ammonia, nitrate, soluble phosphate, chemical oxygen demand, and total organic carbon in runoff in Nebraska (Schepers and Francis 1982), increased concentrations of organic nitrogen, organic carbon, and sediment in runoff in Ohio (Owens et al. 1989, 1996), caused streambank erosion in Pennsylvania (Davis et al. 1991) and Tennessee (Trimble 1994), and increased soil loss in North Dakota (Hofmann and Ries 1991).

In some cases grazing may be even more damaging in wetter than in drier environments because moist soils are more vulnerable to compaction and disturbance than dry soils (Marlow and Pogacnik 1985, Trimble and Mendel 1995, McInnis 1996). In other cases, damage to riparian and stream habitats may be less severe in wetter climates because cattle may be less attracted to streamsides in areas where upland grasses are green and palatable for more months of the year.

Conclusion

The current debate over the environmental impacts and suitability of livestock grazing in arid western ecosystems has resulted in supporters declaring that livestock sometimes benefit streams (Savory 1988). Nearly all scientific studies, both observational and experimental, refute this claim. Livestock do not benefit stream and riparian communities, water quality, or hydrologic function in any way (Table 1). However, their damage can be reduced by improving grazing methods, herding or fencing cattle away from streams, reducing livestock numbers, or increasing the period of rest from grazing (Armour et al. 1994, Elmore and Kauffman 1994). The conclusion that all grazing practices detrimentally affect riparian areas (Elmore and Kauffman (1994) is to be expected since traditional grazing systems were developed for protecting upland grasses, not for protecting riparian plants and streamsbanks (Platts 1991, Saab et al. 1995).

With improved livestock management, previously denuded streambanks may revegetate and erosion may decline (Elmore and Kauffman 1994), but recovery will take longer than if grazing were terminated completely (Myers and Swanson 1995, 1996a, Ohmart 1996). Trimble and Mendel (1995) concluded that "although there may have been improvements in grazing management, the increase of cattle in the West [a doubling over the last 50 years] suggest that grazing impacts will continue into the foreseeable future" (p 233).

New studies go even further by suggesting that new grazing systems have only served to slow the rate of degradation, not reverse it. Sarr et al. (1996), for example, found that ten full years of livestock exclusion was necessary to reverse a negative trend and allow stream conditions to begin to improve. Elmore and Kauffman (1994) best summed up available evidence by stating that "livestock exclusion has consistently resulted in the most dramatic and rapid rates of ecosystem recovery " (p. 216).

Although the possibility of streams recovering their plant cover and ecological functions while providing food and water for livestock use is appealing (i.e. a win-win situation), it is largely contradicted by existing evidence (Table 1). Riparian specialist Robert Ohmart of the University of Arizona questions whether weakened and degraded riparian communities throughout the arid West can "hang onto their thread of existence for another 30-50 years" (Ohmart 1996, p. 272) while waiting for grazed systems to recover.

All discussions of improved grazing systems allude to the fact that the best prescription for stream recovery is a long period of rest from livestock grazing. Even those who strongly believe grazing to be compatible with healthy riparian ecosystems point out that 2-15 years of total livestock exclusion is required to initiate the recovery process (Duff 1977, Skovlin 1984, Clary and Webster 1989, Elmore 1996, Clary et al. 1996). Consequently, streams that are permanently protected from grazing have the highest probability of successful recovery (Claire and Storch 1977, Chaney et al. 1990, Bock et al. 1993, Armour et al. 1994, Fleischner 1994, Rhodes et al. 1994, Ohmart 1996, Case and Kauffman 1997).

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Table 1: Effects of livestock grazing and trampling on aquatic and riparian species and habitats in the western United States.

Influence on	Response	Causes	Impacts	References
Water quality				
Nutrient concentrations	Increase	Runoff from disturbed stream banks; livestock urine and manure deposited into stream; nutrients concentrated in reduced quantity of water	Reduced dissolved oxygen and possible water salinization in isolated pools and downstream lakes; alteration of instream species composition	Schepers et al. 1982 Taylor et al. 1989
Bacteria/ protozoa	Increase	Direct fecal deposition into water; fecal material in runoff; sediments containing buried microorganisms churned up by hoof action	Higher human and wildlife disease-producing potential from pathogens; human health endangered by swimming and other contact	Johnson et al. 1978 Stephenson and Street 1978 Stephenson and Rychert 1982 Tiedemann and Higgins 1989 Tiedemann et al. 1987
Sediment load and turbidity	Increase	Instream trampling; disturbance and erosion from denuded banks; reduced sediment trapping by streambank and instream vegetation; loss of bank stability; increased peak flows from compaction	Sediments blanket spawning gravel, entombing or suffocating fish embryos and juveniles; reduced dissolved oxygen levels in substrate; reduced foraging success by aquatic organisms; disruption of fish migration and respiratory systems of invertebrates; pool infilling; alteraton of benthic food web; reduction of human reservoir storage capacity more rapidly than projected; increased costs for filtraton of domestic water supplies	Lusby 1970 Winegar 1977 Johnson et al. 1978 Stevens et al. 1992

Water temperature	Increases	Increased solar exposure due to reduced shade from streamside vegetation and loss of undercut streambanks, and to widened stream channel that exposes greater water surface to solar radiation; lower summer flow	Increased evaporation and salinity; poor to lethal environment for salmonids and other temperature-sensitive, cold-water species; reduces fish growth due to increased metabolic rate and suppression in appetite; increased competition from warmwater fish; shift from salmonids to non-game fish; increased predation on fish; changes in growth rate and population size of cyanobacteria, algae and other aquatic organisms; increased incidence of lethal water-borne diseases; higher decomposition rates	Duff 1977 Tiedemann and Higgens 1989 Platts 1991 Li, et al. 1994 Tait, et al. 1994 Maloney et al. 1998
Dissolved oxygen levels	Possibly decline	Higher water temperatures; high biological oxygen demand of fecal material and algal blooms	Insufficient oxygen in spawning gravels; reduced rate of food consumption, growth and survival of salmonids and other aquatic species, especially at their early life stages; reduced prey items for fish; reduced decomposition rates; increased toxicity of toxicants	
General reviews of topic	Meehan and Platts 1978, Reiser and Bjornin 1979, Kauffman and Krueger 1984, Skovlin 1984, Bohn and Buckhouse 1985b, Haveren et al. 1985, Ongerth and Stibbs 1987, Platts 1991, USDI 1993, Rhodes et al. 1994, ODEQ 1995a, b, US-EPA 1995, Atwill 1996, Ohmart 1996			

Stream channel morphology				
Channel depth	Increases	Downcutting (incision) due to higher flood energy in high gradient, erosional stream regimes	Lowered groundwater table; narrowing of riparian zone; high flows contained within channel, thus precluding build-up of flood plain; more downstream sedimentation	Winegar 1977 Overton et al. 1994 Knapp and Matthews 1996
Channel width	Increases	Breakdown of streambanks by trampling; increased erosion from greater flood velocity; erosion of stream banks due to reduced resistance from riparian vegetation	Further loss of riparian vegetation; higher water temperatures; decreased water depth	Duff 1977 Marcuson 1977 Platts 1981a Kauffman et al. 1983b Hubert et al. 1985 Stuber 1985 Overton et al. 1994 Matthews 1996
Channel stability during floods	Decreases	Bare streambanks and channel bed easily eroded	Widening of channel; loss of pools and meanders	Marcuson 1977
Water depth	Decreases (except during peak flow)	Wider stream bed	Higher water temperatures; reduced habitat for aquatic organisms	Platts 1981a Hubert et al. 1985 Stuber 1985 Matthews 1996
Channel bed				
--Gravel	Lost in erosional environment	Increased flood velocity and energy; reduction in large woody debris	Reduced spawning habitat and habitat for benthic organisms	Duff 1977
--Fine sediments	Increase in depositional environment	Increased streambank erosion	Suffocation of fish eggs and fry due to low intragravel oxygen levels; degraded stream habitat for benthic organisms; filling in of pools	Duff 1977 Hubert et al. 1985 Owens et al. 1996 Myers and Swanson 1996a,b

Streambank stability	Reduced	Fewer plant roots to anchor soil; less plant cover to protect soil surface from disturbance; shear force of trampling hooves	Increased streambank sloughing; increased erosion and water turbidity; increased channel width	Duff 1977 Gunderson 1968 Marcuson 1977 Platts 1981a Kauffman et al. 1983b Rinne 1985 Stuber 1985 Myers and Swanson 1991, 1992, 1996a Kleinfelder et al. 1992 Overton et al. 1994
Streambank angle	Laid back	Streambank sloughing; livestock trampling	Increased channel width; decreased water depth	Platts 1981a Myers and Swanson 1995 Knapp and Matthews 1996
Streambank undercuts	Reduction in quality and quantity	Streambank breakdown by livestock and loss of stabilizing vegetation	Fewer hiding spaces and pools for fish	Platts 1981a Kauffman et al. 1983b Hubert et al. 1985 Overton et al. 1994 Myers and Swanson 1995 Knapp and Matthews 1996
Channel form	Fewer meanders and unvegetated gravel bars	Increased water velocity; removal of stabilizing vegetation; erosion of stream bank	Increased erosion; fewer pools for fish; decreased streambank roughness	Marcuson 1977
Pools	Decrease in number and quality	Loss of large woody debris; increased sedimentation	Loss of fish habitat; loss of thermal refugia during temperature extremes, reduced salmonid productivity and survival	Duff 1977 Marcuson 1977 Hubert et al. 1985 Myers and Swanson 1991, 1994, 1996a McIntosh et al. 1994
General reviews of topic	Kauffman and Krueger 1984, Skovlin 1984, Armour et al. 1994, Platts 1982, 1991, Chaney et al. 1990, 1993, USDI 1993; Rhodes et al. 1994; Trimble and Mendel 1995; Sarr et al. 1996			

Hydrology (stream flow patterns)				
Overland flow (runoff)	Increases	Reduced water infiltration into soils due to compaction and loss of streamside vegetation	Increase in sheet and rill erosion; increased flooding; reduced groundwater recharge; lowered water table	Orr 1975 Meehan and Platts 1978 Stevens et al. 1992
Peak flow	Increases	Larger volume of runoff flowing directly into channel	Increased stream energy for channel erosion, downcutting of channel bed and gully formation	Platts 1991
Flood water velocity	Increases	Reduced resistence from streambank and instream vegetation and from downed woody debris; increased flood water volume	Increased erosive energy and downcutting; removal of submerged vegetation and woody debris for pool formation; reduced habitat diversity; fish vulnerable to flash floods	Platts 1981a Li et al. 1994
Summer and late-season flows	Decrease	Less water stored in soil; lowered water table	Aquatic organisms stressed by reduced water quantity; less aquatic habitat; higher water temperatures	Ponce and Lindquist 1990 Kovalchik and Elmore 1992 Li et al. 1994
Water table	Lowered	Reduced water infiltration and increased runoff; groundwater drains into incised streambed; deeper channel reduces recharge by stream	Loss of aquatic and riparian species; perennial streams become ephemeral; loss of ephemeral streams	Kovalchik and Elmore 1992 Li et al. 1994
General reviews	Platts 1981b, 1991, Thurow 1991, Chaney et al. 1990, 1993, USDI 1993, Fleischner 1994, Rhodes et al. 1994, Trimble and Mendel 1995			

Riparian zone soils				
Bare ground	Increases	Vegetation consumed and trampled by livestock	Drier soil surfaces; higher erosion and sediment delivery to streams and aquatic habitats	Lusby 1970 Marcuson 1977 Hubert et al. 1985 Schultz and Leininger 1990 Clary and Medin 1990 Stevens et al. 1992 Popolizia et al. 1994
Erosion (water, ice, and wind)	Increases	Soil compaction; removal of vegetational cover; trampling disturbance	Increased sediment load to receiving stream; loss of fertile topsoil; suffication of fish eggs; loss of pools and pool volume; reduction of reservoir capacity	Lusby 1970 Bohn and Buckhouse 1985a Kauffman et al. 1983b
Litter layer	Decreases	Removal of aboveground plant biomass by livestock	Lower infiltration rates; greater runoff and erosion; reduced soil organic matter; warmer, drier soils	Marcuson 1977 Kauffman et al. 1983a Shultz and Leininger 1990 Popolizia et al. 1994 Green and Kauffman 1995
Compaction	Increases	Trampling by livestock on wet, heavy soils; reduced litter and soil organic matter	Decreased infiltration rates and more runoff; reduced plant productivity and vegetative cover	Orr 1975 Clary and Medin 1990 Clary 1995
Infiltration	Decreases	Increased soil compaction from hoof action; reduced plant cover, litter, and organic matter	Increased overland flow and erosion; reduced soil water content and plant growth; lowered water table	Orr 1975 Gifford and Hawkins 1978 Bohn and Buckhouse 1985a
Fertility	Declines	Less soil organic matter; loss of top soil; loss of soil strucure due to trampling	Fewer soil organisms; reduced plant growth	Marcuson 1977
General reviews of topic	Kauffman and Krueger 1984, Skovlin 1984, Chaney et al. 1990, 1993, Thurow 1991, Fleischner 1994, Rhodes et al. 1994; Trimble and Mendel 1995, Belsky and Blumenthal 1997			

Instream vegetation				
Algae	Increase	More sunlight; higher temperatures; higher concentrations of dissolved nutrients	Low levels of dissolved oxygen, especially when algal blooms collapse	Tait et al. 1994 Li et al. 1994 US-EPA 1995
Higher plants (submerged and emergent)	Often decline in abundance	Trampled; buried in deposited sediments; uprooted by strong flows	Reduced trapping of sediments; less food for aquatic organisms; higher water velocity and erosive force	
General reviews	Knight and Bottorff 1984			
Streambank vegetation				
Herbaceous cover, biomass, productivity, and native diversity	Decline	Grazing and trampling by livestock; selective grazing on palatable species; loss of vulnerable species; lowered water table; drier, warmer, more exposed environment	Less detritus (food inputs) for stream and aquatic organisms; higher water temperatures in summer and cooler temperatures in winter; degraded habitat for fish and wildlife; reduced biodiversity; loss of moisture- and shade-dependent species; replacement of riparian specialists with weedy generalists; loss of ecosystem resiliency; higher water velocities during floods; reduced sediment trapping	Duff 1977 Marcuson 1977 Winegar 1977 Kauffman, et al. 1983a Elmore and Beschta 1987 Medin and Clary 1989 Schultz and Leininger 1990 Clary and Medin 1990 Stevens, et al. 1992 Popolizia et al. 1994 Clary 1995 Green and Kauffman 1995 Clary et al. 1996 Knapp and Matthews 1996
Overhanging vegetation	Declines	Grazing and browsing by livestock	Less shade; higher water temperatures; less detritus for stream organisms	Marcuson 1977
Tree and shrub biomass and cover	Decline	Browsing by livestock on shrubs and tree saplings when they are most vulnerable	Decline in streambank stability; increased erosion; reduced stream shade and higher water temperatures; reduction in detritus and essential nutrients; loss of complex vegetation structure for wildlife	Marcuson 1977 Kauffman et al. 1983a Taylor 1986 Schulz and Leininger 1990 Sedgwick and Knopf 1991 Boggs and Weaver 1992 Kovalchik and Elmore 1992 Green and Kauffman 1995

Species composition	Altered	Lowered water table; warmer, drier environment; livestock selection of palatable species; compacted and disturbed soils	Replacement of riparian species by upland species and exotic weeds; reduction in riparian area	Kauffman et al. 1983a Clary and Medin 1990 Schulz and Leininger 1990 Green and Kauffman 1995
Structure (vertical and horizontal)	Simplified	Loss of trees and large shrubs; reduced plant establishment in drier soils	Loss of sensitive bird species; reduction in wildlife habitat	Taylor 1986 Knopf et al. 1988 Medin and Clary 1989
Plant age-structure	Becomes even-aged	Reduced plant establishment and survival due to browsing, grazing, and trampling	Reduced wildlife habitat; loss of riparian-dependent wildlife	Kauffman et al. 1983a
Plant phenology	Altered	Less shade and soil litter create warmer, drier environments in summer and colder environments in winter	Increased frost damage to plants in fall	Kauffman et al. 1983a
Plant succession	Impeded	Late-successional species grazed and browsed	Retrogression	Kauffman et al. 1983a Green and Kauffman 1995
General reviews of topic	Kauffman and Krueger 1984, Knight and Bartorff 1984, Skovlin 1984, Thomas et al. 1979, Chaney et al. 1990, 1993, Fleischner 1994, Ohmart 1996			

Aquatic and riparian wildlife				
Fish				
Species diversity, abundance, and productivity	Decrease	Higher water temperatures increase salmonid mortality (by breaking down physiological regulation of vital processes such as respiration and circulation), and negatively affect fish spawning, rearing, and passage; greater water turbidity, increased siltation and bacterial counts, lower summer flows, and low dissolved oxygen in the water column and intragravel environment reduce fish survival; damage to spawning beds; less protective plant cover; fewer insects and other food items; streambank damage; decreased hiding cover; reduced resistance to water-bourne diseases	Loss of salmonids and other cold-water species; loss of avian and mammalian predators; replacement of cold-water, riparian species with warm-water species	Duff 1977 Marcuson 1977 Stuber 1985 Li et al. 1994 Tait et al. 1994 Dudley and Embury 1995 Knapp and Matthews 1996 Sarr et al. 1996
Behavior	Different use of different habitats	Reduction in preferred habitat types		Matthews 1996
General reviews of topic	Marcuson 1977, Meehan et al. 1977, Reiser and Bjornn 1979, Kauffman and Krueger 1984, Skovlin 1984, Platts 1982, 1991, Fleischner 1994, Rhodes et al. 1994, ODEQ 1995a,b, Ohmart 1996			
Invertebrates				
Diversity, abundance, and species composition	Altered	Higher water temperatures from loss of shade; lower dissolved oxygen levels; increased fine sediments; reduced plant detritus but higher algal biomass for food	Loss of species that require cleaner and colder waters and coarser substrates; increase in algae feeders; fewer palatable species and less food for higher trophic levels; reduced litter breakdown	Rinne 1988 Tait et al. 1994 Erman 1996

General reviews	Meehan et al. 1977, Knight and Bottorff 1984, ODEQ 1995a,b, Sarr et al. (1996)			
Amphibians and reptiles				
Diversity, abundance, and species composition	Decline	Decline in structural richness of vegetative community; loss of prey base; increased aridity; loss of thermal cover and protection from predators; water temperatures lethal to early life stages	Loss of biodiversity and prey for higher trophic levels; loss of native species	Jones 1981 Szaro et al. 1985 Dudley and Embury 1995 Jennings 1996
Birds				
Diversity, abundance and species composition	Altered	Reduction in food, water quality and water quantity; loss of perches, nesting sites, and protective plant cover; loss of complex vegetational structure	Reduction in biodiversity; replacement of riparian specialists by upland species and generalists; loss of some neotropical migrants	Taylor 1986 Sedgwick and Knopf 1987 Knopf et al. 1988 Schulz and Leininger 1991 Clary and Medin 1992 Stacey 1995
General reviews of topic	Kauffman and Krueger 1984, Skovlin 1984, Bock et al. 1993, Fleischner 1994, ODEQ 1995a,b, Saab et al. 1995, Ohmart 1996, Weller 1996			
Mammals (large and small)				
Diversity, abundance, and species composition	Altered (sometimes but not always)	Loss of riparian habitat and food sources; warmer, drier, more exposed environment; behavioral characteristics such as avoidance of livestock	Habitat-use shifts by wildlife; suboptimal nutrition for females and offspring; changes in predator-prey relations; altered herbivory and other ecosystem processes; lower beaver activity with their creation of wetlands; riparian species replaced by upland species and generalists	Winegar 1977 Samson et al. 1988 Medin and Clary 1989 Loft et al. 1991 Schultz and Leininger 1991 Clary and Medin 1992 Clary et al. 1996
General reviews	Thomas et al. 1979, Kauffman and Krueger 1984, Skovlin 1984, Ohmart 1996			
Threatened and endangered species				

Abundance	Reduced	Loss of habitat; disturbance; livestock herbivory; competition with livestock; habitat fragmentation	Possible extinction	Dudley and Embury 1995 USDI 1994a
General reviews	Flather et al. 1994, Horning 1994, Ohmart 1996			

Table 2. Landscape and regional consequences of livestock grazing in streams and riparian ecosystems in the arid West

Downstream waters have higher temperatures and sediment loads
Downstream flood levels are higher
Quantity of water to downstream ecosystems is lower during low-flow periods
Forested connectors and wildlife migratory routes between high and low elevation ranges are lost
The diversity and abundance of migratory birds and wildlife are reduced
Habitat mosaic is homogenized
Corridors for migration of salmonids and other species are fragmented
Areas set aside for human recreation are reduced in quality
Commercial and recreational fishing opportunities are reduced
Domestic water supplies require more filtration and treatment by water-treatment plants, leading to higher utility rates
More sediment is deposited in lakes and reservoirs, thus reducing reservoir life and hydroelectric capacity
Sediments in water damage hydroelectric turbines
Higher sediment loads increase maintenance costs of irrigation canals

Off-Highway Vehicle Trail Impacts on Breeding Songbirds in Northeastern California

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Off-Highway Vehicle Trail Impacts on Breeding Songbirds in Northeastern California

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ABSTRACT Rapid growth in off-highway vehicle (OHV) use in North America leads to concerns about potential impacts on wildlife populations. We studied the relationship between distance to active OHV trail and songbird nesting success and abundance in northeastern California, USA, from 2002 to 2004. We found evidence of greater nest desertion and abandonment and reduced predation on shrub nests <100 m from OHV trails than at nests >100 m from OHV trails. Two of 18 species studied were less abundant at sites on trails than at sites 250 m from trails, and no species were more abundant on trails. Management of OHV trail development should consider possible negative impacts on nesting success and abundance of breeding birds. (JOURNAL OF WILDLIFE MANAGEMENT 71(5):1617–1620; 2007)

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Rapid growth in off-highway vehicle (OHV) use (Bowker et al. 1999) in North America leads to concerns about potential impacts on wildlife populations. The few previous studies on the impacts of OHV on wildlife populations or habitats compared abundance of plants or wildlife between used and unused areas on large spatial scales (e.g., Luckenbach and Bury 1983, Brooks 1999) or studied the impacts of beach OHV use on shorebirds (e.g., Buick and Paton 1989). These studies generally indicate that disturbance or habitat alteration associated with OHV use causes reduction in wildlife populations (Luckenbach and Bury 1983, Lovich and Bainbridge 1999).

Researchers could use results from more in-depth studies on roads and recreational (hiking) trails to predict the impacts of OHV trails on wildlife. Possible mechanisms of demographic change identified in studies of the effects of roads or recreational (hiking) trails on wildlife are mortality from collisions, altered movement patterns, altered reproductive success, and decreases in populations due to loss of habitat (Rich et al. 1994, Forman and Alexander 1998, Miller and Hobbs 1998, Forman et al. 2003). Off-highway vehicle trails could affect wildlife in similar ways. However, seasonal and temporal use patterns, types of vehicles used, and areas developed differ between OHV trails and roads or recreational trails. Therefore, results from studies of the effects of roads and recreational trails on wildlife can only be applied to OHV trails with caution. Direct study is required to understand the effects of OHV trails on wildlife.

We performed a 3-year study to assess effects of active OHV trails on songbird abundance and nesting success at a publicly developed and maintained OHV recreation area. We measured nesting success because it is an important deterministic component of seasonal fecundity (Jones et al. 2005), which in turn influences population growth rate (Saether and Bakke 2000). We present results on the effects of proximity to OHV trails for songbird nest predation, abandonment, and desertion rates. We also present count-based relative abundance indices of common songbird species in relation to proximity to OHV trails.

STUDY AREA

From April to July 2002–2004 we conducted research at the Fort Sage Mountains OHV Area, 3 km northeast of Doyle, California, USA, which is managed by the Bureau of Land Management. In 2002, we established a nest-monitoring plot near the primary OHV trailhead and campground and 70 point count stations in the OHV area. The OHV area consisted of 8,900 ha located at the northern end of the Fort Sage Mountains, and was characterized by a mosaic of vegetation types dominated by big sagebrush (*Artemisia tridentata*), bitterbrush (*Purshia tridentata*), western juniper (*Juniperus occidentalis*), green rabbitbrush (*Chrysothamnus viscidiflorus*), gray rabbitbrush (*C. nauseosus*), and greasewood (*Sarcobatus vermiculatus*). The OHV trail network included narrow (0.5 m) motorcycle trails to wide (3 m) dirt roads. The area received approximately 9,000 visitor-days of use per year during 2002–2004, with most use in March–May and September–November (D. Jackson, Bureau of Land Management Susanville Field Office, personal communication).

METHODS

We located and monitored bird nests to estimate rates of predation, abandonment, desertion, and success using methods described by Martin and Geupel (1993). We recorded locations of nests using a handheld Global Positioning System device and calculated the distance to the nearest OHV trail using the Nearest Features 3.8 extension (Jenness 2004) in ArcView Geographic Information System (GIS) 3.2a. We grouped nests by those close to

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(<100 m) and far from (>100 m) OHV trails to evaluate the effects of trail distance on nesting success because we were constrained by small sample sizes. Splitting nests into these 2 distance categories created 2 groups of approximately equal sizes.

We calculated daily abandonment rates, daily desertion rates, and daily predation rates for nests close to and far from active trails using the Mayfield method (Mayfield 1975) and standard errors following Johnson (1979). We excluded nests that were parasitized by brown-headed cowbirds (*Molothrus ater*) from analyses (n = 3). We defined abandonment as cessation of a nesting attempt prior to egg laying, and we defined desertion as cessation of a nesting attempt when the nest contained either eggs or young. To calculate predation rates, we pooled nests into ground and shrub open cup-nesting species because within these groups, predation rates tend to be similar (Martin 1993), thus reducing the problems created by pooling heterogeneous species. We used the chi-square test as implemented by Program CONTRAST (Hines and Sauer 1989) to test for an effect of distance to active OHV trail on Mayfield estimates of predation, abandonment, and desertion rates.

We used a GIS coverage of trail locations superimposed on a United States Geological Survey Digital Raster Graphic of the Fort Sage Mountains to select 35 on-trail stations and pair them with 35 off-trail stations with regards to slope and elevation. We located off-trail stations 250 m away from their paired on-trail station and we located all off-trail sites between 200 m and 250 m away from the nearest active OHV trail. We visited all paired sites to assess comparability of vegetation cover. In 2 cases we relocated point count station on- and off-trail pairs because of disparate habitat types. We estimated relative bird abundances at these locations using 100-m fixed-radius 5-minute point counts (Ralph et al. 1993). Thus, on-trail station surveys included individuals detected within 100 m of trails, and off-trail station surveys included individuals 100–350 m from trails. We conducted point count surveys at each station twice per year from 2002 to 2004 for a total of 6 visits to each of the 70 stations. All surveys occurred between 0530 hours and 0900 hours on 17 May–25 June.

We measured habitat variables at each station using a 100-m line-intercept transect to estimate shrub cover, and 20 0.1-m^2 sampling plots spaced evenly along the length of the 100-m transect to estimate coverage of grasses, forbs, and bare ground. In addition, we counted trees within a 100-m radius.

We used 2-tailed paired t-tests to compare the mean number of detections per year of 18 bird species detected more than once at both on- and off-trail points, and to compare on- and off-trail vegetation variables. We used mean number of detections per year because of low annual variation and small sample size. We examined the strength of relationships between bird species abundance and 8 vegetation variables (total shrub cover, big sagebrush cover, bitterbrush cover, combined saltbush [*Atriplex* spp.] and greasewood shrub cover, western juniper density, perennial grass cover, annual grass cover, and forb cover) using Spearman correlations. When bird species abundance varied significantly between on- and off-trail points we used analysis of covariance (ANCOVA) to test whether vegetation differences between on- and off-trail points explained variation in bird species abundance, or whether the trail result was independent of measured vegetation attributes. Except where noted previously, we conducted all analyses using Stata 8.0 (StataCorp, College Station, TX). We defined significance at $\alpha = 0.10$ in order to reduce the risk of making a type 2 error due to small sample sizes and associated limited power. Further, we discuss nonsignificant effects as suggestive of possible biological effects (Robinson and Wainer 2002).

RESULTS

We used pooled data from 113 nests of 20 small songbird species to calculate daily desertion rates, 38 nests of 12 species found during building for daily abandonment rates, and 105 nests of 16 open-cup nesting species for daily predation rates. Daily abandonment rate of nests was 4 times higher at nests <100 m from a trail than those >100 m, but this difference was not statistically significant ($\chi^2 = 2.38$, 1 df, $P = 0.12$; Table 1). We only observed desertion within 21 m of trails, and daily desertion rate was significantly greater at nests <100 m from trails than those >100 m from trails ($\chi^2 = 4.02$, 1 df, $P = 0.04$; Table 1). Daily predation rates were approximately 2 times higher in ground-nesting birds than in shrub-nesting birds ($\chi^2 = 3.304$, 1 df, $P = 0.069$; Table 1), and thus separating these categories for analysis of the effect of trails on predation rate is appropriate. Daily predation rate of shrub nests <100 m from trails was approximately half that of nests >100 m from trails, but this difference was not statistically significant ($\chi^2 = 2.62$, 1 df, $P = 0.11$; Table 1). Daily predation rate (Table 1) of ground nests was not different <100 m from trails than >100 m ($\chi^2 = 0.14$, 1 df, $P = 0.91$; Table 1).

Spotted towhee (*Pipilo erythrophthalmus*) and western scrub-jay (*Aphelocoma coerulescens*) were somewhat less abundant at on-trail points (Table 2). Bitterbrush cover differed between on- and off-trail stations ($P < 0.04$), and was weakly correlated with spotted towhee abundance ($r = 0.19$, $P < 0.05$). Big sagebrush cover did not differ significantly between on- and off-trail stations ($P = 0.16$), but was correlated with both spotted towhee ($r = 0.46$, $P < 0.05$) and western scrub-jay abundance ($r = 0.26$, $P < 0.05$). Spotted towhee abundance was 9.33% lower (90% CI: –2.04%, –16.62%) at on-trail sites after accounting for the effects of vegetation (main effect of trail in ANCOVA, $F_{1,32} = 4.62$, $P = 0.039$, partial $R^2 = 0.038$) in a model that included the effects of trail, site pair, sagebrush cover, and bitterbrush cover (ANCOVA, $F_{37,32} = 7.63$, $P < 0.001$). Western scrub-jay abundance was 45.13% lower (90% CI: –3.69%, –86.56%) at on-trail sites after accounting for the effects of vegetation (main effect of trail in ANCOVA, $F_{1,33} = 3.44$, $P = 0.073$, partial $R^2 = 0.069$) in a model that

Table 1. Mayfield daily nest abandonment, desertion, and predation rates by off-highway vehicle (OHV) trail distance at the Fort Sage OHV Area, Lassen County, California, USA, 2002–2004.[a]

	Nests <100 m from trails					Nests >100 m from trails						
	Daily failure rate			Trail distance (m)		Daily failure rate			Trail distance (m)			
Failure cause	Estimate	SE	n[b]	$\bar{x}$	SD	Estimate	SE	n[b]	$\bar{x}$	SD	χ^2	P
Abandonment	0.066	0.029	23	43.7	30.3	0.016	0.016	15	189.3	47.5	2.376	0.123
Desertion	0.006	0.003	62	37.1	27.9	0.000	0.000	51	204.4	64.0	4.024	0.045
Shrub predation	0.031	0.008	38	31.1	27.7	0.058	0.015	32	198.3	68.4	2.618	0.106
Ground predation	0.071	0.018	24	46.7	25.9	0.067	0.022	19	214.9	56.2	0.135	0.908

[a] χ^2 and P values are comparisons of daily rates close to and far from trails using Program CONTRAST (Hines and Sauer 1989).
[b] We pooled species to achieve sample sizes large enough for comparisons between distance categories.

included the effects of trail, site pair, and sagebrush cover (ANCOVA, $F_{36,33} = 7.63$, $P < 0.001$).

DISCUSSION

Our results suggest a positive effect of proximity to OHV trail on nest desertion and abandonment and a negative relationship of proximity to OHV trail on predation rates of nests built in shrubs (Table 1). These effects have opposite net effects on nesting success, making interpretation difficult. However, our results suggest that species that are prone to nest abandonment or desertion, or that do not renest after failure, will be negatively affected by OHV trails. Conversely, some shrub-nesting species may benefit from reduced predation rates. However, because we pooled species for analysis due to small sample sizes, we are unable to infer which species may benefit or suffer due to these particular effects.

We found a tendency for lower predation rates (Table 1) at shrub nests <100 m from trails, counter to the frequently predicted positive effect of habitat edges on predation rates (Paton 1994). The OHV traffic on these trails could have frightened away predators, thus causing reduced predator densities in proximity to OHV trails. This hypothesis is echoed in a study of the effect of recreational hiking trails on songbird nest predation, which found lower predation rates near trails, ostensibly due to a reduced number of predators near trails (Merkle 2002). Conversely, all 5 desertion events observed were very close (<21 m) to trails and abandonment rate was 4 times higher close to trails (Table 1). Because we did not study OHV traffic directly, we are unable to determine whether OHV trails or OHV traffic influenced abandonment, desertion, or predation.

We found a negative effect of OHV trail on the abundance of 2 of 18 species studied, but the effect of OHV trail accounted for little of this variation after controlling for the effect of vegetation among sites. A negative effect of trails on spotted towhee is consistent with another study of this species that shows it avoids recreational trails (Holmes and Geupel 2005). It is unclear from our results whether these 2 species were less abundant on trails

Table 2. Comparison of on- and off-trail abundance ($\bar{x}$ detections/station/yr) of the 18 most common species at 35 pairs of point count stations at the Fort Sage Off-Highway Vehicle Area, Lassen County, California, USA, 2002–2004.

Species		On-trail abundance		Off-trail abundance		
		$\bar{x}$	SE	$\bar{x}$	SE	P[a]
Spotted towhee	*Pipilo maculatus*	1.00	0.22	1.32	0.25	0.099
Black-throated sparrow	*Amphispiza bilineata*	0.99	0.13	1.10	0.15	0.530
Western meadowlark	*Sturnella neglecta*	0.92	0.12	0.86	0.11	0.576
Pinyon jay	*Gymnorhinus cyanocephalus*	0.17	0.08	1.14	0.96	0.324
Lark sparrow	*Chondestes grammacus*	0.63	0.12	0.53	0.12	0.485
Brewer's sparrow	*Spizella breweri*	0.68	0.18	0.37	0.11	0.122
Blue-gray gnatcatcher	*Polioptila caerulea*	0.46	0.11	0.41	0.11	0.678
Bewick's wren	*Thryomanes bewickii*	0.30	0.08	0.31	0.08	0.818
Sage sparrow	*Amphispiza belli*	0.30	0.10	0.26	0.09	0.676
Brown-headed cowbird	*Molothrus ater*	0.17	0.06	0.33	0.11	0.170
Ash-throated flycatcher	*Myiarchus cinerascens*	0.18	0.06	0.28	0.07	0.169
Mourning dove	*Zenaida macroura*	0.24	0.06	0.21	0.05	0.697
Horned lark	*Eremophila alpestris*	0.21	0.09	0.19	0.11	0.790
Western scrub-jay	*Aphelocoma californica*	0.11	0.04	0.20	0.06	0.059
Bushtit	*Psaltriparus minimus*	0.21	0.07	0.10	0.05	0.202
Chipping sparrow	*Spizella passerina*	0.16	0.06	0.12	0.05	0.600
Loggerhead shrike	*Lanius ludovicianus*	0.13	0.06	0.15	0.04	0.676
Gray flycatcher	*Empidonax wrightii*	0.15	0.06	0.11	0.05	0.635
Juniper titmouse	*Baeolophus ridgwayi*	0.08	0.03	0.08	0.04	1.000
Red-shafted flicker	*Colaptes auratus*	0.05	0.02	0.10	0.03	0.117

[a] P values from paired t-tests.

because they avoided trails, or because trails reduced the amount of available habitat. We were unable to quantify such effects because no nearby site comparable to the Fort Sage Mountains OHV area exists to serve as a proper control.

We suggest future studies focus on replication across multiple OHV areas and matched control sites. Larger sample sizes would provide greater power to detect hypothesized effects for which we provide only suggestive results. Future studies should consider that effects of OHV trails or OHV use on the abundance of songbirds might occur at scales >250 m (e.g., Brooks 1999). Approximately 24% of the 8,900-ha study area was within 100 m of an active OHV trail, which suggests that the observed local effects could scale to landscape-level effects. Determining the scale at which disturbances such as OHV trail development and use affect wildlife demography is critical to effective management. Additionally, selection of focal species for demographic research would reduce the analytic and inferential issues created by pooling nest data across species. However, if pooling is necessary, our results highlight the need to control for nest strata in analyses.

MANAGEMENT IMPLICATIONS

Management of OHV trail development should consider possible negative impacts on nesting success and abundance of breeding birds. Areas within 100 m of OHV trails may provide reduced-quality habitat to nesting songbirds, particularly for species that suffer significant losses of annual fecundity due to abandonment or desertion of individual breeding attempts. Limitation of OHV trail development in breeding areas of rare or endangered birds could minimize conflicts over land use between recreation and wildlife conservation.

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SEDIMENT GENERATION FROM FORESTRY OPERATIONS AND ASSOCIATED EFFECTS ON AQUATIC ECOSYSTEMS

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Abstract

Timber harvest operations have been shown to have many effects on adjacent watercourses and on the aquatic ecosystems they support. This may occur from introductions or loss of woody debris, loss of riparian vegetation, accelerated stream bank and bed erosion, the alteration of natural channel form and process, and the reduction of stream habitat diversity. However, the existing literature indicates one of the most insidious effects of logging is the elevation of sediment loads and increased sedimentation within the drainage basin.

Sediment generation from various forestry practices has been studied extensively in the past. Forestry practices which generate suspended sediments include all operations that disturb soil surfaces such as site preparations, clear-cutting, log skidding, yarding, slash burns, heavy equipment operation and road construction and maintenance. From these sources, construction, use and maintenance of logging roads located in close proximity to watercourses produce by far the highest levels of suspended sediment generation in streams.

Three aspects of logging road development and maintenance are known to elevate sediment loads in watercourses: 1) instream and near-stream construction operations; 2) reduction in retention time and associated increase in erosion in the drainage basin; and 3) mass soil movements and/or landslides associated with logging road design and placement.

This literature review examined the effects of increased sediment load and sedimentation on aquatic ecosystems emphasizing forestry operations that generate elevated sediment loads. The review included the effects of sediment on fish (behavioral, physiological and population effects) and the effects of sedimentation on fish habitats (including spawning, rearing, food production, summer and overwintering habitats). A habitat effects relationship was presented which related the concentration and duration of specific sediment exposure events to the alteration of fish habitats. This relationship allows for post-disturbance evaluation of the potential effects on fish habitat.

Many advances have been made in the design and construction of logging roads over the past three decades. Many of these advances have been made in the design of road access systems, while other advances have been made in the development of more sophisticated mitigation techniques. These advances are described and discussed in relation to minimizing sediment generation and subsequent sedimentation in aquatic environments.

Introduction

Forest harvesting practices can elicit a number of physical changes within a watershed. These changes can set up associated responses in a wide range of physical, chemical and biological processes and can substantially alter aquatic habitats and communities. Forestry related activities can influence: stream hydrologic regime by reducing the time between peak rainfall and peak stream discharge and consequently increasing the magnitude of peak seasonal flows; water quality parameters such as temperature and sediment load; and, stream geometry by increasing erosive forces, channel migration, width to depth ratio and altering stream form and process (Meehan 1991; Salo and Cundy 1987)).

Forestry-related activities are not always harmful to freshwater fish communities. Long term and intensive study of logging on fish communities of Carnation Creek, Vancouver Island, British

Columbia, revealed the annual growth of juvenile coho salmon (*Oncorhynchus kisutch*) increased in years immediately following logging which was likely in response to increases in air and water temperatures (Hartman and Scrivener 1990; Holtby 1988; Tschaplinski and Hartman 1983). However, forestry activities have been shown to have adverse effects on resident and migratory fish communities, often related to an increase in the delivery of sediments to streams.

Although frequently viewed as synonymous, suspended sediments and sedimentation are two discrete processes that affect aquatic communities in different ways. Sediment transport in watercourses occurs in three forms, which include wash load, suspended, load and bed load. The entry of sediments to watercourses from upland sources is a natural process; however, when the rate entering a watercourse exceeds the capacity of the watercourse to transport or assimilate the sediment, stress may occur to the aquatic community and the suitability and/or productivity of aquatic habitats may be altered. Excess sediment in rivers and streams has been identified as the largest and most pervasive water pollution problem faced by aquatic systems in North America (Sweeten 1995).

This paper provides a collection of information related to biological responses to sediment synthesized from the literature and attempts to uncover relationships which are present between elevated sediment episodes and biological response. For additional information pertaining to this topic or to obtain additional references, the reader is directed to Meehan 1991, Salo and Cundy 1987, Newcombe (1994), Newcombe and Jensen (1995), Kerr (1995), Waters (1995) and Anderson *et al.* (1996).

Sources of Elevated Sediment Loads

Anthropocentric Sources of Elevated Sediments

Erosion within streams is a natural process and is affected by parameters such as stream flow, channel structure and stability, streambed composition, and disturbance within the watershed such as fire, landslide or ice scour. The disturbance of lands accelerates erosion and increases the delivery of sediment to stream systems. Any activity that is undertaken within a watershed that disturbs land surfaces has the potential to increase sediment delivery to streams. Most of man's activities will increase erosion to some extent in forested watersheds. Man's activities which most frequently increase sediment loads in watercourses include agriculture, mining, forestry, urban development, and stream channel alteration such as dams and channelization, and instream construction associated with developments such as bridges, roads or pipeline and transmission crossings. This paper discusses the effects of sediment increase associated with episodic elevated sediment events in general, however, types of mitigation to minimize sediment load increases which are specific to the forestry industry are discussed in the Mitigation Measures section and sediment sources specific to forest harvest operations are discussed below.

Forestry Sediment Sources

Forestry operations frequently increase sediment delivery into streams. Logging operations have been shown to increase sediment production above natural sedimentation rates (Megahan and Kidd 1972). The activities which commonly result in increased sediment delivery include clear-cutting, skidding, yarding, site preparation for replanting and road construction, use and maintenance (Waters 1995).

Among forestry harvest activities, disturbance associated with logging road construction and operation produces the greatest sediment load increase (Waters 1995; Furniss *et al.* 1991; Cederholm *et al.* 1981). Roads associated with a jammer logging system in the Payette National Forest, Idaho, increased sediment production an average of approximately 750 times over the natural rate over a six-year period following construction (Megahan and Kidd 1972). The average erosion rate from roads on the jammer unit for 1.35 years preceding logging was 56.2 tons/mile^2/day and the average rate for 4.8 years following logging was 51.0 tons/mile^2/day (Megahan and Kidd 1972).

Effects of Elevated Sediment Loads

Sediment Effects on Fish

In response to changes in the environment, ecosystems often undergo changes in community composition and structure. Organisms respond to environmental change in order to avoid or minimize effects on fitness. If an organism can not compensate for a change in the environment and suffers a reduction in fitness, the environmental change is termed a stress (Brett 1958; Kohen and Bayne 1989). Therefore, a stress limits either the rate of resource acquisition or growth and reproduction so that fitness is reduced (Grime 1989).

Stress has been defined as “the sum of all the physiological responses by which an animal tries to maintain or re-establish a normal metabolism in the face of a physical or chemical force” (Selye 1950). Stress occurs when the homeostatic or stabilizing processes of the fish or organism are extended beyond the capabilities of the organism to compensate for the biotic or abiotic challenges. Anthropogenic inputs of sediments into stream and riverine environments can cause stress to aquatic systems and thereby, directly and indirectly impact upon fish behaviour and health. Increased concentrations of suspended sediments can have direct effects on fish behaviour, fish physiology and fish populations (Anderson *et al.* 1996).

Behavioural Effects

Changes in fish behaviour are some of the first effects evoked from increasing concentrations of suspended sediments. Behavioural changes are generally considered benign and transitory. They are easily reversed and do not exhibit a long-lasting impact (Newcombe 1994). Typical responses include an increased frequency of the cough reflex, avoidance of suspended sediments, reduction in feeding and temporary disruption of territoriality. The severity of the behavioural response is associated with the timing of disturbance, the level of stress (and associated energy cost) and the importance of the habitat that the fish may be being excluded from.

The avoidance of suspended sediment plumes is one of the first reactions. Bisson and Bilby (1982) observed this behaviour evoked in juvenile coho salmon at total suspended sediment (TSS) concentrations as low as 88 mg/L. Similar results were recorded by McLeay *et al.* (1987) who found that Arctic grayling (*Thymallus arcticus*) avoided concentrations greater than 100 mg/L.

Increased concentrations of suspended sediment have also been correlated with a reduction in feeding. Feeding rate may be a function of prey visibility. McLeay *et al.* (1987) states that Arctic grayling exposed to suspended sediment concentrations greater than 100 mg/L were slower to recognize the food and more frequently missed a food item when they attempted to eat it. Sigler *et al.* (1984) believes, however, that a reduced feeding is more complex than reduced ability to see prey items, as many fish species (especially benthic feeders) do not use sight to identify prey items, but still exhibit reduced levels of feeding in response to elevated sediment loads.

High concentration of suspended sediments has also been associated with the loss of territoriality and interruption of movements of salmonids. Berg and Northcote (1985) found that territorial behaviour was lost at concentrations exceeding 30 NTU. They indicate that territoriality is influential in the allocation of food and habitat resources. Disruption to territoriality can occur when turbidity limits the visual distance that individuals can see, and when the downstream drift of fish avoiding increased concentrations of suspended sediment disrupts existing territories.

Physiological Effects

Physiological changes can be measured in fish as a response to the increased stress of suspended sediments. The typical measured responses include impaired growth, histological changes to gill tissue, alterations in blood chemistry, and an overall decrease in health and resistance to parasitism and disease. The effects of sediment exposure on each of these physiological effects are discussed below. When compared to sediment exposure events that elicit behavioural responses, longer exposure periods and/or higher concentrations are generally required before physiological responses are expressed. In this respect, physiological responses are more of a chronic effect. The effects are usually a graded response to increasing sediment dose. Impacts evoked from lower doses can be transitory, while those resulting from higher doses can be more lasting and severe.

Growth

An impaired growth rate is generally one of the more sensitive physiological responses to an increase in suspended sediment concentration. Unlike behavioural responses, impaired growth generally requires a longer exposure period before effects are manifested. Sigler *et al.* (1984) found that growth was impaired in juvenile steelhead trout (*Oncorhynchus mykiss*) and coho salmon exposed to fire clay or bentonite clay at concentrations between 84 and 120 mg/L during a 14 to 21 day exposure period. Similar concentrations of 100 mg/L or greater were found to significantly impair growth in Arctic grayling under-yearlings (McLeay *et al.* 1987), largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and redear sunfish (*Lepomis gibbosus*) (Buck 1956). However, growth impairment may be related more to the metabolic demands resulting from stress caused by increased suspended sediment than from a reduction in feeding. The time required before growth impairment was measurable ranged from a low of two weeks for juvenile steelhead trout and coho salmon to a high of six weeks for Arctic grayling under-yearlings (Sigler *et al.* 1984; McLeay *et al.* 1987).

Blood Chemistry

Alteration in blood chemistry resulting from the increased stress of suspended sediments have been found associated with concentrations ranging between 500 to 1500 mg/L (Redding and Schreck 1982; Servizi and Martens 1987). The changes most commonly recorded include an increase in haematocrite, erythrocyte count, hemoglobin concentration, and elevated blood sugar levels (hyperglycemia), plus decreases in blood chloride content, and depletion of liver glycogen (Wedemeyer *et al.* 1990; Servizi and Martens 1987). These increases coincide with the release of stress hormones (i.e., cortisol and epinephrine) and traumatization of the gill, and presumably represent a compensatory response to a decrease in gill function (Newcombe 1994). In addition, Sherk *et al.* (1973) found these changes to be associated with a reduction in the swimming endurance of white perch (*Morone americana*) exposed to 650 mg/L of TSS. Most of the observed changes resulted after four to five days of exposure (Newcombe 1994). Exceptions to this, however, were noticed by Redding and Schreck (1982) who found a significant increase in haematocrit volume within steelhead trout after only nine hours of exposure to 500 mg/L of volcanic ash, clay and topsoil.

Gill Trauma

Increased concentration of suspended sediments are known to physically traumatize gill tissue. The primary mechanisms of action is through physical abrasion of tissue and particle adsorption onto the gill. The types of tissue changes observed include swelling of secondary lamella and hypertrophy (cell swelling) of epithelial cells (Sherk *et al.* 1973); hyperplasia (increase in cell number) of gill tissue (Simmons 1984); and tissue necrosis (Servizi and Martens 1987).

The severity of damage appears to be related to the dose of exposure, as well as the size and angularity of the particles involved. Greater damage is typically observed with larger, more angular particles (Servizi and Martens 1991). These factors could account for the large range in responses seen for different exposure rates. For example, concentrations as low as 270 mg/L are known to cause gill damage in rainbow trout (*Oncorhynchus mykiss*) (after 13 days of exposure (Herbert and Merkens 1961) and yet McLeay *et al.* (1987) found no gill damage in young-of-the-year (YOY) Arctic grayling that were exposed to concentrations as high as 1300 mg/L; the duration of exposure was, however, only four days.

Secondary effects resulting from an infestation of parasitic protozoans were found in juvenile rainbow trout that were exposed to extremely high concentrations of suspended sediments. The trout were exposed to 4887 mg/L for a period of 16 days (Goldes 1983). This author did noted that the protozoan infection and gill architecture was found to be normal 58 days after the exposure ceased.

Resistance

Increased concentrations of suspended sediments have been associated with an overall decrease in the ability to defend against disease and to tolerate chemical toxins. For example, Herbert and Merken (1961) observed rainbow trout to be more susceptible to infestations of fin rot when fish were exposed for 121 days to concentrations of 270 mg/L of diatomaceous earth. Likewise, Servizi and Martens (1991) found a correlation between the prevalence of a viral kidney infection and an increased concentration of suspended sediments in coho salmon. When concentrations of suspended sediments exceeded 100 mg/L, the tolerance of Arctic grayling to the toxicant pentachlorophenol (PCP) decreased (McLeay *et al.* 1987). This observation by McLeay *et al.* (1987) indicates a general decrease in tolerance to increased environmental stressors.

Phagocytosis

A process that may be closely linked to reduced resistance, is phagocytosis. Newcombe and Jensen (1995) discuss the process by which fine particles are enveloped by cells within fish gill and gut tissues and are transported to internal repository tissues. The main organ of repository in fish is the spleen (Newcombe and Jensen 1995). It is hypothesized that through this process, particles could reduce resistance to other stressors by impairing fish health. In addition, particles could trigger tumour induction, especially in circumstances where contaminants were adsorbed to particles in suspension (Newcombe and Jensen 1995).

Lethal Effects

Increased concentrations of suspended sediments and increased sedimentation rates have the potential to affect fish populations. The primary mechanisms of action are through increased egg mortality, reduced egg hatch, a reduction in the successful emergence of larvae, plus the sediment-induced death of juvenile and adult fish. These mechanisms are discussed below.

Egg Mortality

The primary cause of egg death is generally from burial by settled particles. Thin coverings (a few mm) of fine particles are believed to disrupt the normal exchange of gases and metabolic wastes between the egg and water. Sedimentation rates of 0.03 to 0.14 g dry weight sediment/cm^2 (i.e., 1-4 mm depth of silt and clay) significantly reduced the survival of lake whitefish (*Coregonus clupeaformis*) eggs (Fudge and Bodaly 1984). The effects upon egg mortality appear to be more closely related to the sedimentation of particles and less related to the concentration of suspended sediments. Zallen (1931) observed that concentrations of 1000 to 3000 mg/L had no effect upon the survival of mountain whitefish eggs (*Prosopium williamsoni*). Campbell (1954), however, found 100 percent mortality in rainbow trout eggs exposed to TSS concentrations of 1000 to 2500 mg/L. Differences in egg mortality effects associated with elevated sediment loads is related to the size of the sediment particles involved and rates of sediment deposition.

In addition to the concentration of suspended sediments and the size of the particles involved, the duration of exposure appears to be key a factor in determining the effects of sediments on egg survival. Slaney *et al.* (1977) noticed that hatching success for rainbow trout was reduced after 2 months of exposure to 57 mg/L. A significant reduction in the hatching success of white perch and striped bass (*Morone saxatilis*) was observed in only 7 days after exposure to about 1000 mg/L TSS (Auld and Schubel 1978). The magnitude of the effect of sediment exposure may also be influenced by the timing of sediment exposure with respect tot the stage of embryo development. The dose of sediment required to induce egg mortality is greatly influenced by the physical characteristics of the stream which, in turn, affect sediment transport capabilities and the capacity to maintain sediments in suspension or otherwise to result in their deposition.

Juvenile and Adult Fish Death

Juvenile and adult fish generally appear to be more resilient to stress from suspended sediments than other life history stages. Short term increases in TSS concentrations between 11000 and 55000 mg/L appear to be the point at which salmonid mortality significantly increases (Stober *et al.* 1981; Servizi and Martens 1987; Smith 1940). McLeay *et al.* (1983) reported survival of Arctic grayling subjected to moderately high concentrations (1000 mg/L) of fine grained materials (mining silt). Lloyd (1987), in a review of existing information, reported lethal effects to fish at concentrations ranging from 500 to 6000 mg/L. Sigler *et al.* (1984) reported mortality in young of the year coho salmon and steelhead trout at 500 to 1500 mg/L. Based on the information available on sediment and acute effects to fish, it is apparent that the severity of effect caused is a function of many factors which, in addition to concentration, duration particle size and life history stage, may include temperature, physical and chemical characteristics of the particles, associated toxicants, acclimitization, other stressors and interactions of these and other factors (Waters 1995).

Habitat Effects

Habitat Exclusion and Habitat Alteration

In addition to the direct impacts of suspended sediments on fish, increases in sediment loads can also alter fish habitat or the utilization of habitats by fish (Scullion and Milner 1979, Lisle and Lewis 1992). High sediment loads can alter fish habitats temporarily by affecting water quality, making a stream reach unsuitable for use by fish. This habitat exclusion, if timed inappropriately, could have impacts on fish populations if the affected habitat is critical to the population during the period of elevated sediment load. This principle of habitat exclusion is very important one; however, this issue is separate from the issue of direct habitat alteration that will be discussed below.

Sediment episodes can have a prolonged effect on the suitability of habitats within a stream reach through increased levels of sedimentation. In fact, sedimentation is the single most important effect associated with sediment load increases, since sediment loads can alter the gross morphology of streams as well as the composition of the stream bed and associated habitats.

Changes in Stream Bed Porosity

Larger-sized materials, such as fine to coarse sand are quick to settle onto the stream bed. This material may accumulate on the surface of the stream bed or filter down into the inter-gravel spaces. Interstitial spaces can become clogged by the downward or, to a lesser extent, by the horizontal movement of sediment (Beschta and Johnson 1979).

Water movement through the stream bed materials is important for the benthic communities which reside there, and for the developing embryos of fish species who bury their eggs. Inter-gravel water movement is controlled by several hydraulic and physical properties of the stream and its bed. The permeability of the stream bed is determined by size composition of the substrate material, viscosity of the water (temperature dependent) and the packing of the substrate material (Stuart 1953; Cooper 1965). A small increase in the proportion of fine material can severely reduce the porosity and permeability of the gravel bed (Lisle and Lewis 1992) and the ability of alevins to receive adequate oxygen and emerge from the gravel.

Changes in Stream Morphology

In addition to altering stream bed composition, elevated sediment loads can also change channel geometry (Klein 1984). Elevated levels of sediment deposition can reduce the depth of pools and produce a net reduction in riffle areas. This accumulation of streambed deposits can reduce available habitat. For example, deposition of sediments in pools and other areas of instream cover can cause a decrease in the fish holding capacity of a stream reach (Bjornn *et al.* 1977). Smith and Saunders (1965) found that decreased brook trout (*Salvelinus fontinalis*) populations were related to infilling of available cover. Alexander and Hansen (1992) also noted than a decrease in sand bedload sediment was associated with an increase in rainbow trout and brown trout (*Salmo trutta*) populations. Changes in physical morphology of the stream can also inhibit the movement of fish or change the distribution of adult fish (Alabaster and Lloyd 1982).

Channels affected by sediment derived from Anthropogenic disturbance are also more transitory in nature. Fox (1974) found urban watersheds exhibited a 33% monthly change in geometry as compared to a 5% change in less disturbed rural drainages. Sediment material deposited within streams can be in constant motion as bedload transport slowly moves the deposited materials through the system. This material in motion can increase bed scour and bank erosion as the sediment increases the erosive force of the water, by creating a “sand blasting” effect.

Sedimentation Effects on Spawning Habitats

River spawning salmonids typically deposit their eggs in gravel beds commonly found in the upper reaches of river systems. For example, brown trout typically bury eggs in interstitial spaces of the substrata to depths of 9 to 12 cm (Scullion and Miller 1979). Alevins remain in the interstitial spaces until the start of exogenous feeding. The percolation of water through the incubation substrate is an essential factor in determining the survival rate of incubating eggs (Lisle and Lewis 1979).

An increase in percent of fine material in the stream bed can have impacts on egg survival rates (Shaw and Maga 1943; Cordone and Kelley 1961) since it reduces streambed permeability. Lowered permeability reduces the interchange between stream flow and water movement through the redd, resulting in a reduction in the supply of dissolved oxygen to the egg and a hindrance to the removal of metabolites. Slaney *et al.* (1977) reported that rainbow trout egg survival was significantly reduced when spawning gravel contained more than 3% of fines (diameter 0.297 mm). In addition, Hall and Lantz (1969) determined that hatching success of coho salmon and cutthroat trout (*Onchorhynchus clarki*) was reduced by 40 to 80% when spawning substrates contain 20 to 50% fines (1-3 mm diameter).

Even if intergravel flow is adequate for embryo development, sand that plugs the interstitial areas near the surface of the stream bed can prevent alevins from emerging from the gravel (Koski 1966; Phillips *et al.* 1975). For example, the emergence success of westslope cutthroat trout was reduced from 76% to 4% when fine sediment was added to redds (Weaver and Fraley 1993).

Female stream spawning salmonids typically clean an area of the stream bed in which they bury their eggs. This nest building activity flushes sediments and increases the stream bed permeability. With time, sediment conditions within the redd gradually return to ambient levels (Wickett 1954; McNeil and Ahnell 1964). Under normal conditions, this slow increase in sediment intrusion is not a problem; however, increased levels of sediment within a system as a result of anthropocentric disturbance increase the rate and level of sediment intrusion and reduces the period of time in which the redd is clean. The period of time before sediment intrusion into the redd is very important with respect to the survival of salmonid larvae. Studies by Wickett (1954) suggest that sediment accumulation during early embryonic development may result in higher egg mortalities than if deposition occurs after the circulatory system of developing larvae is functional. This may be due to the higher efficiency in oxygen uptake by the embryo or alevin with a functional circulatory system (Shaw and Maga 1942).

Ringler and Hall (1975) documented increased temperature and reduced dissolved oxygen levels of intragravel water in salmon and trout spawning beds because of clearcut logging practices. An associated reduction in resident cutthroat trout populations was attributed to this reduction in spawning habitat suitability. However, the failure to document serious reductions in coho salmon could be related to sediment clearing and removal by these larger fish during redd construction.

Sedimentation Effects on Fish Rearing Habitat

Sediment deposition also affects rearing habitat of juvenile fish since young salmonids frequently use the interstitial spaces in the stream bed for cover. Thus, a reduction in the suitability of potential rearing habitat as a result of sediment introduction is related to a reduction in the space available for occupancy (Reiser *et al.* 1985). When pools and interstitial spaces in gravel fill with sediment, the total amount of habitat available for rearing is reduced (Bjornn *et al.* 1977). Griffith and Smith (1993) found that numbers of juvenile rainbow trout and cutthroat trout decreased due to lack of available cover in heavily embedded gravel substrata. Interstitial space is particularly important during winter because juvenile fish live in these areas making them especially susceptible to impacts from increased sedimentation (Bjornn *et al.* 1977). Without these inter-gravel refugia, young fish may abandon the stream or move to less suitable areas where survival rates may be reduced.

Sedimentation Effects on Food Supply

Sedimentation can affect fish populations by altering the available food supply. Increased concentrations of suspended sediments and increased rates of sedimentation can reduce the primary productivity of the impacted area. Periphyton communities are likely the most susceptible to the scouring action of suspended particles or burial by sediments. At concentrations exceeding 115 mg/L, suspended sediments can reduce light penetration and primary productivity (Singleton 1985). A reduction in primary productivity has the potential to appreciably decrease the food supply of macrobenthos that graze on periphyton (Newcombe and Macdonald 1992). Many macrobenthic organisms are, in turn, used as a food source by fish.

Increased sediment loads in streams can also have an effect on zooplankton and macrobenthos. Sediment release can effect the density, diversity and structure of resident invertebrate communities (Gammon 1970; Lenat *et al.* 1981). A number of studies have demonstrated decreases in invertebrate densities and biomass following sedimentation events (Wagener 1984; Mende 1989). Increases in sediment input may reduce the density of invertebrates by directly affecting aspects of their physiology or by altering their habitat. Suspended sediments can have an abrasive effect on invertebrates and interfere with the respiratory and feeding activities of benthic animals (Tsui and McCart 1981). Increased sediment deposition may also reduce the biomass of invertebrates by filling the interstitial spaces with sediments and by increasing invertebrate drift or covering the benthic community in a blanket of silt (Cordone and Kelley 1961; Tsui and McCart 1981). Increases in sediment deposition that affect the growth, abundance, or species composition of the periphytic (attached) algal community will also have an effect on the macroinvertebrate grazers that feed predominantly on periphyton (Newcombe and MacDonald 1991).

A change in particle-size distribution in the stream bed can alter the habitat and make it unsuitable for certain species of invertebrates. Gammon (1970) noticed that an increase in suspended sediments from 40 to 120 mg/L resulted in a 25 to 60% decrease in the density of stream macroinvertebrates. Likewise, Slaney *et al.* (1977) found that a 16 hour pulse of suspended sediments (2500 to 3000 mg/L) led to a 75% reduction of invertebrate biomass within the most affected areas.

Sedimentation can alter the structure of the benthic invertebrate community by causing a shift in the proportion from one functional group to another. For example, streams with clear water normally contain a high proportion of invertebrates in the shredder group; however, if sediment deposition is substantially increased, shifts to other groups such as grazers (Bode 1988) or collector-gatherers may occur (Wagener 1984). Some studies indicate that increased inputs of sediments cause a shift towards chironomid-dominant benthic communities (Rosenberg and Snow 1975; Dance 1978; Lenat *et al.* 1981).

Benthic fauna possess behavioural and morphological adaptations which limit them from being displaced in a unidirectional flow environment (Hynes 1973). Invertebrate drift, however, is a continuous redistribution mechanism that occurs in most stream ecosystems. It is an important factor in the regulation of population density (Williams and Hynes 1976), in the dispersion of aggregations of young larvae (Anderson and Lehmkuhl 1967), in the abandonment of unsuitable areas (Williams and Hynes 1976), and in the recolonization of areas after disturbance (Barton 1977).

Invertebrate drift may be induced by elevated suspended sediment levels (Rosenberg and Weins 1978). Increased rates of downstream drift by macrobenthos can be induced by concentrations as low as 23 mg/L (Rosenberg and Snow 1975). Drifting affords invertebrate taxa that are sensitive to increased sediment loads the opportunity to avoid areas which become unsuitable as a result of high suspended sediment levels. Conversely, invertebrate drift is considered to be the most important component of ecosystem recovery following stream disturbances (Williams and Hynes 1976; Barton 1977; Young 1986). This is especially true in areas of swift-flowing waters (Waters 1964).

Sedimentation Effects on Overwintering Habitat

The magnitude of impact upon fish resulting from increased concentrations of suspended sediments and levels of sedimentation can vary seasonally. It has been argued that the lowered metabolic requirements during winter conditions may in some ways provide a protective influence to conditions such as gill trauma and decreased gill function (C. Newcombe, BCMELP, pers. comm.). However, the ability of the fish to compensate for the stress of suspended sediments is influenced by a number of factors including the physiological condition of the fish and its ability to respond to the stress.

Early live stages (i.e., eggs, alevins) of many salmonids are found in the stream bed during the winter months. These stages are particularly sensitive to the effects of increased concentrations of suspended sediments and the deposition of fines sediments. The introduction of sediments during the winter, therefore, has the potential to appreciably influence these early life stages.

Bjornn *et al.* (1977) found that the number of juvenile salmon that a stream can support in winter was greatly reduced when the inter-cobble spaces were filled with fine sediment. The decreased carrying capacity was a function of both a loss of substrate cover for juvenile fish and a reduction in food as benthic invertebrate communities changed. Bjornn *et al*. (1977) suggested that the summer rearing or winter holding habitat may be more influential to the carrying capacity of a stream reach than embryo survival.

During winter fish generally experience decreased energy reserves and as such search for habitat that allows them to reduce energy expenditures (Clapp *et al.* 1990; Nickelson *et al*. 1992). Preferred habitats are species dependent; however, for most salmonids preferred habitats are located in low velocity areas such as pools and behind instream cover were focal velocities are low (Vondracek and Longanecker 1993; Griffith and Smith 1993; Modde *et al.* 1991; Heggenes and Saltveit 1990; Cunjak and Power 1986; Tschaplinski and Hartman 1983). By remaining in low velocity areas, fish are able to minimize their energy expenditures and hence reduce the rate of metabolic depletion (Cunjak and Power 1986).

Land-use activities that increase the delivery of fine materials to streams can significantly affect the overwintering survival of resident fish. A mechanism of potential impact is a depletion of critical energy reserves as a result of increased physiological stress, alterations in behaviour and/or exclusion from preferred sites of overwintering habitat. This is particularly deleterious to fish species and life stages that prefer to overwinter within the interstitial spaces of the stream bed. The net loss in energy reserves will depend on the concentration of sediment and the duration of impact. Dependent upon existing energy reserves, the fish may be able to tolerate the energy depletion attributable to an increase in the cough reflex and reduced feeding, but may not be able to tolerate the energy depletion associated with displacement from critical habitats.

Preferred winter habitat areas of low current velocity are often predisposed to sedimentation (Cunjak 1996), and lower flows often experienced during winter may result in higher rates of sediment deposition. Bjornn *et al*. (1977) found, during sediment experiments, that the spring freshet from snow melt was rarely sufficient to transport sediment out of pools; therefore, the damage to these areas is frequently of a longer-term than sediments deposited in more erosive habitats. Due to natural factors, the availability of winter habitat is generally less than that of summer habitat and may be more influential in the determination of the stream's natural carrying capacity (Cunjak 1996; Mason 1976). A further reduction in the abundance of already limited winter habitat may significantly affect the overall fish population of a watercourse (Hartman and Scrivener 1990). Additive to this problem may be a reduction in food supply resulting from benthic drift or burying of food supplies. Elwood and Waters (1969) observed that increased sedimentation reduced the population of invertebrates and hence the capacity of the stream to support brook trout. A reduced food supply, and a greater expenditure of energy in food search and avoidance of higher concentrations of suspended sediments may significantly impact upon the fish's ability to compensate for negative physiological changes and the ability to survive the winter.

Sediment Load Biological Response Relationships

The Dose/Response Approach

One method that has been developed to address the issue of quantifying the adverse effects of TSS on fish is the ranked effects model first put forward by Newcombe and MacDonald (1991). This model compiled information from more than 70 studies on the effects of inorganic suspended sediments on freshwater fish (mainly salmonids) and invertebrates, and ranked the severity of impacts from 1 to 14 (rank effects). Linear regression was used to correlate ranked effects with intensity (concentration x duration) of increased suspended sediment load (Newcombe and MacDonald 1991). Since the effect of elevated TSS levels on fish is a function of both the concentration of suspended sediment and the duration of the exposure, Newcombe and MacDonald (1991) developed a Severity Index (SI). This Index provides a standardized relative measure of exposure. It is the natural logarithm of the concentration ($mg \cdot L^{-1}$) multiplied by hours of exposure (i.e., Ln $mg \cdot h \cdot L^{-1}$). This SI provides a convenient tool for predicting effects of episodes of elevated suspended sediments of known concentration and duration.

The usefulness of the Newcombe and Macdonald (1991) concentration-duration response model has been questioned in the past (Gregory *et al.* 1993). The main concerns with the approach are the highly variable nature of the data used to develop the severity of effects model that reduces its predictive power, and the concern that the model is unrealistically simplistic (Gregory *et al.* 1993). A separate concern associated with the Newcombe and Macdonald (1991) severity of effects model is the severity index (Ln $mg \cdot h \cdot L^{-1}$) assumes a unit increase in episode duration (in hours) has a similar effect as a unit increase in concentration (in mg/L) (Anderson *et al.* 1996).

In 1994, the ranked effects model was further refined (Newcombe 1994). Using 140 articles on suspended sediment pollution, Newcombe developed a database of nearly 1200 datapoints concerning the effects of suspended sediments and associated effects upon marine and freshwater biota. With this database, regression analysis was used to relate severity of effect to the dose of TSS for specific fish species or assemblages. This approach was used to describe the dose/response relationship for the effects of suspended sediments on salmonid fishes (Newcombe and MacDonald 1991), on juvenile salmon (Newcombe 1994) and for other coldwater fishes (Newcombe 1994) using sub-sets of the dataset which are presented in complete form in Newcombe (1994).

In addition to the dose/response relationships presented in for salmonid fishes (Newcombe and MacDonald 1991) and coldwater fishes and underyearling trout (Newcombe 1994), Newcombe and Jensen (1995) further expand upon dose/response relationships of aquatic resources to sediment exposure. Newcombe and Jensen (1995) presented six sediment dose/response relationships for specific fish communities exposed to elevated sediment loads. One of the important additions to the analysis presented in Newcombe and Jensen (1995) was the linkage of grain size of sediment to the nature of ill-effect associated with sediment exposure. The dose/response relationships presented in Newcombe and Jensen (1995) were organized according to four variables: taxonomic group; life stage; natural history and estimated predominant particle size range of the sediment episode. The dose/response relationships were characterized by three variables: [x], [y], and [z], where:

[x] = duration of exposure expressed as the natural log of hours;
[y] = concentration of sediment expressed as natural log of mg/L;
[z] = severity of ill effect (SE).

Since the work of Newcombe and MacDonald (1991) and Newcombe (1994) used the Stress Index (Ln concentration · duration), the dose/response relationships presented were of the form:

Equation 1: $z = a + bx$
where:
SE = severity of ill effects
a = intercept;
b = slope of the regression line
x = stress index value

The six dose/response relationships presented in Newcombe and Jensen (1995) do not characterize response using the stress index, and are presented in the form:

Equation 2: $z = a + b(\text{Ln } x) + c(\text{Ln } y)$

The dose/response relationships (Newcombe and MacDonald 1991; Newcombe 1994; Newcombe and Jensen 1995) provide insight into the relationship between sediment release and adverse effects on a variety of fish communities. These relationships provide increased precision for the prediction of response of a particular fish species or assemblage of species based on a given dose of suspended sediment.

The database used to determine these relationships relied heavily on the physiological response of fish to increases in sediment load; therefore, the relationships presented may not be directly applicable to the prediction of physical alteration to fish habitats due to sediment load increases and increased sedimentation nor the long term impacts from reduced growth or the possible exclusion from certain habitats.

Developing the Habitat Effects Database

Anderson *et al.* (1996) attempted to develop effects relationships for fish habitat response to increased sediment. The first step in developing more specific dose/response criteria for habitat effects was to search the literature for studies reporting TSS concentrations and their effects on fish habitat. Several fundamental criteria had to be satisfied for data from a report to be included in the expanded database that was developed. The report had to give at least: 1) a concentration of TSS; 2) a duration of exposure to 3) one or more identified species of fish; and 4) a description of the effect. The severity of effect, and occasionally other data, sometimes had to be inferred from the qualitative descriptions. A total of 18 reports, containing some 53 new documentations of TSS effects, were retrieved in the literature search (Anderson *et al.* 1996).

Dose/Response Relationships of Freshwater Habitats to Sediment Releases

Severity of ill-effects rankings on a scale from 0-14 were assigned to each documented effect following the severity-of-ill-effects scale published by MacDonald and Newcombe (1993), Newcombe (1994), and Newcombe and Jensen (1995). The nature of the observed habitat effect was assigned one of the following class effect rankings:

- SE = 3 Measured change in habitat preference;
- SE = 7 Moderate habitat degradation - measured by a change in the invertebrate community;
- SE = 10 Moderately severe habitat degradation - as defined by measurable reductions in the productivity of habitat for extended periods (months) or over a large area (kms);
- SE = 12 Severe habitat degradation - as measured by long-term (years) alterations in the ability of existing habitats to support fish or invertebrates; or ,
- SE = 14 Catastrophic or total destruction of habitat in the receiving environment.

Emphasis was placed on sediment release events (i.e., effects of sediment pollution events rather than chronic erosion and sediment load problems). The database excluded any datapoints for which the extent of habitat modification could not be ascertained from the primary manuscripts. The database was reduced to 35 entries (Anderson *et al.* 1996) and was used in the analysis described below to develop a relationship between sediment dose and habitat effects.

The dose/response approach of Newcombe and MacDonald (1991) and Newcombe (1994) defines dose as the product of TSS concentration (C in mg/L) and duration of exposure (T in hours). This definition of dose is strictly empirical and reflects the observation that the product of concentration and duration bears a closer correlation with ranked effects than concentration alone. The inherent assumption is that brief exposures to high doses of TSS are equivalent in effect to prolonged exposure of much lower doses. Since severity of effect is determined based on a linear relationship with dose ($\ln C \cdot T$) of the form $SE = a + b (\ln C \cdot T)$, the biological receptor response (SE) is assumed to respond to an effective dose in which concentration and duration are equally as important (i.e., the Effective Dose = C^nT; where $n = 1$).

However, it has been proven in much of the literature related to the response of biological receptors to toxic agents, that the relationship between concentration and duration is often more complex. That is, a high concentration for a very short time can cause a higher or lower response than can a low concentration for a longer time (Zelt 1995). In essence, by assuming a linear response (as measured by SE) to dose (as a function of $\ln C \cdot T$), it is assumed that a unit increase in concentration (in mg/L) is equal to a unit increase in time (in hours). This assumption may or may not be a valid one. In an effort to address the potential for non-linearity in the relationship between concentration and duration in determining the effective dose of sediment (i.e., Effective Dose $=C^n \cdot T$; where $n \neq 1$), Newcombe and Jensen (1995) used multiple regression analysis to develop severity of effect relationships based on concentration and duration:

Equation 3: $$z = a + b(\ln x) + c(\ln y)$$

This approach, in effect, allows for different factors (slopes) to be assigned separately to the variables of concentration and duration.

In order to explore the relationship between concentration and duration in influencing habitat change, Anderson *et al.* (1996) also used multiple regression analysis to analyze the habitat effects database. This analysis identified a relationship between sediment exposure and habitat effects that can be described by the equation:

Equation 4: $z = 0.637 + 0.740\mathrm{Ln}(X) + 0.864\mathrm{Ln}(Y)$;
$r^2(\mathrm{adj}) = 0.627$; $n=35$; $p<0.001$.

Statistics for the multiple regression relationship presented are summarized in Table 1. The "T" statistic for each slope in the regression (Table 2) is an expression of the importance of each variable with respect to the relationship derived. The higher score attributed to duration indicates its importance in determination of habitat effects. This indicates that concentration and duration affect the extent of habitat alteration in dissimilar ways or in other words, that the effective dose of sediment is a function of a non-linear relationship between the two predictive variables (i.e., Effective Dose $=C^n \cdot T$; where $n \neq 1$).

Table 1
Statistics for the Multiple Regression Relationship (Equation 4)

Variable	Coefficient	Std. Error	Std Coef	Tolerance	T	P (2Tailed)
Constant	0.0637	1.293	0.000		0.493	0.625
Ln Con.	0.864	0.176	0.520	0.973	4.903	0.000
Ln Duration	0.740	0.111	0.706	0.973	6.652	0.000

Discussion

Confounding Factors

The dose/response relationships that have been developed make generalizations about the anticipated level of effects to the aquatic environment that may result from elevated sediment levels. Since these are generalizations, the actual effects that are realized by a sediment release episode may be more or less severe based on a number of confounding factors.

The potential for adverse effects on fish and their habitats associated with sediment release is a function of increasing particle size (Newcombe 1996). More information relating dose-response relationships between specific fish guilds or habitat types as a function of particle size range is required in order to develop a better understanding of this confounding factor.

The angularity or mineralogy of suspended particles may play a important role in the potential for physiological or toxicity effects (Newcombe and Jensen 1995). The angularity of a particle may be of particular importance with respect to gill abrasion of fish within the receiving environment, and may also influence the rate of infiltration of particles into the stream bed. Meanwhile, the mineralogy of the particle may be important since the particle itself may have some potential chemical activity at the cellular level (Newcombe and Jensen 1995). In addition, the potential for contaminants adsorbed to sediment particles is also a concern, since contaminated sediments could have more dramatic effects than those which might be caused by the increase of sediment load alone.

The amount of material that intrudes into the gravel bed has been shown to be highly dependent on the grain-size distribution of the transported sediment as well as that of the gravel bed. If the suspended sediment load is composed of very fine material, the gravel pores tend to fill from the bottom to the top of the pavement layer. If the suspended particles are larger in size, angular or platelet in shape, a film can develop within the substrata which will tend to limit the intrusion of additional sediments into the interstitial spaces of the stream bed (Beschta and Jackson 1979). Beschta and Jackson (1979) concluded that the finer the suspended sediment, the greater the potential was to fill interstitial voids.

The shape of the stream bed substrata may also affect sediment deposition. Under low flow conditions, rounded stream bed substrata tend to accumulate more sediment than angular substrata, whereas, during high flows, the reverse is true (Meehan and Swanston 1977). This may be due to the reduced turbulence levels at the gravel bed in rounded stream beds during low flows, while at higher discharges a flow separation zone can develop behind angular materials causing greater sediment deposition (Reiser *et al.* 1985).

The temperature of the water can have an impact on the severity of the effects caused by a sediment release event. The oxygen holding capacity of water and the metabolic and respiratory rates of fish are influenced by water temperature. Consequently, the effects of sediment exposure may be greater in seasonably warm waters than in seasonably cold water (Newcombe and Jensen 1995). However, during winter conditions aquatic organisms may be especially vulnerable to additional stressors. Since stress has been defined as the sum of all physiological responses, the severity of effects that are caused by sediment release will be a function of the level of stress at the time of, or before, the period of elevated sediment load.

Factors Influencing the Risk of Habitat Alteration

Many attributes may influence the extent of sediment-induced aquatic habitat alteration. The principal factor which influences the extent of habitat alteration is the increase in sediment load associated with watershed disturbance. In addition, the sensitivity of the exposed habitats and the length of time habitats are likely to be impaired are also important factors which influence the level of habitat alteration.

Considerations regarding the sensitivity of the receiving environment include the susceptibility to alteration of the habitats within the receiving environment, and the timing of the elevated sediment loads. In addition to overall sensitivity of the watercourse and the sensitivity of the habitats it supports, a related consideration is the species and life stages which may be present during times of instream activity associated with forest harvesting or road construction. Certain life stages are especially sensitive to increases in sediment load (such as developing eggs and larvae, or overwintering fish – particularly juvenile salmonid in streams); as a result, the presence of these life stages during instream construction would increase the sensitivity of the watercourse to disturbance.

An additional related consideration regarding construction timing sensitivity is the flow conditions within the watercourse during periods of elevated sediment loads. Watercourse discharge influences the concentration of suspended sediments, transport and deposition of materials as well as the extent of habitat present and the ability of resident biota to avoid areas of elevated sediment.

The duration of habitat impairment is one of the most critical consideration in relating the extent of habitat alteration to aquatic community impact since habitat alteration will only affect the aquatic community if the altered habitat would have been used during the period of impairment. Therefore, the level of concern associated with habitat alteration increases as the duration of impairment increases. The duration of impairment is considered to be the length of time before deposited sediments are flushed from the watercourse into a less sensitive area such as a lake and are normally viewed as the number of life history stages which are impacted during the period in which the habitat is in an altered state.

As a result, evaluation of the extent of habitat alteration associated with elevated sediment loads must consider the level of sediment load increase (concentration and duration), the nature of the habitats and communities affected and the duration of likely impairment caused. The severity of effects approach which have been developed by Newcombe and others are not easily applied to the prediction of habitat change and attributing justifiable numbers to abstract concepts such as system sensitivity is difficult at best. Resource managers need to apply expert judgement to ensure that models and assumptions are not applied blindly and that model results do not violate the most important management tool, which is common sense.

Mitigation Measures

Although most disturbances within a watershed inevitably increase the amount of erosion, the delivery of sediments to streams resulting from disturbances can be largely circumvented by proper design and planning. As discussed previously, logging road construction and operation can dramatically increase sediment loads in streams. The amount of disturbance caused by road construction and maintenance depends upon its design standard, gradient, total distance of road and intensity of use. For example, Megahan and Kidd (1972) indicated that proper siting and construction of roads could eliminate much of the sediment loading associated with mass erosion in steep terrain.

The density of logging road distribution can be a major factor in determining the associated increase in sediment loads in streams (Waters 1995). Cederholm *et al.* (1981) documented the greatest accumulation of fine sediments in streambeds associated with road areas that exceeded 2.5% of the total basin area. In addition, the length of logging roads also influences sediment delivery to streams. Cederholm *et al.* (1981) calculated total road lengths of 2.5 km of road per km^2 of watershed basin produced sediment more than four times natural rates. As a result, sediment mitigation measures have concentrated on the minimizing sediment associated with logging roads.

The logging system used is a critical factor in the determination of road density. For example, the use of jammer logging systems in the past has required a dense network of roads since these logging systems require a maximum road spacing of approximately 150 m. In areas of steep terrain, this approach to logging may disturb soil on 25 percent of the total logging area. High lead logging is a method that reduces the level of disturbance since a reduced road network is required to support this method. In steep topography areas in Idaho, high lead logging has reduced disturbance to less than 10% of the logging area. On the same types of slopes, jammer logging roads spaced 60-120 m apart disturbed 25-30% of the total area (Rice *et al.* 1974). Skyline logging systems permit an even wider road spacing of 500 m or more, depending on local conditions and topography. This reduces the area disturbed by 75% and may provide a greater buffer area for sediment filtration between the road and the stream channel (Megahan and Kidd 1972). Skyline, balloon and helicopter systems have been developed to permit the logging of steep topography with a minimum amount of road disturbances (Rice *et al.* 1974). Binkley (1965) estimated that skyline yarding would save from 2.2 to 2.9 kilometres of road over that required for high lead in a 1600 ha drainage area to be logged.

Control measures to minimize sediment delivery to streams associated with accelerated erosion within watersheds disturbed by forest harvest operations are of paramount importance in minimizing elevated sediment loads in streams. Since the best mitigation measure to minimizes sediment loads in streams is to minimize the amount of erosion in the watershed, design features for access planning can be implemented which will minimize erosion associated with logging roads. Waters (1995) summarizes design features for the reduction of erosion from logging roads. This summary table is reproduced as Table 2.

Disturbance within watersheds inevitably increases sediment loads within adjacent watercourses. These increases in sediment load can have substantial effect on fish and on their habitat. Through the proper planning and design of forest harvest systems, the level of sediments delivered to streams can be minimized allowing for the existence of both forestry and fish.

Table 2
Logging Road Mitigation Measures
(Taken from Water 1995)

Design Feature	Method	Purpose
road placement	avoid streams and steep slopes	reduce erosion & mass soil movement
road length	few, short roads	reduce total area of exposed roadbed
road width	narrow as practicable	reduce area of disturbance
road grade	5-15%, not flat, minimum 3% for drainage	avoid rapid run-off
road surface	gravel, crushed rock roadbed	reduce roadbed erosion
cut slopes	vertical or near vertical cut	reduce excavation and erosion of slope
fill slopes	avoid road drainage and woody debris in fill	stabilize fill slopes
road drainage	outslope drainage on shallow slopes, inside drainage on steep grades	disperse drainage
inside drainage	ditch inside road	carry run-off along road
cross-drainage culvert	underground pipe or log construction	drain inside ditches or waterways
water bar	low hump, 30 ^{o} angle downslope	disperse run-off from roadbed
broadbased dip	wide drainage dip or bench	disperse run-off from roadbed
stream crossing	minimize number, use appropriate method	reduce direct aquatic impact
vegetation planting	seed grass, plant trees	reduce exposed surface
daylighting	cut canopy to permit sunlight penetration	promote drying of roadbed
abandonment	close access, remove crossings, install dips and water bars	avoid subsequent use and maintenance

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RESEARCH ARTICLE

Long-term grazing study in spring-fed wetlands reveals management tradeoffs

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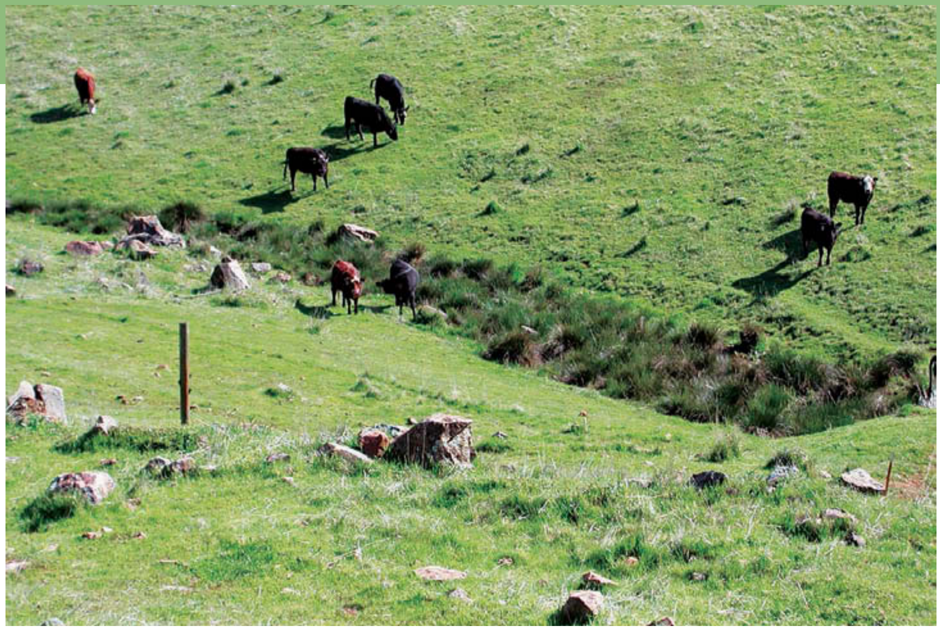


Oak-woodland springs provide green habitat and water throughout California's Mediterranean-style dry season, making them highly desirable ecosystems, islands of biodiversity and high productivity.

Spring-fed wetlands perform many important functions within oak-woodland landscapes, and livestock grazing modifies these functions. We used 10-year (long-term) and 3-year (paired-plot) experiments to better understand grazing management effects. We studied spring ecosystem responses in plant composition, diversity and cover; channel morphology; water quality; aquatic insects; and greenhouse gases. Lightly and moderately grazed wetlands exhibited lower insect family richness than ungrazed springs. Plant cover was maintained for the first 7 years of grazing, and plant diversity was not significantly affected. At the same time, removal of grazing decreased emissions of the greenhouse gas methane, and increased nitrate levels in spring waters. The results reveal important management tradeoffs relative to key response variables. In general, light cattle grazing at springs appears to be desirable from an ecosystem function perspective.

Wetland ecosystems are highly productive and valued for numerous reasons including wildlife habitat, biodiversity, water quantity and quality, and human uses. They are also relatively small ecosystems, occupying less than 1% of the state. Because livestock are thought to damage the physical, chemical and biological integrity of these systems, they are subject to government regulations, ranging from seasonal use requirements to complete livestock removal (Allen-Diaz and Jackson 2002). Livestock grazing can affect the functioning of spring-fed wetlands by acting as a nutrient filter and altering plant community composition (Jackson 2002).

These systems are also highly variable, making it difficult to predict responses to management (Allen-Diaz et al. 2001). For example, first-order (headwater) and fourth-order streams (such as the Yuba River) may have similar vegetation, but their responses to grazing may differ because of substrate (the bedrock, gravels and soils on which plants grow), slope or other environmental differences.

Spring-fed wetlands of the oak woodlands fall into two broad categories — rocky and marshy (Allen-Diaz and Jackson 2000). Where spring water emerges in and around rocky substrates, little soil development occurs. Water quickly forms channels, and overstory trees and shrubs are frequently present. The rocky wetlands typically maintain two distinct zones, an area immediately surrounding the emergent water source (referred to as springs) and the resulting channelized creek (fig. 1A).

On more shallow slopes, the flow of emergent water is slower and more diffuse, allowing the development of dense, herbaceous vegetation, which further reduces flow. These marshy sites typically do not support trees or shrubs, probably because of anaerobic soil.

Grazing and soil-water research

Rangelands occupy about 57 million acres in California. About 42% of these acres are privately owned and provide most of the forage for California's cattle industry. Approximately 9,000 miles of streams and 125,000 acres of wetlands occur on California rangelands. Many consider livestock grazing on rangeland a potential nonpoint source of pollution and thus, a serious threat to the health of California waters. Our research carefully examines these concerns.

In a long-term study, Experiment A, we tracked species composition and cover for more than a decade, primarily on rocky-type wetlands, under three levels of grazing intensity at the UC Sierra Foothill Research and Extension Center (SFREC) east of Marysville, Calif. (Allen-Diaz and Jackson 2000). Species composition was recorded in early June

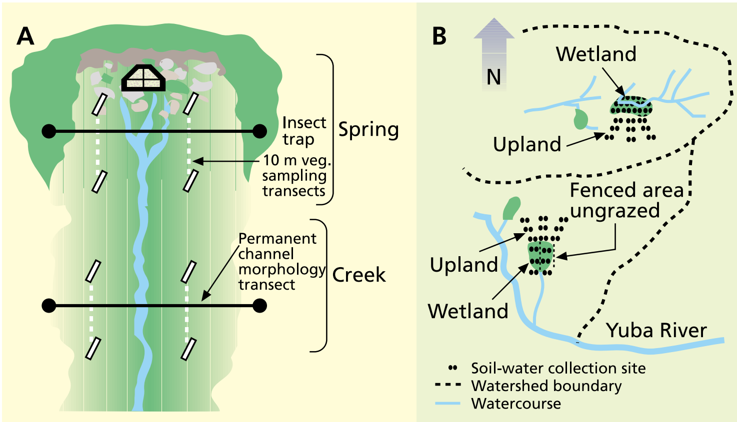


Fig. 1. Typical morphology of rocky spring-fed wetlands; Experiment (A) and (B) study layouts.

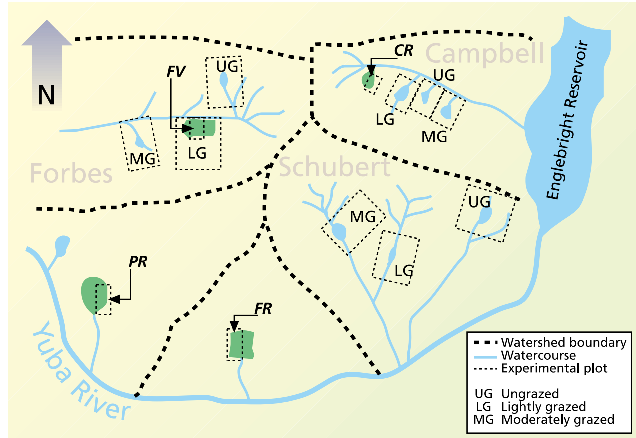


Fig. 2. Location of experimental plots at the UC Sierra Foothill Research and Extension Center. The 10-year (Experiment A) sites were in Forbes, Campbell and Schubert watersheds; the 3-year study (Experiment B) sites were Forbes Valley (FV), Poleline Ridge (PR), Campbell Roadside (CR) and Fireline Ridge (FR).

each year from permanent 16.25- or 32.5-foot (5- or 10-meter) line-point transects, which were randomly located on either side of the channel within the wetland zone. Changes in stream-channel shape (morphology)(Allen-Diaz et al. 1998) and water quality (Campbell and Allen-Diaz 1997) were also examined for approximately 5-year periods on the same sites (fig. 2). Channel morphology changes were recorded along two randomly placed 32.5-foot (10-meter) line-point transects located perpendicular to the flow of the spring or creek. We collected water samples at the springhead, or spring source, and in the creek and analyzed them in the field using a HACH DREL2000 Water Testing Kit.

In a paired-plot study at the SFREC, Experiment B, we examined marshy springs to closely evaluate soil-water nitrate levels and greenhouse-gas emissions for 3 years. We collected soil water from preinstalled, porous, soil-water cup samplers (Model 1900, SoilMoisture Equipment, Santa Barbara). When possible, samples of upland soil water and wetland surface water were collected monthly (fig. 1B). Within the wetland, surface-water samples were collected in 100-milliliter (mL) specimen cups to assess nitrate output and compare with upland soil-nitrate levels. The UC Division of Agriculture and Natural Resources Analytical Laboratory analyzed water samples by a diffusion-conductivity analyzer. Carbon (CO_2), nitrogen (N_2O) and methane (CH_4) gas emissions were collected monthly from March to September 2002 (with the exception of May) in vented static flow chambers. Gas samples were analyzed by gas chromatography (SRI Instruments, Torrance, Calif.).

All sites had historically similar fall-winter-spring grazing histories that left approximately 600 to 750 pounds per acre residual dry matter (RDM), or aboveground biomass, in the uplands. In 1993, sites within watersheds were randomly assigned to the following treatments in a randomized block design:

- Grazing removal (ungrazed, UG, approximately 1,200 to 1,500 pounds per acre upland RDM).
- Light grazing (LG, approximately 800 to 1,000 pounds per acre upland RDM).
- Moderate grazing (MG, approximately 600 to 700 pounds per acre upland RDM).

Experiment B took place on the marshy sites, to take advantage of their greater area. Marshy springs were sampled in 1999 and 2000. Then, these sites were divided so that grazing treatment comparisons could be made within sites (Jackson 2002). This meant that only two treatment levels could be compared — moderate grazing and grazing removal. Posttreatment samples were collected in 2001 (fig. 2).

Grazing effects on wetlands

Plant species composition. Changes in species composition provided evidence of fundamentally different vegetation dynamics in these systems. One way to examine these changes is with a CCA (Canonical Correspondence Analysis) site score, an index variable that collapses species composition into one measure. This statistical technique is useful for interpreting plant community structure that is related to environmental variables (McCune and Grace 2002). For example, in this case we used it to compare changes in species composition related to grazing levels in Experiment A. The CCA site scores were much more variable from year to year for springs than for creeks on the rocky-type wetlands (fig. 3). Implications for management are that species composition can be manipulated by altering the grazing intensity along creeks. In springs, how-

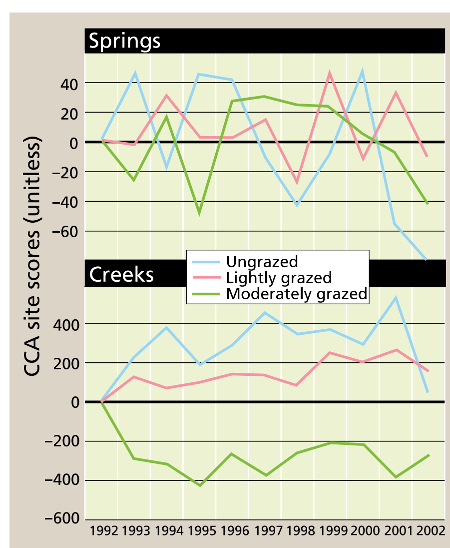


Fig. 3. Species composition over time as affected by grazing intensity in Experiment A.

Cattle grazing causes visual changes in oak-woodland spring structure. However, spring composition is stable over time, and hoof-caused hummocks do not result in detrimental changes to composition, productivity or water quality.

ever, species composition is controlled by the vagaries of climate, not by grazing intensities at the levels we studied.

Herbaceous diversity. No significant differences in the total number of species (relative to pretreatment 1992 values) were observed at any of the wetland sites. Common species are listed in table 1. In both experiments, there were no changes in the relative amounts of native and nonnative species over time under any grazing treatment. Lightly grazed wetlands maintained greater species evenness (maximum when all species have the same number of individuals) and diversity (Shannon-Weaver and Simpson indices) relative to 1992 pretreatment values than either ungrazed or moderately grazed plots (Jackson 2002). At creeks, moderately grazed plots maintained greater relative total species, evenness and diversity than lightly grazed and ungrazed plots, which were not significantly different from each other.

On marshy springs (Experiment B), we observed decreased diversity with grazing removal for 1 year. Our results indicate that light grazing on spring-fed wetlands and moderate grazing on resultant down-slope creeks maintain current plant diversity.

Herbaceous cover. Because plant cover conserves soil, improves water quality and is correlated with plant productivity, it is an important measure of ecosystem health. After 7 years, we found no significant differences in herbaceous cover among grazing intensity treatments. However, by 2002, moderate grazing resulted in a significant decrease in plant cover. Sustained grazing at moderate or higher intensities on these systems is not desirable from an ecosystem conservation perspective, to prevent significant erosion and prevent undesirable changes in species composition. However, our short-term study showed that occasional moderate grazing does not significantly affect plant cover.

Channel morphology. Five years of data from permanent cross-section transects of the springs and resultant creeks in Experiment A showed no changes in channel morphology due to grazing treatment (Allen-Diaz et al. 1998). Ungrazed springs and creeks exhibited more year-to-year variability than grazed springs and creeks, although these differences were not statistically significant. Channel widening and flattening of waterways can have important effects on fish populations, especially in second and lower-order streams.

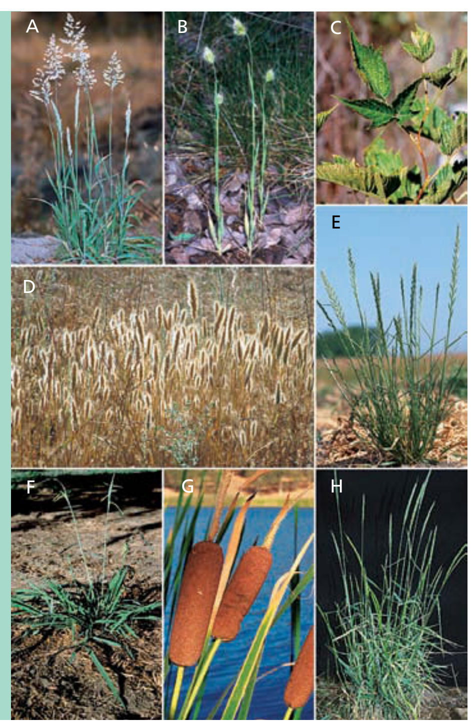
Water quality. Over a 5-year period we monitored nitrate, orthophosphate, dissolved oxygen, temperature and pH in surface water emerging from rocky spring-fed systems. We found no

TABLE 1. Dominant herbaceous-layer plant species in springs and creeks at Sierra Foothill Research and Extension Center, Experiments A and B

Common name	Species	Family	Native/ introduced*
Blue wild-rye **(H)**	*Elymus glaucus*	Poaceae	N
California blackberry **(C)**	*Rubus ursinus*	Rosaceae	N
California grape	*Vitis californica*	Vitaceae	N
Common spike-rush	*Eleocharis macrostachya*	Cyperaceae	N
Dallis grass **(F)**	*Paspalum dilatatum*	Poaceae	I
Dogtail **(B)**	*Cynosurus echinatus*	Poaceae	I
False brome	*Brachypodium distachyon*	Poaceae	I
Flat sedge	*Cyperus odoratus*	Cyperaceae	N
Hedge nettle	*Stachys albens*	Lamiaceae	N
Italian ryegrass **(E)**	*Lolium multiflorum*	Poaceae	I
Italian thistle	*Carduus pycnocephalus*	Asteraceae	I
Narrow-leaved cattail **(G)**	*Typha angustifolia*	Typhaceae	N
Rabbitfoot grass **(D)**	*Polypogon monspeliensis*	Poaceae	I
Rattlesnake grass	Briza minor	Poaceae	I
Ripgut brome	*Bromus diandrus*	Poaceae	I
Seashore vervain	*Verbena litoralis*	Verbenaceae	I
Soft chess	*Bromus hordeaceus*	Poaceae	I
Velvet grass **(A)**	*Holcus lanatus*	Poaceae	I
Water cress	*Rorippa nasturtium-aquaticum*	Brassicaceae	N

* N = native, I = introduced (Hickman 1993).

Photo credits: Joseph M. DiTomaso (A–E), Jack Kelly Clark (F), ANR Communication Services (G), Suzanne Paisley (H).



Removal of livestock grazing resulted in increased levels of nitrate in wetland waters and thus higher levels of nitrate pollution compared to grazed springs.

significant differences among grazing intensity treatments in Experiment A (Campbell and Allen-Diaz 1997).

In Experiment B, marshy spring-fed wetlands appeared to intercept and retain nitrate as it moved along its hydrologic path from upland soils to emergent spring waters (Jackson and Allen-Diaz 2002). Furthermore, the removal of livestock grazing from these wetlands impaired the ability of the springs to retain nitrate. Grazing removal allowed dead plant material to accumulate (fig. 4), thereby inhibiting plant production (hence, plant nitrogen demand), resulting in stream-water nitrate concentrations that far exceeded the U.S. Environmental Protection Agency's surface-water maximum standard of 714 μmol[5] (micromoles[5] or 10 parts per million [ppm])(fig. 5).

Aquatic insects. Aquatic insects are frequently used to evaluate the ecological integrity of streams. Reduced community numbers may indicate organic pollution or habitat degradation. Insects were identified to the family level from a 1-year sample (collected quarterly) in Experiment A. Analysis was limited to families with aquatic genera; wholly terrestrial families were excluded. Lightly grazed and moderately grazed wetlands exhibited lower family richness than ungrazed springs at each sampling date. The lowest cumulative family-richness values (sum of all families for the year) were found for moderately grazed springs followed by lightly grazed springs.

Greenhouse gases. Methane (CH_4) is a greenhouse gas that is important to global climate change. It is "radiatively active," warming the lower atmosphere by absorbing thermal radiation. Methane contributes approximately 20% of terrestrial trace gases into the atmosphere (Bouwman 1990; Cicerone and Oremland 1988). The atmospheric concentration of methane has increased from 0.7 to 1.7 ppmv (parts per million by volume) in the last 200 years (Tyler 1991). When compared to carbon dioxide (CO_2), the relative contribution of methane to the Earth's energy balance exceeds its relative concentration in the biosphere. This is because methane generally absorbs reflected radiation 25 times more effectively than carbon dioxide (Bartlett and Harriss 1993).

Wetland systems are generally considered sources of methane because biomass rots in anaerobic conditions (Schlesinger 1997). We assessed the effects of grazing on methane fluxes (the amount released into the air) in spring-fed wetlands in Experiment B (Oates 2002). Air temperature had the strongest influence on methane flux, followed by soil water content and grazing presence or absence. The mean methane flux was 9.29 ± 4.37 mg CH_4–C/m^{-2}/hr^{-1} (mean ± S.E.) with a range of –0.19 to 147.88 mg CH_4–C/m^{-2}/hr^{-1} (methane flux measured as carbon in the form methane in milligrams per square meter per hour). Water content at these sites was 39.66% ± 2.29%, with a range of 14.51% to 60.64%. Grazing removal significantly decreased methane emissions; grazed was 16.01 ± 8.48 mg CH_4–C/m^{-2}/hr^{-1}, and ungrazed was 2.57 ± 1.15 mg CH_4–C/m^{-2}/hr^{-1}.

Guidelines and research gaps

Spring-fed wetlands are small, patchy ecosystems nestled within a matrix of oaks and annual species; they are important in overall landscape structure and function in a way that is disproportionate to their size. Much of the water exiting California oak-woodland watersheds passes through these highly productive spring-riparian zones, which are located at the interface of the terrestrial-aquatic ecosystem. Our data indicates that wetland vegetation in these zones, typically cattails *(Typha angustifolia)*, sedges, rushes and perennial grasses, act as nutrient filters for waters emerging at the soil surface.

High herbaceous plant production is one of the key factors for maintaining ecosystem services, by promoting carbon sequestration and nutrient conservation from the terrestrial landscape. A factor such as grazing, which influences ecosystem productivity, is an important control on ecosystem services.

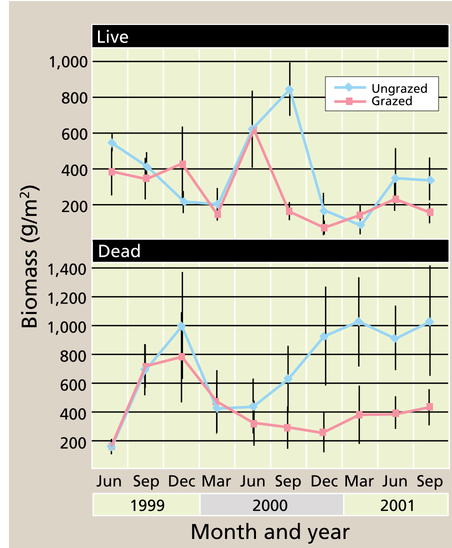


Fig. 4. Live and dead plant biomass from grazed and ungrazed plots in Experiment B.

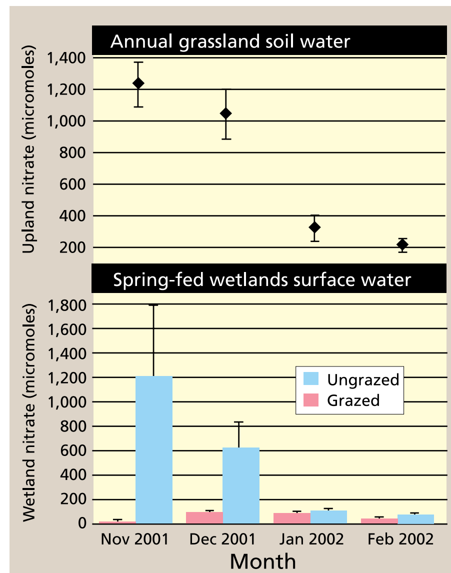


Fig. 5. Soil and surface-water nitrate concentrations from Experiment B during winter 2001-2002.

Livestock grazing also shapes plant communities in these systems. Our studies show that nutrients (nitrogen) from the surrounding environment flow into the spring systems, supporting great productivity in concert with water and energy surpluses. Removal of livestock grazing resulted in increased levels of nitrate in wetland waters and thus higher levels of nitrate pollution compared to grazed springs. When grazing was removed, these

Jack Kelly Clark

Research on the ecological impacts of cattle grazing was conducted over a 10-year period at the UC Sierra Foothill Research and Extension Center in Browns Valley (near Marysville).

Spring-wetland ecosystems evolved with grazing wildlife, and later domestic animals. Light livestock grazing fosters plant diversity and helps to maintain nitrate levels in spring waters.

systems underwent greater changes in plant composition resulting in decreased plant diversity. Some degree of herbivory appears desirable from an ecosystem function perspective, although consistently high grazing intensities will reduce herbaceous cover to undesirable levels.

Future work should examine ecosystem interactions with the atmosphere as greenhouse-gas concentrations continue to rise. The removal of livestock grazing from these systems, especially during the early summer when the combination of temperature and soil water is at an optimal level for methane production, may reduce methane emissions. However, nitrate levels in spring waters increase with grazing removal, and preliminary data shows that grazing removal also increases the production of the greenhouse gas nitrous oxide (N_2O).

In addition to introduced cattle, spring-wetland systems are grazed by wildlife of all kinds. Grazing is an integral part of these systems; it evolved with them, and the plants and wildlife grazers (and later domestic grazers) are adapted and continue to adapt to each other. These studies demonstrate the importance of and need for long-term research, and show that tradeoffs exist for different management scenarios and different measured environmental factors.

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Grazing effects on spring ecosystem vegetation of California's hardwood rangelands

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Abstract

Three watersheds at the University of California's Sierra Foothill Research and Extension Center (SFREC), Marysville, Calif. were selected to study cattle grazing effects on the vegetation surrounding cold-water springs and their downslope creeks. Three spring-creek systems from each of 3 watersheds were randomly assigned to grazing treatments (9 total). Treatments were ungrazed, lightly grazed (1,500 kg•ha^{-1} residual dry matter), and moderately grazed (1,000 kg•ha^{-1} residual dry matter) based on degree of use in upland pastures encircling the spring-creek systems. Total herbaceous cover at springs varied significantly among the 6 years only once (greater in 1994 than all others) covarying with previous year's rainfall. Grazing intensity did not affect total herbaceous cover at springs. A year X grazing treatment interaction ($P<0.05$) was detected for total herbaceous cover at spring-fed creeks. Three years after grazing removal, total herbaceous cover on ungrazed creek plots surpassed cover at moderately grazed and lightly grazed plots. Moderately grazed plot herbaceous cover declined steadily throughout the first 3 years, while lightly grazed cover remained relatively stable. Plant community composition and stability by year and grazing treatment were analyzed with TWINSPAN. With few exceptions, stable plant communities persisted on sites regardless of grazing intensity or cover changes. Total herbaceous cover was sensitive to interannual fluctuations, especially under increased grazing intensities. This attribute renders cover a more useful gauge of ecosystem health than plant composition as the latter may not provide evidence of potentially deleterious grazing X climate interactions until after soil erosion or water table characteristics are seriously, perhaps permanently, altered.

Key Words: Riparian, creek, cover, grazing, species composition

Resumen

Se seleccionaron tres cuencas hidrológicas de la Estación de Investigación y Extensión Sierra Foothills de la Universidad de California con el objetivo de estudiar los efectos del apacentamiento del ganado en la vegetación circundante de manantiales de agua fría y sus arroyos pendiente abajo. En cada cuenca, tres sistemas de manantial-arroyos se asignaron aleatoriamente a los tratamientos de apacentamiento (9 en total). Los tratamientos fueron; sin apacentamiento, apacentacentado ligeramente (1,500 kg ha^{-1} de materia seca residual) y apacentacentado moderadamente (1,000 kg ha^{-1} de materia seca residual), los tratamientos se basaron en el grado de utilización de tierras arriba que circundan los sistemas de manatial-arroyos. La cobertura total de la vegetación herbácea en los manantiales varió significativamente solo una vez en 6 años (en 1994 fue mayor que los otros años) y los cambios se correlacionaron con la precipitación del año anterior. La intensidad de apacentamiento no afecto la cobertura total de la vegetación herbácea de los manantiales. En la variable cobertura total de la vegetación herbácea de los manantiales-arroyo, se detecto una interacción año x tratamiento de apacentamiento ($P<0.05$). Después de 3 años de que el apacentamiento se suspendió, la cobertura total herbácea de las parcelas de los arroyos sin apacentar superó a la cobertura registrada en las parcelas con apacentamiento ligero y moderado. La cobertura de herbáceas de las parcelas con apacentamiento moderado disminuyo paulatinamente durante los primeros tres años, mientras que la cobertura de las parcelas con apacentamiento ligero permaneció relativamente estable. La composición y estabilidad de la comunidad vegetal por año y tratamiento de apacentamiento se analizaron con TWINSPAN. Con pocas excepciones, comunidades estables de plantas persistieron en los sitios, independientemente de la intensidad de apacentamiento o los cambios de cobertura. La cobertura total herbácea fue sensible a las fluctuaciones interanuales, especialmente bajo las mayores intensidades de apacentamiento. Este atributo, en comparación con la composición de la comunidad, representa una mejor medida de la salud del ecosistema que este ultima puede no proveer de evidencia del daño potencial de la interacción apacentamiento x clíma hasta que la erosión del suelo o las características de agua freática son seriamente alteradas.

The ecology and management of California's annual grasslands, woodlands, and savannas have been well studied (Bartolome and Standiford 1992, Bartolome and McClaran 1992, Bartolome et al. 1994, Heady et al. 1992, Standiford et al. 1997). However, few studies examine cold-water spring ecosystems of these rangelands, or the potential for grazing effects on them (Allen-Diaz et al. 1998).

Managers assess range condition or health by comparing existing plant community composition to a standard (Dyksterhuis 1949, Busby et al. 1994). Plant species cover is the variable most often estimated for generating range condition classes. Although grazing management changes often effect changes in vegetation cover, Bartolome (1984) and Milchunas and Lauenroth (1993)

The authors thank James Bartolome and 2 anonymous reviewers for review and comments; Irene Timossi for information system management; Mark Spencer, Clay Taylor, Chris Campbell, Cheryl Lovato Niles, Jeff Fehmi and others for field sampling assistance; and Mike Connor and Sierra Foothill Research and Extension Center personnel for range management. The Integrated Hardwood Range Management Program, University of California funded this project.

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suggested that species composition does not necessarily track cover variability and may lead to erroneous conclusions concerning long-term ecosystem productivity. Busby and Cox (1994) suggested that soil degradation and water quality parameters may be more important than plant species composition for assessing ecosystem health.

We tested the effects of different cattle grazing intensities on vegetation surrounding springs over 6 years on California's oak-dominated hardwood rangeland. We hypothesized that reduced grazing intensity would induce plant community composition change. However, we held no a priori notions regarding compositional change direction.

Study Site

Research was conducted at Sierra Foothill Research and Extension Center (SFREC). The study site was described in detail in Allen-Diaz et al. 1998. Briefly, SFREC has an average annual precipitation of 72 cm•yr^{-1}. Air temperatures in the region range from monthly averages of 32.0°C in July to 2.2°C in January. Dominant vegetation is blue oak (*Quercus douglasii* Hook. & Arn.)/gray pine (*Pinus sabiniana* Douglas) woodlands and savannas with introduced annual grass and forb understories. Soils in this area are classified as Auburn (loamy, oxidic, thermic, Ruptic-Lithic Xerochrepts) and Argonaut (fine, mixed, thermic Mollic Haploxeralfs) series (Herbert and Begg 1969).

As described previously, springs and their resultant creeks are quite small (about 0.5 m wide at the source). Wetland species (Table 1 for typical spring spp.) remain green in summer and sharply delineate a spring's boundaries, which extend an average of 3 m perpendicular to flow, forming oval-like borders. The highly palatable, perennial vegetation of spring ecosystems encourages intense cattle use. This is especially evident during summer months as upland annual grasses dry (Table 1 for typical annual grassland spp.). Concentrated utilization of springs creates visually striking vegetation impacts that motivated this study.

Three spring-creek systems were selected from each of 3 SFREC watersheds (Campbell, Schubert, and Forbes) for grazing treatment application (Allen-Diaz et al. 1998). Watersheds were selected for the presence of undeveloped springs, geographic proximity, and similar management histories. These watersheds had all been grazed by cattle at a moderate level (800 to 1000 kg•ha^{-1} residual dry matter) since 1960 when the ranch came under University of California ownership. Prior to this ownership change, it is believed that greater livestock use levels existed (Kinney 1996), however, these use levels have not been quantified. Impediments to wildlife herbivory do not exist, nor do we believe any ever have in the past.

Methods

Grazing treatment

Each pasture within a watershed was randomly assigned a grazing treatment, ungrazed (UG), lightly grazed (LG), or moderately grazed (MG); applied from 1993 through 1997. The approximately 2 ha pastures were grazed by cattle following autumn germination (usually November). Cattle were then removed for the winter and returned to pastures during the spring season annual vegetation rapid-growth phase (February through April). Residual herbage on a dry matter basis (RDM) estimates were then made and cattle were returned to pastures in May to meet target RDM levels (Table 2 for annual RDM estimates). The number of animals placed in a pasture during any period was variable as it was based on available upland forage. Indeed, more animals were often placed on the lightly grazed plots because of site and interannual productivity differences (Allen-Diaz et al. 1998).

Table 1. Partial herbaceous-layer flora of springs and uplands.

Species	Common name	Family
Springs		
Brachypodium distachyon L..	False brome	Poaceae
Briza minor L.	Rattlesnake grass	Poaceae
Carduus pycnocephalus L.	Italian thistle	Asteraceae
Cynodon dactylon L.	Crabgrass	Poaceae
Cynosurus echinatus L.	Dogtail	Poaceae
Cyperus niger Ruiz Lopez & Pavon	Sedge	Cyperaceae
Cyperus odoratus L.	Sedge	Cyperaceae
Eleocharis macrostachya Britton.	--[1]	Cyperaceae
Holcus lanatus L.	Velvet grass	Poaceae
Lolium multiflorum Lam.	Italian ryegrass	Poaceae
Mimulus guttatus DC.	Monkey flower	Scrophulariaceae
Paspalum dilatatum Poiret	Dallis grass	Poaceae
Polypogon monspeliensis L.	Annual beard grass	Poaceae
Rorippa nasturtium-aquaticum Hayek	Water cress	Brassicaceae
Rubus ursinus Cham. & Schldl.	Blackberry	Rosaceae
Stachys albens A. Gray.	Hedge nettle	Lamiaceae
Stellaria media Villars	Common chickweed	Caryophyllaceae
Typha angustifolia L.	Cattail	Typhaceae
Verbena bonariensis L.	--[1]	Verbenaceae
Vitis californica Benth.	California grape	Vitaceae
Uplands		
Avena fatua L..	Wild oat	Poaceae
Bromus diandrus Roth	Ripgut brome	Poaceae
Bromus hordeaceus L.	Soft chess	Poaceae
Bromus madritensis L.	Red brome	Poaceae
Carduus pycnocephalus L.	Italian thistle	Asteraceae
Centaurea solstitialis L.	Yellow star thistle	Asteraceae
Elymus glaucus Buckley	Blue wildrye	Poaceae
Erodium botrys Bertol.	Filaree	Geraniaceae
.*Erodium cicutarium* L'Her. ;	Filaree	Geraniaceae
Galium aparine L.	Common bedstraw	Rubiaceae
Geranium molle L.	Geranium	Geraniaceae
Hordeum murinum L.	Barley	Poaceae
Lolium multiflorum Lam.	Italian ryegrass	Poaceae
Medicago polymorpha L.	California burclover	Fabaceae
Nassella pulchra Barkworth	Purple needlegrass	Poaceae
Phalaris aquatica L.	Harding grass	Poaceae
Taeniatherum caput-medusae Nevski	Medusahead	Poaceae
Torilis nodosa L.	Wild carrot	Apiaceae
Trifolium hirtum All.	Rose clover	Fabaceae
Vulpia myuros C. Gmelin	Annual festuca	Poaceae

[1]No known common name

Table 2. Annual residual dry matter (RDM) estimates made each June from 3 clipped plots in pastures surrounding spring sites to estimate grazing intensity.

		Grazing Year (September-June)						
Watershed	Treatment	1992/3	1993/4	1994/5	1995/6	1996/7	Mean	SE
		(kg•ha^{-1})						
Campbell	UG[1]	2586	1404	2347	1227	1478	1808	274
	LG[2]	1256	462	2240	1066	790	1163	301
	MG[3]	1248	798	1685	1040	742	1103	171
Schubert	UG	6092	3031	5899	---[4]	1915	4234	1043
	LG	1453	1413	3461	1592	878	1759	442
	MG	1597	402	1013	1788	578	1076	272
Forbes	UG	5019	2648	5317	---[4]	3114	4025	670
	LG	5436	1807	5211	3403	3119	3795	680
	MG	2509	1399	2224	1434	1660	1845	222

[1]Ungrazed
[2]Lightly grazed
[3]Moderately grazed
[4]Data not available

Cattle were English-cross yearlings, including Angus, Red Angus, and Hereford breeds that averaged 455 kg (Table 3). Actual use data for pastures surrounding springs were provided in Allen-Diaz et al. (1998). Livestock were managed under a comprehensive SFREC health care program, which included immunization against locally common cattle diseases several times annually.

Livestock grazing during the growing season has little effect on forage composition or productivity on California's hardwood rangelands (Bartolome and McClaran 1992). Grazed California annual grass rangelands are managed so that target residual dry matter levels are achieved at the end of the annual grass growing season (late May) or before the new rains start (September) (Bartolome et al. 1980, Heady et al. 1992). Domestic grazing animals are generally allowed to graze rangeland pastures on a semi-continuous or periodic rotational basis so that *at the end of the growing season* RDM target levels for the grassland are met. Animal management objectives include maintaining or enhancing body weight, and animal numbers are adjusted depending on the quality and quantity of forage available at any time during the annual grass growth cycle. Other elements of the rangeland landscape, such as springs, are not managed for per se. Rather, these ecosystems experience cattle use levels in proportion to their desirability as green forage and place of water.

Experienced Research Center annual grassland range managers monitored grazing treatment intensity during each treatment period. Cattle were left on a site until a visually estimated residual dry matter (RDM) target level was attained during any grazing period. To quantify grazing treatment levels, upland RDM was measured annually (June) by comparing aboveground herbaceous biomass from 3 randomly-located 0.0625-m^2 clipped quadrats per treatment pasture.

Vegetation sampling

Depending on physical constraints imposed on transect length, a permanent 10-m or 5-m vegetation sampling transect was randomly established on each side of a spring and each side of a creek at each cohort in 1992 and was sampled annually in spring through 1997. Species cover was also estimated quasi-weekly 6 times from 11 April 1997 to 1 June 1997 to examine within-season variability. Transects were run parallel to spring or creek flow. The line-point method (Heady et al. 1959) was used to determine plant species cover 3 weeks after late spring (late May) cattle removal from grazing treatments. Vegetative "hits" were determined according to the first foliar intercept in the herbaceous layer. Total herbaceous cover was calculated as total hits minus non-vegetation hits divided by total hits. Plant species identification and nomenclature followed The Jepson Manual (Hickman 1993).

Statistical analyses

Potential differences in total herbaceous cover at springs and creeks among grazing treatments were assessed with split-plot, repeated measures ANOVA (S-PLUS 1993). Three watersheds comprised a blocking variable that effectively reduced error variance by partitioning site-to-site variability out of the ANOVA model. Each watershed was subjected to 3 grazing treatment levels (whole plot) repeatedly measured over 6 years (and 6 weeks for 1997; split plot). Homoscedasticity was verified for both factors but significant covariance among years indicated non-independence across factor levels rendering univariate analyses inappropriate for significance tests of temporal variation (Winer 1971). Hence, MANOVA was performed on year-wise orthogonal contrasts of total cover for tests of time. Orthogonal contrasts were created by multiplying total cover by a coefficient matrix whose determinant equaled zero. MANOVA was then performed using yearly responses as dependent variables (Venables and Ripley 1997). When significant main or interaction effects were detected, pooled standard errors of the differences among means were examined to ascertain which treatment levels gave rise to significant differences main or interaction effects.

Temporal changes in plant species composition across grazing treatments were assessed with TWINSPAN, a divisive, polythetic classification program (Hill 1979). Default cut-levels were used to classify each pasture-year combination.

Results

Grazing treatments did not affect herbaceous cover at springs. Interannual variation was significant only in 1994, when total herbaceous cover was greater than cover totals in 1995 and 1997 ($P < 0.05$).

Table 3. Cattle abundance and mean weight in each grazing period for 3 treatments. (Actual use dates and number of animals per treatment can be found in Allen-Diaz et al. 1998).

Grazing treatment period	Animal type	Animal weight
		(kg)
November-December 1992	cow	360-405
March–May 1993	cow	360-405
November–December 1993	cow	360-405
March–May 1994	cow	455
May 1994*	steer	225
December 1994–May 1995	cow	455
February 1996–May 1996	cow, calf	455, 214
November 1996–May 1997**	cow, calf, bull	455, 214, 475

*adjustment to reach RDM standards
**7 bulls for 2 days on Campbell LG

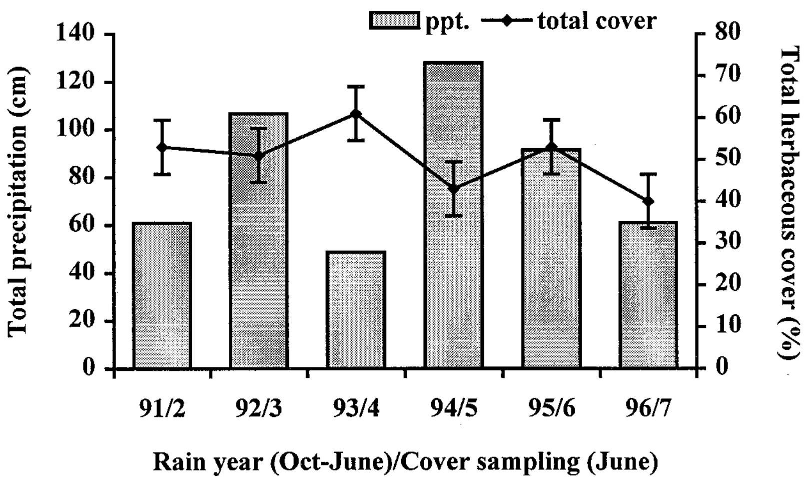

Fig. 1. Annual precipitation and total herbaceous cover at springs.

Precipitation well above average in the 1992/93 rain-year coupled with below average precipitation for the 1993/94 rain year probably contributed to increased cover in 1994 and reduced cover in 1995 (Fig. 1).

A significant grazing X year interaction was determined for total herbaceous cover at creek sites ($P < 0.05$). Grazing treatment groups did not differ significantly for 1992 and 1993 (Fig. 2). But, in 1994, ungrazed (UG) cover increased dramatically from 63% to 86% while the moderately grazed (MG) group declined from 74% to 59% total herbaceous cover. The UG group retained greater cover than MG and lightly grazed (LG) in 1995 and 1996, but not in 1997. The LG group maintained the most stable total herbaceous cover values over the study period ranging between 45–55%. The MG group cover ranged from 80% at the beginning of the study (1992) to 34% one year after the lowest rainfall year (1994).

TWINSPAN classification analysis showed that only one spring transitioned to a new community type, a result of *Rubus* spp. expansion at the Forbes moderately grazed (MG) spring plot (Fig. 3). The transition occurred between 1992 and 1993 and has an associated eigenvalue (λ) of 0.434 indicating a high *goodness-of-split* (Jongman et al. 1995). All other splits were based on dominant tree presence (*Salix* spp., λ = 0.556), *Typha angustifolia* L. presence (λ = 0.474), and *Rhamnus californica* Eschsch. presence (λ = 0.359), which changed little during the study.

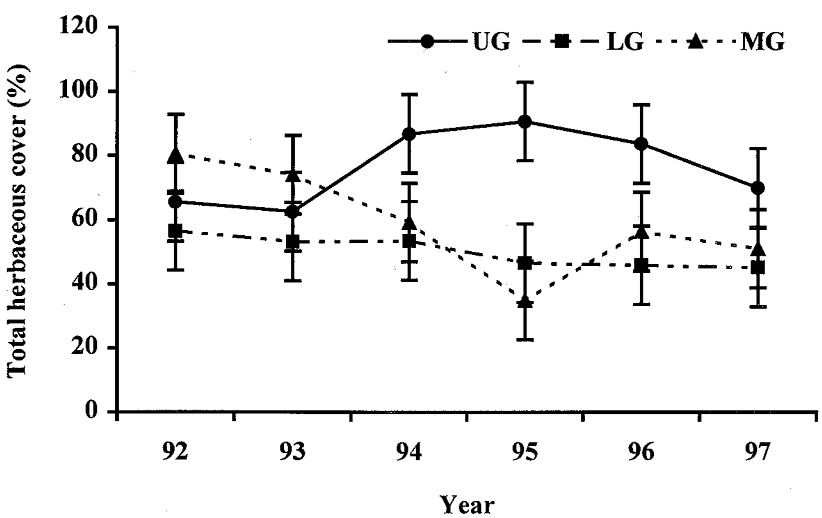

Fig. 2. Total herbaceous cover by grazing treatment at creeks.

Results based on 1997 quasi-weekly sampling showed no grazing treatment effects and no temporal variation in total herbaceous cover at springs. A year X grazing interaction for total herbaceous cover at creeks ($P < 0.01$) reflected a difference between LG and MG for the initial 1997 sampling week (11 April). Lightly grazed and MG groups then stabilized at ~50% herbaceous cover for the remainder of the 1997 6-week sampling period. The TWINSPAN results showed no compositional changes over the 6-week sampling period.

Discussion

Domestic livestock have grazed the Sierra Foothill Research & Extension Center (SFREC) area for about 120 years. Although grazing intensity has certainly varied during this period, springs and their resulting creeks likely experienced periods of intense utilization, especially during summer. With the introduction of SFREC range management (1960), use at springs remained concentrated but less continuous, allowing for vegetative recovery and regrowth of perennial vegetation during rest periods. At SFREC, cattle are moved often, usually dictated by research needs.

While mean residual dry matter (RDM) target levels were met for the 6-year period, moderate- and light-grazed targets were transposed in several years (Table 2). This resulted from difficulties in reconciling visual upland RDM estimates with visual observations of spring use and because the presence of particular species at certain sites (*Typha angustifolia* L., *Rhamnus californica* Eschsch.) somewhat physically deterred riparian use until surrounding annual grasses had been heavily utilized. Moderately grazed and lightly grazed (LG) targets were also occasionally transposed because of upland vegetation regrowth under more favorable weather conditions after animals had been removed. Hence, cattle were sometimes removed from pastures before upland RDM targets were met. We do not believe that the minor interannual differences in RDM target levels affected the overall 6-year results.

In fact, our study continued the 120-year history of periodic, intense use of the spring systems at both LG and moderately grazed (MG) treatment levels, but especially at MG levels. The ungrazed (UG) treatment can be thought of as release from 120 years of MG use. Grazing treatments, including complete protection, did

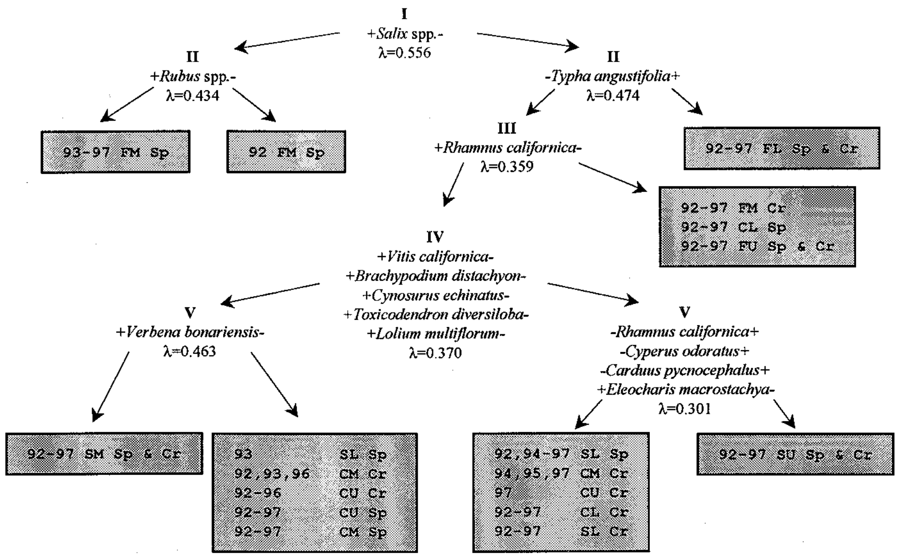

Fig. 3. TWINSPAN classification results. Roman numerals indicate different division levels. Numbers in shaded boxes indicate year, first letter denotes watershed (F=Forbes, S=Schubert, C=Campbell), second letter denotes grazing treatment (M=moderately grazed, L=lightly grazed, U=ungrazed), "Sp" denotes spring-site, "Cr" denotes creek-site. Classification divisions were based on presence (+) or absence (-) of noted species. Strength of a given division is measured by eigenvalue (λ; 0.00-1.00 range).

not alter total herbaceous cover at springs or species composition at springs or creeks during the 6-year study.

The finding that MG plots changed in total cover more among years at creeks and springs than either the LG or UG groups, possibly under the influence of rainfall fluctuations, is compelling. Annual rainfall patterns control variations in species dominance on California's annual grasslands (Pitt and Heady 1978). It appears that these same patterns could similarly influence herbaceous cover in the perennial vegetation of low-flow spring systems, but with a 1-year time lag (Fig. 1). We did not observe interannual rainfall induced shifts in vegetation composition, an anticipated result, because seed bank, edaphic characteristics, and water, which are known to affect annual uplands, are less limiting at spring ecosystems.

We speculate that prolonged drought in moderately grazed situations probably reduces herbaceous cover at both springs and creeks, but without an immediate impact on plant community composition until some low-cover threshold is crossed (sensu Bartolome 1984). We hypothesize that soil loss and decreasing water table depth would eventually produce compositional changes on moderately grazed systems, given drought conditions. As spring flows seasonally waned, the changing pattern of water table drawdown shown to affect composition in wet meadows (Allen-Diaz 1991) should eventually induce compositional transitions. Shallow-rooted annual species would move into formerly saturated areas. It is unknown whether these changes would be fluid enough that a return to pre-drought spring-flow would result in a linear, reverse transition to pre-drought species composition and spatial extent. However, given that this system has been moderately grazed for 120+ years, enduring several droughts, it seems apparent that the plant communities found presently represent some relatively stable result of Mediterranean-type climatic regimes. Monitoring to detect threshold response of these spring-creek systems under continued grazing exclusion and increased grazing intensity (target ~600 kg•ha^{-1} RDM) will continue.

Instead of showing more sensitivity to grazing, ample water supply seems to buffer deleterious grazing effects at springs and spring-fed creeks. This buffering apparently stabilizes perennial spring vegetation composition. We conclude that species composition is not sensitive to existing grazing systems that include periodic intense use coupled with overall moderate grazing levels on uplands. Instead, spring vegetation composition appears resistant to changes in both grazing and climatic conditions.

However, total vegetative cover appears to be a more useful metric for abiotic and biotic resource conservation in these systems. We base this conclusion on our finding that total herbaceous cover is sensitive to weather variability, especially under higher grazing intensities. Because total cover is directly linked to erosion rates and hydrologic processes (Busby et al. 1994, Busby and Cox 1994), total cover may prove to be an appropriate monitoring parameter for these systems.

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CONTROLLED EXPERIMENTS ON SOIL COMPACTION PRODUCED BY OFF-ROAD VEHICLES IN THE MOJAVE DESERT, CALIFORNIA

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SUMMARY

(1) Soil compaction is being caused by intensive use of off-road recreational vehicles in the California deserts. Vehicle tracks produced by different numbers of motorcycle and four-wheel drive passes were made on the Mojave Desert of California on both a Typic Haplargid and a Typic Torripsamment soil in an attempt to estimate minimum amounts of soil compaction which may produce significant reductions in growth of desert annual plants.

(2) A motorcycle produced much smaller increases in soil strength than did a four-wheel drive vehicle.

(3) Soil strength of drying compacted soil (even slightly compacted soil) increased at a much greater rate than soil strength of drying uncompacted soil. This may be an explanation for observed reductions in annual desert plant growth even on areas with a relatively small amount of compaction.

INTRODUCTION

Off-road motorcycle riding has increased greatly since the early 1960s. At the beginning of the decade there were less than 400 000 registered motorcycles in the United States (Sheridan 1979). By 1976, 8·3 million motorcycles were reported to be in use, with 5·4 million, or 66% in use off-road at some time (Motorcycle Industry Council, Inc., 1977). Numbers of other types of off-road vehicles (ORVs) were less certain but Sheridan (1979) estimated that over 3 million four-wheel drive vehicles were in operation in the United States with perhaps half used regularly for off-road driving. Because much of this activity occurs on public land, there has been concern about damage to these areas. Presidential Executive Orders 11644 and 11989 require agencies responsible for management of public lands, in the U.S.A., to adapt policies which protect public resources, including soil.

Damage to the soil caused by ORVs was due in part to soil compaction. Off-road vehicle activity has been shown to greatly increase soil compaction in areas of intense traffic (e.g. Wilshire & Nakata 1976; Wilshire, Nakata, Shipley & Prestegaard 1978; and Webb, Ragland, Godwin & Jenkins 1978). J. A. Adams & A. S. Endo (unpublished BLM data) used aerial photographs to estimate total areas, in the California deserts, which have had the largest amounts of intense soil compaction due to off-road vehicles, such as motorcycles and four-wheel drive vehicles. Out of a total of around 10 100 000 ha of desert land, approximately 495 ha (0·0049%) were estimated to be highly compacted campsites, or pit areas, which were virtually devoid of vegetation; 2406 ha (0·024%) were estimated to contain compacted trials on hill climbs; and 16 391 ha (0·16%) were estimated to have a

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relatively high frequency of highly compacted motorcycle and four-wheel drive trails on more level terrain. Surface area of highly compacted trails, in the last category, generally ranged from 5 to 10%. Even though a very low percentage of the desert has received substantial amounts of soil compaction, localized effects in some areas are considerable. These result in scars on the landscape (especially on hill climbs) which are highly visible, and for the most part denuded of vegetation. Regrowth and revegetation are particularly slow on the compacted soil.

The purpose of our research was to investigate the minimum amounts of off-road motorcycle or four-wheel drive vehicle driving which would cause significant reduction in the establishment and growth of desert annuals, in subsequent years. Soil strengths under tracks created by single or increasing numbers of vehicle passages were measured and related to later responses of desert annuals. Annual plant responses will be discussed in detail in another paper. Increases in soil strength with drying were also characterized in tracked and untracked soil. The relationships between soil water and soil strength need further study to better understand the implications of increased resistance to root growth of desert annuals during periods of soil drying.

MATERIALS AND METHODS

The study sites

Field studies were initiated in 1977 in the Mojave Desert of southern California and continued up to 1979. Soil compaction was produced by driving over both wet and dry desert soil with different types of off-road vehicles at five sites. Illustrative data from the two sites with the most homogeneous values of soil strength in untracked soil are presented in this paper. Site one in Stoddard Valley has coarse loamy, mixed, thermic Typic Haplargid soil with surface textures of loamy coarse sand. The top of the argillic horizon is at a depth of about 60 cm. In the top 60 cm, the soil averages 5% clay, 19% silt, and 76% sand. Site two in Johnson Valley has mixed, thermic, Typic Torripsamment soil with surface textures of coarse sand. Little profile development is apparent and the top 60 cm of soil has average values of 1% clay, 8% silt, and 91% sand.

Methods of compacting the soil

A four-wheel drive 1975 Ford 'Bronco' and a 1973 Yamaha 175 cm^3 DT2 'Enduro' motorcycle were used to produce soil compaction. The Bronco had tyres which were 19·7 cm wide and inflated to 2·7 kg cm^{-2}. The vehicle and driver together weighed 2190 kg. The motorcycle had a front tyre and a 'knobby' back tyre which were 8·3 and 10·2 cm wide, respectively. The motorcycle and driver together weighed 188 kg.

The Bronco and motorcycle were driven across the study plots at a steady speed of about 15 km h^{-1} to produce sets of tracks which consisted of one, three, five, ten, twenty and 100 (the last set for the motorcycle only) vehicle passes. The multiple passes along the same track were made with as little lateral spread as possible. Typical track widths are shown in Table 1. A large part of this spread resulted from greater divergence of a small proportion of the passes used to make a track. Most vehicle passes and all subsequent penetrometer measurements (see below) were made close to the centre of the tracks.

Measurements of soil strength

Soil strength was measured in the field with a hand-held recording penetrometer (Carter 1967) using a 1·27 cm^2, 30° cone tip. According to Taylor (1971, 1974) the mechanical

TABLE 1. Track widths created by different numbers of four-wheel drive Ford 'Bronco' and motorcycle passes

	Width of track (cm)	
No. of passes	Bronco	Motorcycle
1	23	13
3	30	15
5	30	15
10	40	20
20	45	30
100	–	50

resistance of soil to expanding roots is best characterized by penetrometer soil strength measurements. Measurements were made when the soil was near field capacity (in order to produce the most comparable measurements) and later when the soil had become drier. Soil moisture was measured as mass wetness by drying samples for 24 h at 105 °C. Values of soil mass wetness are given in Tables 2–5. Soils under shrub canopies were avoided

TABLE 2. Mean soil strength (kg cm^{-2}) produced on 19 August 1977 by different numbers of Ford 'Bronco' passes on wet soil at site 1 and measured on three successive dates when mass wetness values (MW) for the top 30 cm of soil in the control were 6·0%, 5·1% and 1·8% respectively

(a) 19 August 1977; MW = 6·0%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	4·4 Z*	9·2 Z	7·9 Z	5·5 Z	6·8 Z	8·0 Z
1	9·7 Y	10·3 Z	11·6 Z	9·7 Z	8·5 ZY	8·3 Z
3	13·6 X	18·5 Y	17·9 Y	16·2 Y	11·3 YX	13·1 Z
5	12·9 X	20·2 Y	21·9 Y	16·8 Y	13·3 X	15·7 Z
10	16·2 X	26·1 X	22·4 Y	18·0 Y	12·9 X	12·5 Z
20	19·8 W	31·3 W	30·4 X	23·4 X	18·9 W	14·8 Z

(b) 26 August 1977; MW = 5·1%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	2·2 Z	11·3 Z	13·4 Z	13·4 Z	9·2 Z	10·5 Z
1	6·3 Z	16·6 ZY	16·7 Z	15·6 ZY	13·5 Z	10·6 Z
3	4·9 Z	21·9 YX	27·1 Y	23·4 YX	18·2 ZY	12·5 Z
5	8·0 Z	26·3 X	26·4 Y	24·1 YX	15·5 ZY	14·0 Z
10	5·3 Z	39·2 W	37·7 X	27·7 X	19·3 ZY	14·0 Z
20	8·2 Z	53·1 V	46·8 W	29·8 X	25·6 Y	14·9 Z

(c) 6 September 1977: MW = 1·8%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	12·3 Z	21·1 Z	20·4 Z	18·8 Z	19·2 Z	19·3 Z
1	30·6 Y	49·5 Y	45·4 Y	29·9 Z	22·5 Z	17·2 Z
3	39·3 Y	55·1 Y	45·0 Y	27·1 Z	18·3 Z	12·8 Z
5	44·1 Y	54·9 Y	50·0 Y	32·6 Z	19·2 Z	15·6 Z
10**	–	–	–	–	–	–
20**	–	–	–	–	–	–

* Column means followed by the same letter are not significantly different at $P < 0{\cdot}01$ according to the Student–Newman–Keuls procedure.

** The soil could not be penetrated due to its extreme strength upon drying (>67 kg cm^{-2}).

TABLE 3. Mean soil strength (kg cm^{-2}) produced on 24 January 1978 by different numbers of motorcycle passes on wet soil at site 1 and measured on three successive dates when mass wetness values (MW) for the top 30 cm of soil in the control were 6·3%, 3·7% and 3·2% respectively

(a) 24 January 1978; MW = 6·3%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	1·2 Z*	5·9 Z	8·7 ZY	6·9 ZY	6·5 ZY	7·4 Z
1	0·7 Z	5·3 Z	6·1 Z	5·8 Z	6·7 ZY	7·9 Z
3	1·0 Z	8·6 Y	8·2 ZY	7·8 ZY	6·7 ZY	7·9 Z
5	3·2 Z	10·3 YX	9·1 ZY	6·6 Z	5·3 Z	7·4 Z
10	2·3 Z	13·9 W	9·3 ZY	10·3 YX	6·7 ZY	6·7 Z
20	3·3 Z	11·7 XW	10·6 Y	8·0 ZY	6·9 ZY	6·8 Z
100	12·5 Y	21·7 V	17·4 X	11·5 X	8·9 Y	9·5 Z

(b) 20 April 1978; MW = 3·7%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	1·5 Z	10·3 Z	10·0 Z	9·4 Z	8·5 Z	11·3 Z
1	1·2 Z	10·4 Z	9·2 Z	7·4 Z	7·2 Z	10·3 Z
3	–	–	–	–	–	–
5	3·6 Z	12·2 Z	12·1 ZY	15·5 Z	11·2 Z	8·2 Z
10	6·4 Z	25·9 Y	17·5 ZY	11·4 Z	10·1 Z	10·5 Z
20	9·5 Z	27·9 Y	19·5 Y	14·4 Z	11·5 Z	12·5 Z
100	39·2 Y	40·6 X	21·1 Y	14·2 Z	13·9 Z	9·5 Z

(c) 23 August 1978; MW = 3·2%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	2·7 Z	14·7 Z	13·9 Z	12·9 Z	10·1 Z	16·7 Z
1	2·9 Z	19·2 ZY	17·1 Z	14·5 Z	11·8 Z	13·4 Z
3	4·9 ZY	30·9 ZY	21·5 Z	12·4 Z	12·5 Z	17·6 Z
5	3·8 ZY	35·2 Y	30·9 Z	21·3 Z	15·3 Z	16·0 Z
10	7·1 ZY	28·2 ZY	22·1 Z	13·4 Z	7·5 Z	7·3 Z
20	10·6 Y	35·0 Y	21·0 Z	20·7 Z	20·3 Z	14·8 Z
100**	–	–	–	–	–	–

* Column means followed by the same letter are not significantly different at $P < 0{\cdot}01$ according to the Student–Newman–Keuls procedure.

** The soil could not be penetrated due to its extreme strength upon drying (>67 kg cm^{-2}).

when making penetrometer measurements, whether in tracked or untracked soils. The sandier soils under shrubs at sites one and two occupy much less total area than intershrub soils and have much lower values of soil strength when tracked or untracked than the corresponding controls or treatments between shrubs.

Soil strengths at depths of 5, 10, 15, 20, 25, and 30 cm below the surface were analysed statistically with the Student–Newman–Keuls Test (Sokal & Rohlf 1969).

RESULTS

Comparisons of soil strengths produced by passes of the motorcycle and Bronco

The Bronco produced greater increases in soil strength than did the motorcycle when both were driven over wet soil at site one or site two an equivalent number of times (Tables 2 and 3 show data for site one). The increases in soil strength produced by driving the Bronco on wet soil at site two (Table 5) were not as great as those produced by driving the

TABLE 4. Mean soil strength (kg cm^{-2}) produced by different numbers of Ford 'Bronco' passes on dry soil (mass wetness in the top 30 cm of soil was 0·8%) at site 1 and measured on three successive dates when average soil mass wetness values (MW) for the top 30 cm of soil in the control were 6·3%, 3·7% and 3·2% respectively

(a) 30 December 1977; MW = 6·3%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	2·6 ZY*	5·3 Z	4·7 Z	5·7 Z	5·6 Z	6·7 Z
1	0·9 Z	3·9 Z	3·9 Z	3·0 Z	4·5 Z	5·6 Z
3	1·4 Z	4·7 Z	4·8 Z	4·3 Z	4·4 Z	5·3 Z
5	4·5 Y	6·1 Z	5·8 Z	5·5 Z	6·9 Z	5·7 Z
10	7·3 X	6·8 Z	4·7 Z	4·8 Z	4·0 Z	6·6 Z
20	10·7 W	12·1 Y	8·7 Y	5·8 Z	5·4 Z	5·3 Z

(b) 20 April 1978; MW = 3·7%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	1·5 Z	13·1 Z	13·1 Z	9·8 Z	9·1 Z	9·0 Z
1	2·6 Z	10·4 Z	9·4 Z	7·7 Z	7·4 Z	7·4 Z
3	5·0 Z	14·2 Z	11·9 Z	9·0 Z	8·3 Z	9·7 Z
5	16·8 Y	21·3 Y	12·2 Z	10·4 Z	8·8 Z	9·8 Z
10	10·8 ZY	22·1 Y	12·0 Z	7·9 Z	8·1 Z	9·5 Z
20	17·5 Y	26·3 Y	14·3 Z	8·3 Z	6·7 Z	7·5 Z

(c) 23 August 1978; MW = 3·2%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	4·0 ZY	14·5 Z	14·0 Z	12·1 ZY	9·9 ZY	14·9 Z
1	1·1 Z	14·7 Z	16·1 Z	11·0 Z	9·3 Z	13·8 Z
3	4·8 ZY	24·2 Y	20·1 Z	16·4 ZY	14·1 ZYX	14·7 Z
5	10·1 YX	29·7 Y	22·3 Z	18·2 ZY	17·1 YX	20·3 Z
10	10·1 YX	52·2 X	35·8 Y	18·2 ZY	18·3 X	17·5 Z
20	14·5 X	48·1 X	36·5 Y	20·5 Y	14·8 ZYX	12·4 Z

* Column means followed by the same letter are not significantly different at $P < 0{\cdot}01$ according to the Student–Newman–Keuls procedure.

Bronco on wet soil at site one, possibly because the wet sand at site two may have compacted less readily than the wet loamy sand at site one. Bodman & Constantin (1965) reported that loamy sands were the soils most susceptible to density increases under loading.

Increases in soil strength produced at different soil depths

All statistically significant increases in soil strength compared to the controls occurred within the top 25 cm. Driving the Bronco on wet soil compacted the soil to a greater depth than driving either the Bronco on dry soil or the motorcycle on wet soil.

Tracks produced by driving the Bronco on wet soil at site one (Table 2(a)) showed significant differences in soil strength with as little as three passes at a depth of 25 cm. However, driving the Bronco on dry soil at site one produced no significant differences below 15 cm (Table 4(a)) and significant differences in soil strength at a depth of 15 cm were produced only after twenty passes. Driving the motocycle on wet soil at site one produced significant differences in soil strength between the control and 100 pass tracks (Table 3(a)) down to 20 cm but not below.

TABLE 5. Mean soil strength (kg cm^{-2}) produced by different numbers of Ford 'Bronco' passes on wet soil at site 2 measured on two successive dates when average soil mass wetness values (MW) for the top 30 cm of soil in the control were 6·0% and 1·3%, respectively. Average MW at the time of compaction (December 30 1977) was 3·3%

(a) 1 January 1978; MW = 6·0%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	1·3 Z*	5·1 Z	6·5 Z	7·2 Z	8·7 Z	12·1 Z
1	1·8 Z	5·7 Z	7·5 Z	8·8 Z	9·7 Z	10·5 Z
3	2·7 Z	8·9 Y	12·4 Y	12·4 Y	11·1 ZY	8·5 Z
5	4·5 Z	12·1 X	15·5 X	15·9 YX	11·6 ZY	11·1 Z
10	3·9 Z	12·6 X	17·3 X	18·4 X	14·8 Y	13·9 Z
20	8·9 Y	17·6 W	20·5 W	16·2 YX	12·6 ZY	14·4 Z

(b) 20 April 1978; MW = 1·30%

No. of passes	Depth (cm) 5	10	15	20	25	30
0	4·1 Z	17·6 ZY	19·9 Z	15·7 Z	14·6 Z	15·6 Z
1	8·7 ZY	14·3 Z	17·4 Z	16·5 Z	14·1 Z	11·9 Z
3	12·3 Y	24·5 Y	27·1 Z	20·6 Z	14·2 Z	11·6 Z
5	25·1 X	48·8 X	49·6 Y	33·3 Y	19·3 Z	16·2 Z
10**	–	–	–	–	–	–
20**	–	–	–	–	–	–

* Column means followed by the same letter are not significantly different at $P < 0.01$ according to the Student–Newman–Keuls procedure.

** The soil could not be penetrated due to its extreme strength upon drying (>67 kg cm^{-2}).

For twenty or fewer motorcycle passes, significant differences in soil strength between the control and treatments only occurred at a depth of 10 cm (but not below this depth).

Tracks made with the Bronco on the wet sand at site two caused significant increases in soil strength as deep as 25 cm below the soil surface, when compared to the control (Table 5(a)). Larger significant increases in compaction occurred at a depth of 25 cm in the wet soil at site one (Table 2(a)) than at site two.

In addition to possible differences in compactability of the sand at site two compared to the loamy sand at site one the wetting front from previous rainfall had an average depth of about 25 cm when Bronco tracks were made at site two (Table 4(a)) compared to an average depth of around 45 cm when Bronco tracks were made at site one (Table 2(a)). The more shallow wetting depth at site two may have limited the depth of compaction.

Relationships between compaction and soil drying

As the soils of both the compacted and the control areas (tracked and untracked) became drier, with time, the rate of increase in soil strength was much greater in the compacted zones (Tables 2–5).

Rates of increase in soil strength after 19 August 1977 (Table 2(b)) were much greater under tracks created by higher numbers of four-wheel drive passes, and therefore compacted to a greater degree initially, than under tracks created by smaller numbers of passes. By 6 September 1977 (Table 2(c)), 20 days after a 3·25 cm rainfall, even the area compacted by a single vehicle pass had a significantly higher soil strength than the control. Strengths under the ten and twenty pass treatments exceeded 67 kg cm^{-2} (the operational ceiling of the penetrometer) on 6 September 1977. The only exceptions to the increases in soil strength with time were measured at depths of 5 cm on 26 August 1977. This was probably due to a small amount of rainfall, 0·20 cm, measured on 23 August 1977, about 6 km north of the site.

The relationship between soil strength, soil depth, days after significant rainfall (3·25 cm of rainfall measured about 6 km north of the site), and number of transits in a vehicle track were analysed for the Bronco tracks made on wet soil at site one (Table 2) by multiple linear regression as $S = 5{\cdot}83 - 0{\cdot}11(D) + 0{\cdot}88(R) + 3{\cdot}23(T)$; $r = 0{\cdot}68$ where:

S = Soil strength (kg cm^{-2})	$r^2 = 0{\cdot}46$ so about 54% of the
D = Depth (cm)	variation in soil strength
R = Number of days after a rain	remains unexplained.
T = Number of vehicle passes	

Some soil strength values under the one, three, and five pass tracks were over 67 kg cm^{-2} on 6 September 1977. The equation represents an underestimate of increased strength with compaction and drying.

Large increases in soil strength of drying compacted areas compared to drying control areas also occurred in the sandy soils at site two (Table 5). Single pass tracks remained close in value to the control after drying (Table 5(b)) but tracks created by five vehicle passes had very large increases in strength compared to controls between 12 January 1978, and 20 April 1978. All values of soil strength in the tracks created by ten and twenty passes exceeded 67 kg cm^{-2} on 20 April 1978.

Tracks made by the Bronco, on dry soil at site one (Table 4), showed large increases in soil strength when compared to untracked soil following subsequent wetting and drying. Soil strength values from single pass tracks of the motorcycle on wet soil and the Bronco on dry soil were similar to the controls even under the driest conditions (Tables 3(c) and 4(c)). Mean values for multiple pass tracks of the Bronco on dry soil and the motorcycle on wet soil increased substantially in comparison to the controls as the soil became drier. The 100 pass motorcycle tracks reached soil strengths in excess of 67 kg cm^{-2} on 23 August 1978.

DISCUSSION

Soil strengths exceeding 20 kg cm^{-2}, when measured at about field capacity, have been reported to cause very limited root extension of alfalfa (*Medicago sativa* L.), corn (*Zea mays* L.), and cotton (*Gossypium hirsutum* L.) (Grimes, Miller & Wiley 1975 and Grimes, Sheesley & Wiley 1978). Desert annuals would be subjected to substantial periods of drying more often than agricultural crops. Mirreh & Ketcheson (1972) reported that soil strength in laboratory tests increased at a greater rate with decreasing matric potential at higher bulk densities than at lower bulk densities. They stated that as soils dry beyond the tensiometer range the decrease in cross-sectional area of interstitial water becomes so great that water bonds are lost and soil strength increases less rapidly with further decreases of matric potential. This may eventually result in no additional strength increases and could even result in a decrease in strength with further drying of the soil. In compacted soil the greater proportion of small pores causes larger amounts of interstitial water to be retained as matric potential decreases. The greater numbers of water bonds remaining in compacted soil during drying produce a larger increase in strength as matric potentials decrease. The much greater rate of increase in soil strength of compacted soil when drying than in non-compacted soil shown in our experiments may be related to the same phenomena. There may be greater water content in drying compacted soil than in drying non-compacted soil because of reduced plant growth and transpiration on the compacted soil. As an illustration, mass wetness of the top 15 cm of soil under the twenty pass track had an average value of 2·9% (5·3 cm^3 cm^{-3} volume wetness) on 6 September 1977; whereas the top 15 cm of adjacent untracked soil had an average mass wetness of 1·1% (1·8 cm^3 cm^{-3} volume wetness). The track had a large reduction in growth of annuals in the summer of 1977 which may have resulted in a larger water content. Higher values of water content in tracked soil as compared to untracked soil also occurred in some of the tracks created by smaller numbers of passes which also had significant reductions in density of annuals.

Merrill & Rawlins (1979) concluded that differences in soil strength associated with the different irrigation treatments appeared to be the predominant factor controlling root distribution of sorghum plants (*Sorghum bicolor* (L.) Moench.) grown in lysimeters with three different irrigation frequencies. Differences in frequencies of desert rains will also produce differences in soil strength which, similarly, should be expected to affect root growth of desert annuals, apart from plant stress effects produced by decreasing soil water potentials *per se*. Where the increases in soil strength with drying are intensified by compaction, the effects of drying can cause even greater reductions in plant growth.

Significant reductions in annual cover compared to controls occurred at site one during April 1979 in tracks created by only one Bronco pass on wet soils, twenty Bronco passes on dry soil, or five motorcycle passes on wet soil. All of these tracks had soil strength (measured near field capacity) which were substantially less than 20 kg cm^{-2}. The great sensitivity of desert annuals to the compaction probably resulted from greater periods of drying during the growing season which caused large increases in soil strength of tracked soil when compared to untracked soil.

Desert annual plant response to compaction varied with seasonal rainfall characteristics. The springs of 1978 and 1979 had very different rainfall patterns. Between 1 December 1978 and 25 May 1979, 9·70 cm of rainfall was measured in Stoddard Valley approximately 6 km north of site one. During 1978, a wetter year with more frequent rains totaling 19·63 cm between 18 December 1977 and 1 May 1978, soil strength would have remained closer to the minimum values (e.g. Table 2(a)). Compacted soil had less reduction in plant growth during 1978 when compared to untracked soil than during the spring of 1979.

Desert annual response also varied with species. Relatively large, taprooted, annual dicotyledons such as *Chaenactis fremontii* Gray and *Erodium cicutarium* (L.) L'Her had significant reductions in cover in all track treatments compared to controls in April, 1979 (Table 2). In contrast *Schismus barbatus* (L.) Thell, a grass with fibrous roots, had significantly higher cover in comparison to controls for one, three, ten, and twenty pass tracks. The coleoptile and fibrous root system of the grass allow greater ease of emergence and root growth, respectively, than is the case for the taprooted dicotyledons. Greater amounts of water available to the grass in compacted soil (because other non-grass species were reduced in density and size) may be another reason for increased growth of grasses in the track.

Because soil water was characterized by mass wetness rather than soil water potential it is difficult to relate the availability of soil water to plants. Annual plants showed no evidence of water stress at an average mass wetness value of 1·8% at site one.

Soils with intensive use by off-road vehicles (e.g. on campsites, pit areas, or vehicle trails) generally had higher measured values of wet soil strength than the maximum values measured under the tracks of our studies. Soil strengths measured in wet soils of campsites frequently ranged from about 35 to over 67 kg cm^{-2}. Trails intensely used by motorcycles, had wet soil strengths which typically ranged from 20–60 kg cm^{-2}. The areas of low to moderate compaction such as we have studied may cover a larger total area than the more highly compacted soils.

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Impacts of Vehicles on Natural Terrain at Seven Sites in the San Francisco Bay Area

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ABSTRACT / The impacts of off-road vehicles on vegetation and soil were investigated at seven representative sites in the San Francisco Bay area. Plant cover of grass and chaparral (with shrubs to 4 m tall) have been stripped by the two- and four-wheel vehicles in use. Impacts on loamy soils include increased surface strength (as much as 275 bars), increased bulk density (averaging 18%) to depths of 90 cm or more, reduction of soil moisture by an average 43% to 30 cm depths, greatly reduced infiltration, extension of the diurnal temperature range by as much as 12°C, and reduction of organic carbon by an average 33% in exposed soils. Very sandy soils respond similarly to vehicular use except that moisture is increased and surface strength of beach sand is decreased.

These physical and chemical impacts reduce the land's capability of restoring its vegetative cover, which in turn adversely affects animal populations. Both the loss of plant cover and the physical changes caused by vehicles promote erosion. Measured soil and substrate losses from vehicular use zones range from 7 to 1180 kg/m^2. The estimated erosion rate of the Chabot Park site exceeds the rate of erosion considered a serious problem by a factor 30, it exceeds United States Soil Conservation Service tolerance values by a factor of 46, and it exceeds average San Francisco Bay area erosion rates by a factor of 17. The resulting soil losses are effectively permanent. Neither the increased sediment yield nor the increased runoff is accomodated on the sites of use, and both are causing adverse effects to neighboring properties.

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Introduction

The rapid increase in recreational use of vehicles on natural terrain is bringing about changes in the landscape that are unmistakable even to the casual observer. However, there are few quantitative studies of the physical, chemical, and biologic impacts of this type of land use (Webb and Wilshire, 1978). This paper covers in detail some of the physical and chemical modifications of the soil in response to use by vehicles at seven sites in the San Francisco Bay area. Quantitiative data are also given on erosion in the areas of use, including the effects of management practices on erosion.

The seven sites were chosen for study from among 99 localities that we have examined in the greater Bay area (Table 1). Use of the land at many of these sites is not authorized. Only four sites (Table 1, Nos. 28, 40, 48, 56) involved pre-use trail planning and preparation. One site (Table 1, No. 40) shows judicious use of existing knowledge of erodibility and techniques for minimizing the loss of soil. Erosional effects have not been successfully contained within the boundaries of any site that we have examined. Within this framework it is our purpose to describe the erosional consequences of those modifications so that they can be used as guides to the environmental costs of off-road vehicular recreation.

Setting of the Sites

The seven study sites, named informally, are Ellicott Station, Montara, China Camp, Pinole, Rio Vista, Chabot Park, and Metcalf Road (Table 1).

Ellicott Station

The Ellicott Station site (Fig. 1) is on a low northwesterly trending ridge and adjacent flats south of Santa Cruz. The ridge is underlain by beach and dune sands and its west face is an ancient sea cliff. At the foot of the ridge is an ephemeral pond that is the breeding ground of an endangered species of salamander (the Santa Cruz long-toed salamander; R. C. Stebbins, personal communication, 1976). A network of motorcycle trails, eroded to depths of more than 2m, on the face of the ridge lies astride the migration path of the salamanders (Fig. 2). Because of this threat to the salamanders the area was acquired in 1976 by the United States Fish and Wildlife Service as an ecologic reserve and closed to vehicular use.

Montara

The Montara site (Fig. 1) is an area of dispersed trails and hillclimbs in rugged chaparral-covered terrain overlooking the coast north of Half Moon Bay (Fig. 3). Motorcycle trails were cut across natural terrain off a secondary road by the vehicles themselves.

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Table 1 Off-Road Vehicle Localities in the San Francisco Bay Area

Locality (informal name)	Township & range	Type of vehicle use	Duration of use	Management for ORV use
Marin County				
1. Novato N	T4N R7W	Motorcycle, 4-wheel	>1 yr	None
2. China Camp	T2N R5W	Motorcycle, 4-wheel	>4 yrs	None, posted
3. Smith Ranch Rd.	T2N R6W	Motorcycle, 4-wheel	>2 yrs	None, posted
4. Marin Civic Center	T2N R6W	Motorcycle, 4-wheel	>2 yrs	None
San Mateo County				
5. San Bruno Mt.	T3S R5W	Motorcycle, 4-wheel	>3 yrs	None
6. Westborough	T3S R5W	Motorcycle, 4-wheel	>3 yrs	None
7. Sneath Lane	T3S R5W	Motorcycle, 4-wheel	3 yrs	None
8. Daly City	T3S R6W	Motorcycle, 4-wheel	>1 yr	None
9. Peninsula Ave.	T4S R4W	Motorcycle, 4-wheel	>1 yr	None
10. Mori Pt.	T4S R6W	Motorcycle	?	None, posted
11. Fassler Ave.	T4S R6W	Motorcycle,	>2 yrs	None
12. Montara	T4-5S R6W	Motorcycle,	>4 yrs	None
13. Montara E.	T4-5S R6W	Motorcycle	>4 yrs	None, posted
14. Moss Beach	T5S R6W	Motorcycle, 4-wheel	>1 yr	None
15. El Granda	T5S R6W	Motorcycle, 4-wheel	>4 yrs	None
16. Whipple Ave.	T5S R3W	Motorcycle	>1 yr	None
17. Sugarloaf	T5S R4W	Motorcycle, 4-wheel	>1 yr	None
18. De Anza Blvd.	T5S R4W	Motorcycle, 4-wheel	>1 yr	None
19. Cordilleras Rd.	T5S R4W	Motorcycle, 4-wheel	>10 yrs	None
20. Edgewood Rd.	T5S R4W	Motorcycle, 4-wheel	>2 yrs	None, posted
21. Canada	T5S R4W	Motorcycle, 4-wheel	>2 yrs	None
22. Carlmont H.S.	T5S R4W	Motorcycle	>1 yr	None
23. Page Mill Rd. #3	T6S R3W	Motorcycle, 4-wheel	>1 yr	None
24. Upper Purisima Creek	T6S R4W	Motorcycle, 4-wheel	>1 yr	None
25. Lower Purisima Creek	T6S R5W	Motorcycle, 4-wheel	>1 yr	None
26. Skylonda	T6S R4W	Motorcycle	>2 yrs	None
27. Skyline #1	T7S R3W	Motorcycle, 4-wheel	>1 yr	None
28. Knob Hill Ranch	T7-8S R4W	Motorcycle	>1 yr	Private motorcycle facility
Santa Clara County				
29. Arastradero Rd.	T6S R3Q	Motorcycle, 4-wheel	>1 yr	None
30. St. Josephs W.	T7S R2W	Motorcycle	>1 yr	None
31. Eastbrook Ave.	T7S R2W	Motorcycle, 4-wheel	>1 yr	None
32. I-280-1	T7S R2W	Motorcycle	>1 yr	None
33. Shotgun Bend	T7S R3W	Motorcycle, 4-wheel	>1 yr	None
34. Black Mt. Rd.	T7S R3W	Motorcycle, 4-wheel	>1 yr	None
35. Neary Ranch	T7S R2W	Motorcycle	1 day, each of 2 years	None
36. Canyon View Dr.	T6S R1E	Motorcycle	4–5 yrs	None
37. Porter Lane	T6S R1E	Motorcycle, 4-wheel	>1 yr	None
38. Simoni Dr.	T6S R1E	Motorcycle, 4-wheel	>1 yr	None
39. Skyview Dr.	T6S R1E	Motorcycle, 4-wheel	>1 yr	None
40. Calabazas Creek	T6S R1W	Motorcycle	>1 yr	None, posted
41. Lafayette St.	T6S R1W	Motorcycle	>2 yrs	Established motocross course
42. Trimble Rd.	T6S R1W	Motorcycle, 4-wheel	>2 yrs	None, posted
43. Robin St.	T6S R2W	Motorcycle	>1 yr	None
44. Mary Knoll	T7S R2W	Motorcycle	>1 yr	None
45. Elena Rd.	T7S R2W	Motorcycle, 4-wheel	>1 yr	None, posted
46. Stevens Creek	T7S R2W	Motorcycle	>1 yr	None, posted
47. Lexington Res.	T8S R1W	Motorcycle, 4-wheel	>2 yrs	None, posted
48. Riverside Golf Course	T8S R3E	Motorcycle	>1 yr	None
49. Metcalf Rd.	T8S R2E	Motorcycle	2 yrs	Designated site on-site supervision

Table 1 Off-Road Vehicle Localities in the San Francisco Bay Area (*Continued*)

Locality (informal name)	Township & range	Type of vehicle use	Duration of use	Management for ORV use
50. Loma Prieta	T9S R1E	Motorcycle, 4-wheel	>1 yr	None
51. Uvas Reservoir	T10S R2E	Motorcycle	>1 yr	None, posted
Alameda County				
55. Livermore E	T2S R3E	Motorcycle	>1 yr	None, posted
56. Chabot Park	T2S R2W	Motorcycle	±19 yrs	Designated site maintained
57. Sunol Blvd.	T3S R1E	Motorcycle	>2 yrs	None
58. Mission Park	T3S R1E	Motorcycle, 4-wheel	>2 yrs	None
59. I-580/I-680	T3S R1W	Motorcycle	>1 yr	None
60. Palomares Creek Rd.	T3S R2W	Motorcycle	>1 yr	None
61. Strobridge School	T3S R2W	Motorcycle, 4-wheel	>1 yr	None
62. Garin Reg. Park	T3S R1W	Motorcycle, 4-wheel	>1 yr	None
63. Union City E	T3S R1W	Motorcycle, 4-wheel	>1 yr	None
64. Cal. State Hayward	T3S R2W	Motorcycle	>1 yr	None
65. Central Blvd.	T3S R2W	Motorcycle, 4-wheel	>1 yr	None
66. Los Positas Rd.	T3S R2E	Motorcycle, 4-wheel	>1 yr	None
67. Coyote Hills S	T4S R2W	Motorcycle, 4-wheel	?	None, posted
68. Coyote Hills N	T4S R2W	Motorcycle	>1 yr	None, posted
69. Durham Rd.	T5S R1W	Motorcycle	>1 yr	None
70. Durham Rd. NE	T5S R1W	Motorcycle	>1 yr	None
Contra Costa County				
71. The Crossings	T1N R1W	Motorcycle, 4-wheel	>1 yr	None
72. Pine Hollow Rd.	T1N R1W	Motorcycle	>1 yr	None
73. Dana Hills	T1N R1W	Motorcycle, 4-wheel	>1 yr	None
74. Treat Lane	T1N R1W	Motorcycle, 4-wheel	>1 yr	None, posted
75. Orinda	T1N R3W	Motorcycle, 4-wheel	>1 yr	None
76. Happy Valley	T1N R3W	Motorcycle, 4-wheel	>1 yr	None
77. Briones Reservoir	T1N R3W	Motorcycle, 4-wheel	>1 yr	None
78. Pleasant Hill W	T1N R2W	Motorcycle, 4-wheel	>1 yr	None
79. Kirker Pass Rd.	T1N R1W	Motorcycle, 4-wheel	>1 yr	None
80. Pt. Richmond	T1N R4W	Motorcycle, 4-wheel	±20 yrs	None, posted
81. Pleasant Hill	T1N R2W	Motorcycle, 4-wheel	>1 yr	None
82. Bear Creek Rd.	T1N R3W	Motorcycle, 4-wheel	>1 yr	None
83. Danville SE	T1S R1W	Motorcycle	>1 yr	None, posted
84. Alta Mesa Dr.	T1S R2W	Motorcycle	>1 yr	None
85. Moraga W. 1	T1S R3W	Motorcycle	>1 yr	None
86. Moraga W. 2	T1S R3W	Motorcycle	>1 yr	None
87. Moraga E	T1S R2W	Motorcycle	>1 yr	None
88. El Cerro Blvd.	T1S R1W	Motorcycle	>1 yr	None, posted
89. Martinez	T2N R2W	Motorcycle	±10 yrs	None
90. HW 4/I-680 SW	T2N R2W	Motorcycle	>1 yr	None
91. HW 4/I-680 NW	T2N R2W	Motorcycle	>1 yr	None
92. Hilltop Dr.	T2N R4W	Motorcycle, 4-wheel	>1 yr	None
93. Appian Way	T2N R4W	Motorcycle, 4-wheel	>1 yr	None
94. Pinole	T2N R4W	Motorcycle, 4-wheel	±12 yrs	None
95. El Portal	T2N R4W	Motorcycle, 4-wheel	>1 yr	None
96. Antioch	T2N R1-R2E	Motorcycle, 4-wheel	>1 yr	None
97. HW 4/HW160	T2N R1E	Motorcycle, 4-wheel	>1 yr	None
Solano County				
98. Rio Vista	T3N R2E	Motorcycle, 4-wheel	>4 yrs	None
99. Fairfield	T5N R1W	Motorcycle, 4-wheel	>1 yr	None, posted

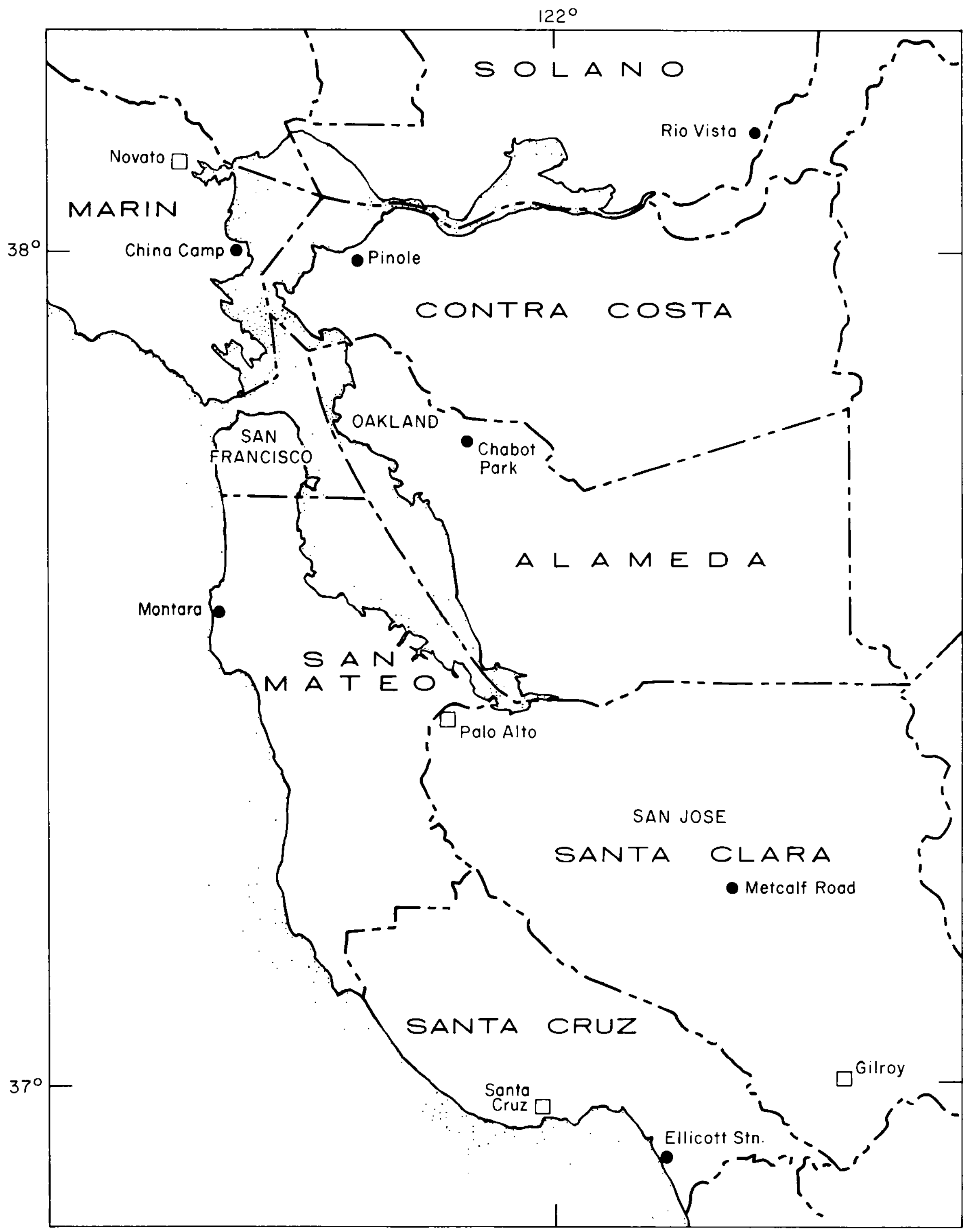

Figure 1. Index map showing locations of the sites studied.

Figure 2. Eroded motorcycle trail at Ellicott Station. This and other parallel trails lie astride the migration path of an endangered species of salamander.

China Camp

China Camp is situated next to the San Francisco Bay east of San Rafael (Fig. 1). A salt marsh interfingers with a riparian habitat. Small outlying hills are completely surrounded by the salt- and freshwater flats and give way inland to rugged hills covered by oak–grass and oak–chaparral (Fig. 4). There are several Indian mounds in the immediate vicinity. Motorcycle and four-wheel vehicular use has been heavy at China Camp, but this is only the hub of an area of use that extends for 3 km or more to the west.

The qualities of the site led to its acquisition by the California State Parks Foundation in 1976, but not before serious damage was sustained from vehicular use: An area of salt- and freshwater marsh interface has been severely rutted and compacted, six Indian mounds have been tracked and eroded, and bedrock is exposed in trails on south- and west-facing slopes.

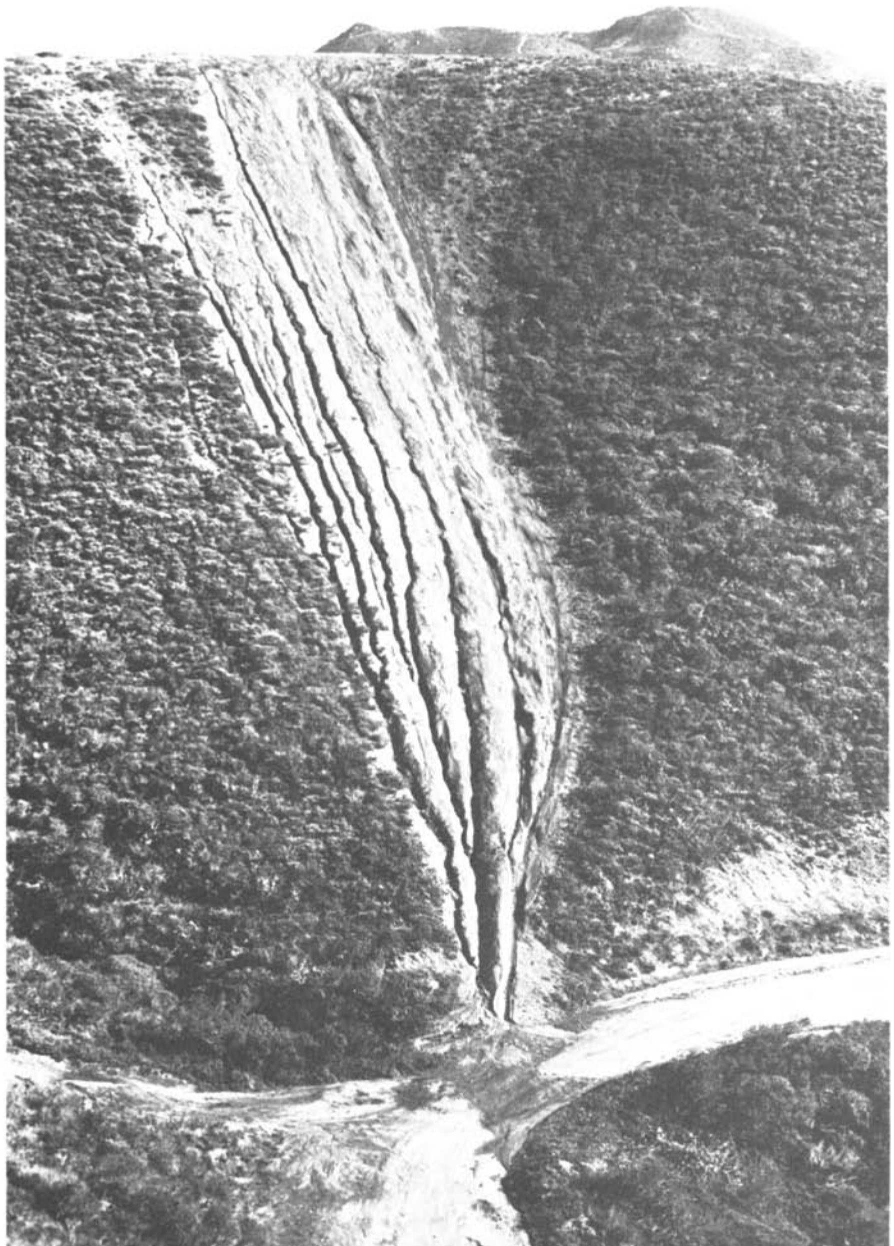

Figure 3. One of numerous motorcycle hillclimbs at the Montara site. The trails were established by the vehicles in the thinner parts of the chaparral cover and are now eroded to depths of nearly a meter.

Pinole

Pinole is a small area (Fig. 1) of two- and four-wheel vehicular use that is nearly surrounded by suburbs and faces Interstate Highway 80 (Fig. 5). The grasslands that have provided easy access to vehicles are now crisscrossed by myriad trails; bedrock is exposed in many of these trails, especially on a northeast-facing slope. A formerly dense chaparral with shrubs as much as 4 m tall is now a "warren" of intersecting trails, first opened by four-wheel vehicles.

Rio Vista

The Rio Vista site (Fig. 1) is located along low grass-covered bluffs facing the Sacramento River in the upper estuary. The area once had a fine beach but the shoreline has been severely degraded by dredging, quarrying, and vehicular use. A conspicuous strand-line sand deposit, the geologic significance of which is as yet unstudied, mantles the lower half of the bluffs (Fig. 6). Hillclimbing has resulted in extensive vehicular "quarrying" of this deposit.

Chabot Park

The Chabot Park site (Fig. 1) is an area designated for motorcycle use in the East Bay Regional Park District. It occupies the top of a narrow chaparral-covered ridge, on the northeast side of which is the Upper San Leandro Reservoir (Fig. 7). Access was originally gained along a powerline corridor and from Redwood Road into small grassland areas. Many of the trails have been opened through the tough chaparral by bulldozing and are maintained the same way, leading to extensive soil loss.

Metcalf Road

Metcalf Road, a 225-acre facility (Fig. 1), is operated by the National Safety Council on Santa Clara County land. The area is of moderately steep terrain on the east side of Santa Clara Valley near San Jose (Fig. 8). The gentler slopes are in oak–grassland, formerly used for grazing, and the steeper terrain is covered by oak–chaparral.

Site Characteristics

The seven sites examined in detail were chosen to illustrate the effects of differences in bedrock, soils, terrain, vegetation, precipitation, duration and type of use, and types of management on the impacts of vehicles.

The variants of bedrock and types of soil are shown in Table 2. Varieties of bedrock include volcanic and plutonic igneous rocks, sandstone, siltstone, shale, unconsolidated marsh deposits, and strand-line sand. These have given rise to 16 types of soil (Table 2); effects of vehicles on eight of these have been examined (Fig. 9).

The terrain used at the seven sites reflects to some extent vehicular capability in that it is diverse and emphasizes moderately steep to steep slopes. Approximate percentages of slopes of different gradients are given in Table 3, along with typical gradients and uninterrupted lengths of individual trails. These are critical factors for evaluating erosion potential.

Figure 4. A small part of the two- and four-wheel vehicular trails at the China Camp site. These south-facing slopes have very shallow soils, which have been completely stripped from extensive areas within the vehicular trails and in gullies formed by excessive runoff from the trails.

Figure 5. Two- and four-wheel vehicle trails cut through grass and chaparral at the Pinole site.

Types of vegetation (Table 4) and their distribution are important characteristics of the sites, because of both the role that vegetation plays in stabilizing the soil and its ability or inability to deter expansion of vehicular use. How different kinds of plants respond to indirect impacts of vehicular use is also important (Webb and others, 1977; Wilshire and others, unpublished data). While vehicles typically gain access into areas with low plant cover, motorcycles have established broad trails in chaparral with shrubs as much as 2 m high, and four-wheel vehicles have broken trail through chaparral with shrubs as much as 4 m high in the Bay area. Access to areas with larger vegetation is commonly gained along little-used or abandoned roads, utility corridors, fire trails, or logging trails.

Precipitation, mostly as rainfall, varies from 39 to 66 cm at the seven sites (Table 4). Detailed rainfall intensity data that are site specific are not available. According to records compiled by Oberste-Lehn in 1976 (unpublished data), rainfall in the San Francisco Bay region generally follows a cycle of 1–4 years duration. However, the second (1977) consecutive drought year does not follow the typical cycle.

Table 1 lists the types of vehicular use identified and the duration of use where known. The Metcalf Road and Chabot Park facilities were developed specifically for motorcycle use, the former in May 1975 and the latter about 19 years ago. The Metcalf Road facility is closely supervised by on-site personnel, whereas the Chabot Park facility has no on-site supervision. Unauthorized expansion of areas of use has been difficult to control even with on-site supervision at Metcalf Road because of the lack of natural barriers to vehicular entry. Unauthorized use at Chabot Park is largely restricted by chaparral that is not easily penetrated by motorcycles. All other areas are unsupervised: trails have been opened by the vehicles according to their capability and natural barriers, the trails are not maintained, and no mitigation of erosion is employed.

Soil Modifications from Vehicular Use

The physical response of soils to vehicular use was measured by determining surface strength, bulk density, soil moisture, and soil temperature. Rates of infiltration were measured at one site. The chemical response was measured by determining organic

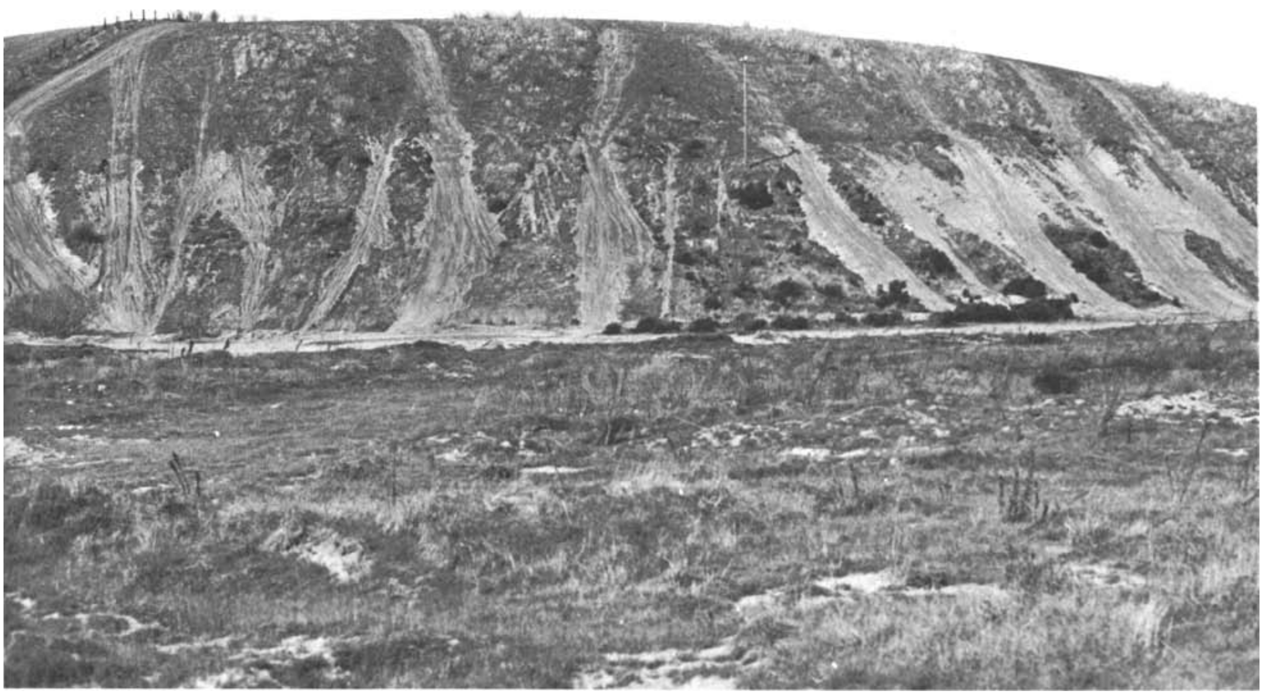

Figure 6. Vehicle hillclimbs at the Rio Vista site. The lower part of the bluff is mantled by sand deposited during a stagnant high-water period; the strand line at the top of the sand deposit is deeply notched by vehicle use.

carbon content. Nutrient variations will be reported in a later publication.

Surface strength

Modification of surface strength by vehicular use was measured by penetrometer readings taken at regular intervals across individual trails. Measurements began in unused terrain on one side of a trail and ended in unused terrain on the opposite side, thus allowing direct comparison of adjacent used and unused soils on the same slope (Fig. 10). At least three such profiles were typically measured at three different elevations along a single trail, and as many as nine were measured on some trails. In all, 39 profiles were measured. In addition, longitudinal profiles were measured that consisted of one measurement in and one out of an individual trail at regular intervals along the trail. This allowed direct statistical evaluation of paired samples (Fig. 11). In general, however, the changes in surface strength are so pronounced that statistical demonstration of differences is not necessary.

It is clear from the data shown in Figs. 10 and 11 that vehicular use has resulted in substantial increase in surface strength for all soil types studied except unconsolidated sand. The surface strengths in heavy use zones range up to 275 bars and are comparable to those of established unpaved roads. Some of the higher values of penetration measured on trails represent temporary accumulations, which are removed with the first rain, of unconsolidated, mechanically eroded debris on hard surfaces. Vehicular use of unconsolidated sand resulted in generally lower surface strength.

The increased surface strength of most soil types is the result of compressive and shear stresses that destroy the open structure of the surface layer of soil. If deeper soil layers had originally higher strengths because of their overburden and lower organic contents, increased surface strength would have resulted from their erosional exposure. The higher in-trail strengths are known, from our measurements, to have persisted for at least 6 years after closure of an area used by motorcycles for only 3 years (Panoche Hills; Synder and others, 1976). The decreased surface

Figure 7. A nearly completely denuded hillside at the Chabot Park site. Remanants of the original soil mantle are visible at the far right of the photograph.

strength of sand resulting from vehicular use may reflect in part loss of cohesive effects of surficial crusts (Wilshire and Nakata, 1976).

Bulk density

The effects of soil compaction were measured by determining bulk density from auger samples of known volume. All samples are paired so that the properties of soils in adjacent used and unused areas can be directly compared. In general three samples were taken at 10-cm vertical intervals at each station, but a number were taken to depths of a meter or more. Paired samples were normally taken at three or more elevations along individaul trails. Bulk density variations determined on more than 130 samples representing eight soil types are given in Table 5 and Fig. 12.

Increases in bulk density caused by vehicular use average about 8% for sandy soil (samples 1, 2, and 5, Fig. 9) and 18% for more silty and clayey soils. Increased bulk density generally occurs to a depth of 30 cm and has been observed in places at depths of 90 cm or more.

The increased bulk densities reflect the effects of vehicular compression of the soil. Experimental studies (Hovland and Mitchell, 1972) indicate that the cylinder of compression beneath a vehicle wheel is sandwiched by a zone of dilation in which shear failure occurs. The zone of compression is much deeper and more comprehensive in its effects on soil structure than is the zone of dilation. With repeated vehicular passes the compressive stresses are generally transmitted to deeper soil layers. Substantial increases in bulk density have been measured to depths of a meter in vehicular trails (Synder and others, 1976). Studies of soil compaction caused by logging operations (Power, 1974) demonstrate persistence of compaction sufficient to inhibit plant growth for 40 years.

Clay-rich soil does not show a consistent pattern of increased bulk density as a result of vehicular use as noted by Webb and

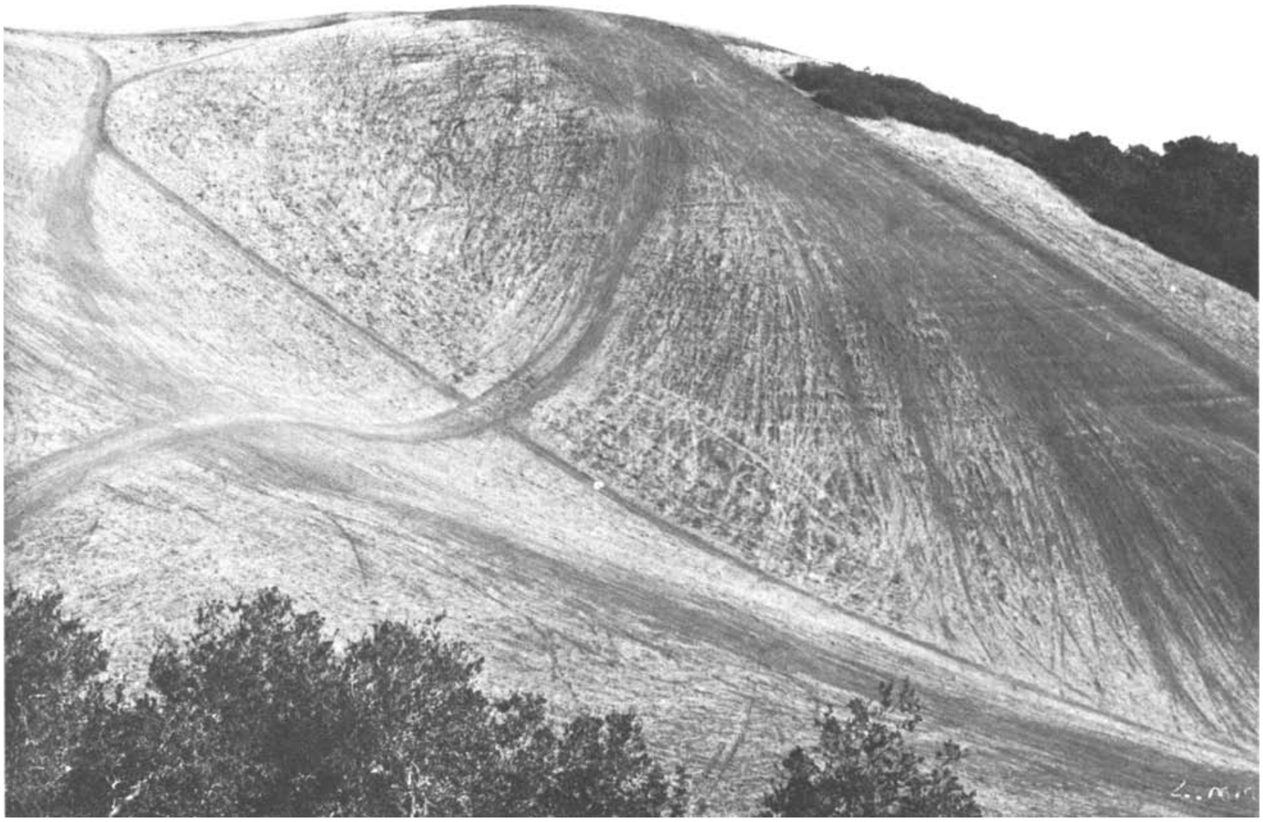

Figure 8. Motorcycle hillclimb at the Metcalf Road motorcycle facility. These soils appear to be more resistant to vehicular use than most, but many trails are worn through to bedrock after only 2 years of use.

others (1977). This may be caused in part by absorption of compressive stress by collapse of soil into desiccation fractures which are much more prominent in the clayey soils than in other soils.

Soil moisture

Soil moisture was determined by weight loss upon drying the bulk density samples (Table 5, Fig. 13). Soil moisture shows substantial reduction as a consequence of vehicular use of the more silty and clayey soils, but generally increased in samples from areas with very sandy soils (Fig. 9, Nos. 1, 2, and 5). Increased moisture in the impacted sandy soils averages 23% to 30 cm depths, and losses in the less sandy soils average 43% to depths of 30 cm.

The reduction of soil moisture in the more common loamy soils reflects changes in structure and composition of the soil brought about by compaction and erosion. The sandy soils, however, also show modest increases in bulk density as well as increased soil moisture. The differences in behavior of sandy soils occur at least in part because vehicular impacts generally destroy fragile surficial crusts in sand, whereas the vehicular impacts create surficial seals on less sandy soils. This would result in ready intake of water in sandy soils and low infiltration, as measured, in other soils. In fact, the surficial seals formed by vehicular use of very clayey soils at Hollister Hills (Webb and others, 1977) caused large soil moisture reductions with no increase of subsurface bulk density, simply by preventing intake of moisture.

Infiltration

The rate of infiltration was measured on adjacent impacted and unimpacted soil only at the Rio Vista site. Five paired tests in and out of trails on silty loam soil were run with a ring infiltrometer (Fig. 14). Measurements were made both before and after

Table 2 Bedrock and Soil Types

Locality	Bedrock lithologies	Bedrock age	Soil series USDA soil surveys	Soil types USDA soil surveys	Soil types this study	Soil thickness and substrate erodibility
Ellicott station	Sandstone	Pleistocene	Pfeiffer Baywood Elder Elkhorn	Sandy loam Loamy sand Sandy loam Sandy loam	Sand	18–30 cm+; bedrock, poorly consolidated and highly erodible
Montara	Granite	Mesozoic	Miramar Sheridan	Coarse sandy loam Sandy loam	Gravelly sand	30–50 cm; bedrock weathered and poorly consolidated to 12–15 m depths
China Camp	Sandstone, siltstone, shale, basalt, marine and marsh sediments, nonmarine alluvium and marsh deposits	Upper Jurassic–Upper Cretaceous, Quaternary	Parrish Laughlin	Gravelly loam Loam	Gravelly sandy loam	20–50 cm; bedrock highly variable in resistance to erosion
Pinole	Siltstone, shale, sandstone	Miocene–Pliocene	Los Osos	Clay loam	Silt loam	10–120 cm; bedrock moderately resistant to erosion
Rio Vista	Nonmarine unconsolidated sand, sandy silt and gravel; unconsolidated alluvial sand	Pleistocene Recent	Montezume	Clay Sand	Silt loam Sand	30–70 cm; older alluvium (Montezuma Fm.) moderately resistant to erosion
Chabot Park	Sandstone, siltstone, shale	Upper Cretaceous	Gaviota Los Gatos Los Osos	Rocky sandy loam Loam Silty clay loam	Loamy sand	25–90 cm; bedrock variable in resistance to erosion
Metcalf Road	Basalt, gabbro, serpentinite, sandstone, shale	Upper Jurassic–Upper Cretaceous	Gilroy	Clay loam	Sandy Loam	10 cm–1 m+; bedrock generally resistant to erosion

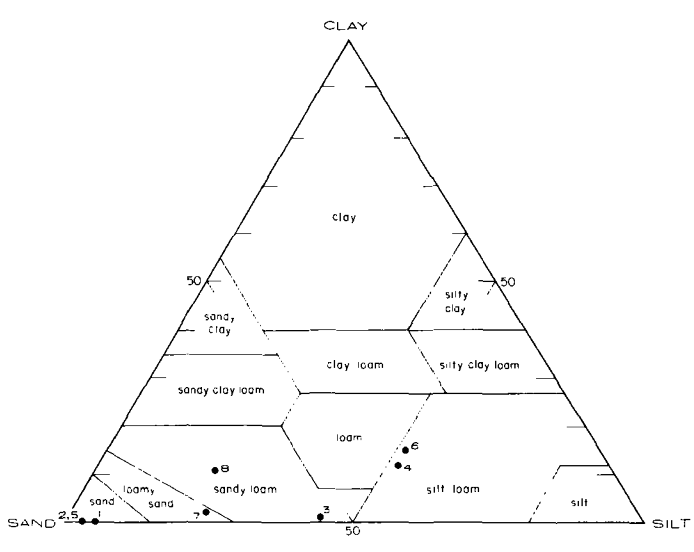

Figure 9. Textural classification of soil at: (1) Ellicott Station; (2) Montara; (3) China Camp; (4) Pinole; (5) Rio Vista (sand); (6) Rio Vista (Montezuma formation); (7) Chabot Park; (8) Metcalf Road. Size analyses by wet and dry sieving and by hydrophotometer; clay/silt break at 0.0039 mm.

removal of leaf litter and roots from the upper 4 cm of the unused terrain, with comparable results. Infiltration rates on unused soil averaged 1 mm/1.6 sec, and those on tracked soil averaged 1 mm/60 sec.

The greatly reduced infiltration rate in vehicle-used soil results from both compressive and shear stresses that act to reduce porosity and permeability of the soil.

Soil temperature

Diurnal variations in temperature were measured at five sites (Table 6, Fig. 15). Simultaneous readings were taken hourly from thermistors emplaced at five different depths in adjacent impacted and unimpacted soils. The most pronounced differences of temperature in impacted and unimpacted soils occurred at shallow depths, but differences were found to the greatest depth measured (18 cm).

The effects of vehicular use are to extend the diurnal temperature range so that the soil is warmer (by 6°–10°C) in the day and cooler (to 2.5°C) at night. The differences diminish at depth, and below about 10 cm there is no significant difference in night temperature in and out of the vehicle-impacted zones. Temperatures remain higher during the day (about 3°–4°C) in impacted soils at depths of 18 cm.

The extended diurnal range in temperature in vehicle-impacted soils is the combined result of loss of the shading and of transpirational effects of plants and the change in insulating

Table 3 Slope Data

Locality	Percent of slope* 0–5	6–15	16–30	31–50	51–70	>70	Typical slopes of ORV trails (%)	Typical slope lengths of ORV trails (m) No break	Between bars
Ellicott Stn.	<1	41	21	34	4	0	30–40	20–80	
Montara	<1	<1	5	25	43	27	60–75	90–100	
China Camp	<1	12	19	31	28	10	40–60	20–60+	
Pinole	<1	6	37	51	6	0	20–60	40–60+	
Rio Vista	66	7	18	9	0	0	0–70	15–70	
Chabot Park	<1	<1	21	35	18	26	25–50	60–97	20–40
Metcalf Rd.	<1	4	37	32	24	3	25–50	25–80+	

*Data from San Francisco Bay Region Slope Map, U.S. Geological Survey (1972).

characteristics of the soil caused by compaction. The importance of the thermal insulation provided by plants is underscored by the very close similarity of our measured variations in soil temperature to the changes in diurnal temperature of stream water in a clear-cut forest (Lee and Samuel, 1976). However, the increased bulk densities increase the heat conductivity and decrease the heat capacity of soil (Shul'gin, 1957), thereby contributing to the diurnal response of temperature. Webb and other (1977) report similar effects on temperature of soils used by vehicles at Hollister Hills.

Organic carbon and pH

Organic and inorganic carbon contents and pH of 26 paired samples representing eight types of soil were determined (Table 7). Samples were taken at different elevations along individual vehicular trails and represent the upper 10 cm of soil in and out of the trails. The samples were split, ground to -75 μm, and filtered to remove soluble salts. Total carbon and inorganic carbon were determined on a LECO-NEBCO analyzer; organic carbon was determined by difference.

The soils in vehicle-impacted areas generally have lower contents of organic carbon than their unused equivalents, although a few samples show no significant difference or opposite results. Reduced content of organic carbon in the upper 10 cm of soil in vehicle-impacted areas averages 42% for sandy soils and 33% for more silty soils. The average reduction for all samples is 36%. These changes reflect that the A horizon has been stripped from the trails, exposing deeper horizons. Although the organic carbon content can generally be expected to be lower in deeper soils than in the A horizon, soil mass-wasted from upper parts of vehicular trails may be incorporated in the soil mantle at the base of slopes by vehicular compaction. This is believed to be the explanation for the higher contents of organic carbon of some impacted soils collected at the base of slopes or low on the slope (Table 7, China Camp, first sample; Pinole, third sample; Rio Vista, first sample; Metcalf Rd., first sample). The second sample under Montara in Table 7 cannot be explained this way.

The soil pH, determined on the same samples used for measurements of organic carbon, shows some change, either to more alkaline or more acidic values, for impacted soils compared to their unused equivalents (Table 7). The changes do not correlate with contents of organic carbon. The chemical changes responsible for these differences are under investigation.

Implications of the Physical and Chemical Modifications of the Soil

The general effects of vehicular use on the physical and chemical properties of the soils are increased surface strength, increased bulk density, decreased soil moisture of the loamy soils, decreased rate of water infiltration, extension of the diurnal temperature range, and reduced content of organic carbon of the exposed soil. Similar, although not identical, physical responses of many soils to vehicular use, whether recreational or agricultural, have been reported (Arndt, 1966; Davidson and Fox, 1974;

Table 4 Vegetation and Precipitation

Locality	Types of vegetation	Mean annual precipitation* Max.	Min.	2 Year 6 hour precipitation* Max.	Min.
Ellicott Stn.	Chaparral, grass, trees	22	20	1.40	1.32
Montara	Chaparral	22	20	1.40	1.32
China Camp	Trees, grass, chaparral	22	20	1.40	1.32
Pinole	Grass, trees, chaparral	22	20	1.40	1.32
Rio Vista	Grass	18	16	1.24	1.16
Chabot Park	Chaparral, grass	26	24	1.56	1.48
Metcalf Rd.	Grass, trees, chaparral	16	14	1.16	1.07

*Data from Rantz (1971).

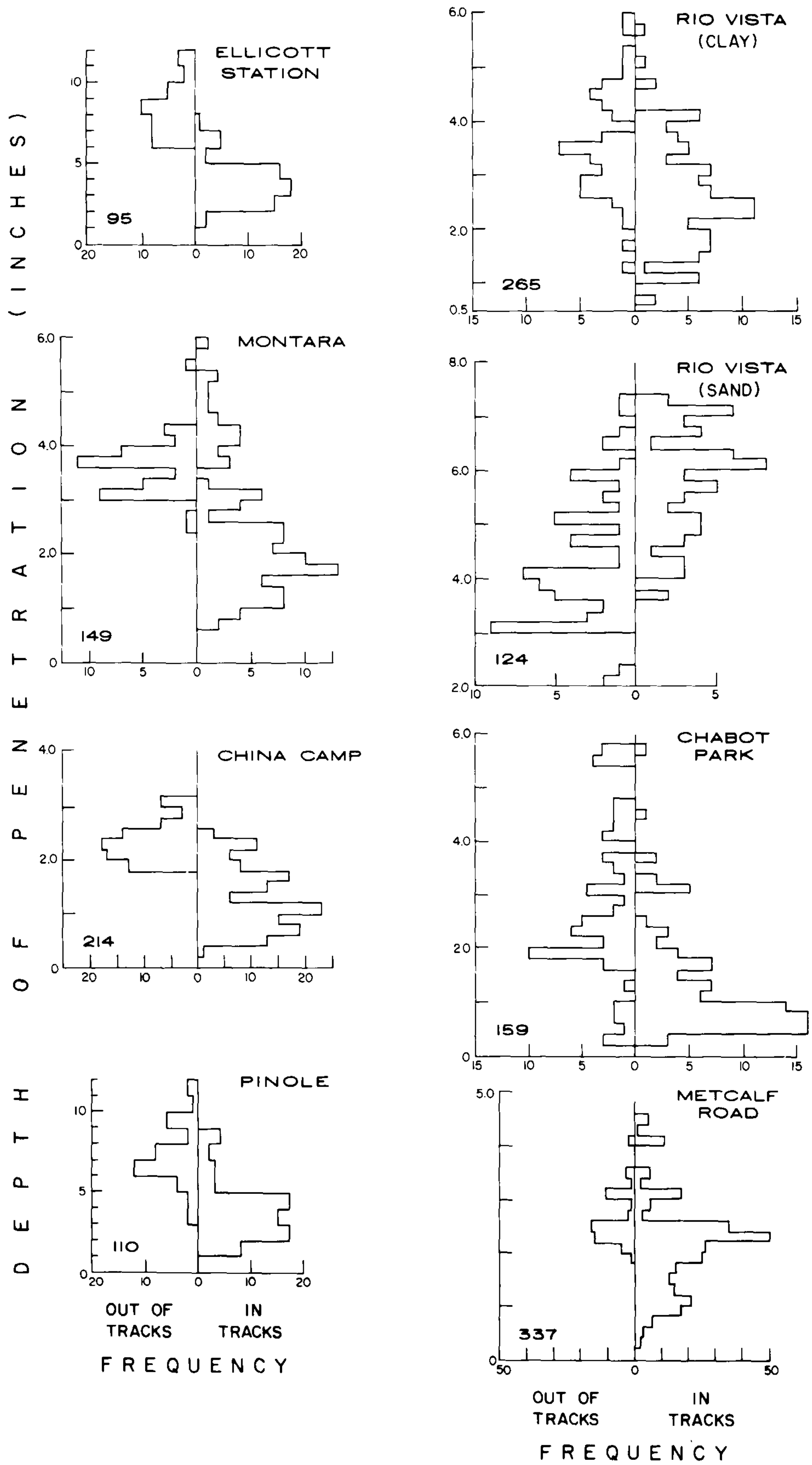

Figure 10. Histograms showing frequency (in percent) of penetrance values (in inches) measured in and out of vehicle trails at each of the localities studied. The number in the histogram indicates the total number of measurements made.

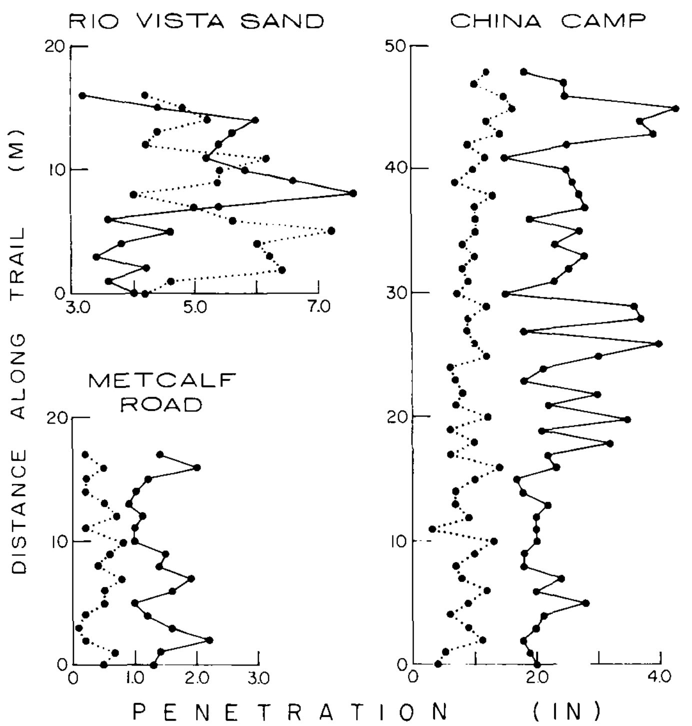

Figure 11. Longitudinal penetrometer profiles consisting of one measurement in (dotted line) and one out of trail at the same elevation.

Liddle and Moore, 1974; Snyder and others, 1976; Webb, 1977; Webb and others, 1977; Wilshire and Nakata, 1976). The effects of human and other animal trampling also yield similar physical responses (see reviews by Speight, 1973; Liddle, 1975).

These marked changes in the physical and chemical properties of the soil have important implications for the biologic productivity of the land, its vulnerability to erosion, and the spread of damage to areas not directly impacted.

Effects of vehicular impacts on biologic productivity

Extensive studies have been made on the effects of strength of soil, bulk density, and soil moisture on growth of crop plants. Considerable emphasis is placed on the importance of impedance by surface strength to root extension (Phillips and Kirkham, 1962; Taylor and Gardner, 1963; Taylor and Burnett, 1964: Barley and others, 1965; Taylor and Bruce, 1968; Lowry and others, 1970; Grimes and others, 1972; Voorhees and others, 1975) and emergence of seedlings (Arndt, 1965; Taylor, 1971). However, increased bulk density and concomitant decreases in soil moisture and air also retard growth of roots (Veihhmeyer and Hendrickson, 1948; Meredith and Patrick, 1961; Phillips and Kirkham, 1962; Taylor and Gardner, 1963; Grable and Siemer, 1968; Taylor and Ratliff, 1969; Lowry and others, 1970; Voorhees and others, 1971).

Grimes and others (1972) state that soil strengths greater than about 18 bars are extremely restrictive to elongation of roots, and Taylor and Burnett (1964) report a limiting strength for root penetration in one soil of about 30 bars. Values of penetration in vehicular use zones (Fig. 10) that are less than about 1.7 in. exceed the strengths found to be restrictive to elongation of roots, and values less than about 1.3 in. exceed the limiting value reported by Taylor and Burnett (1964).

Veihmeyer and Hendrickson (1948) report limiting bulk densities for root penetration of sunflowers in clayey soils of 1.6–1.7 g/cm^3 and 1.75 g/cm^3 for sandy soil. Voohees and others (1971) reported significant restriction to penetration of roots in soil with a bulk density of 1.8 g/cm^3. The average bulk densities of the upper 10 cm of vehicle-impacted soil at Ellicott Station, China Camp, Metcalf Road, and Montara exceed the limiting densities reported by Veihmeyer and Hendrickson (1948). Vehicle-impacted soils from the other three sites show substantial increases of bulk density in comparison to unused soils, but the average values for the upper 10 cm of soil range from 1.3 to 1.5 g/cm^3.

Root extension into soil is also governed by the size and abundance of fractures. Observations on desiccation fractures in clayey soils at Metcalf Road show prominent effects of vehicular use on dimensions of fractures similar to those reported by Wilshire and Nakata (1976). Desiccation polygons in unused soil are as much as 1.3 m in diameter; the fractures are as much as 15 cm wide and more than 55 cm deep. More typical fracture systems have polygon diameters of 15–45 cm, fracture widths of 1–3 cm and depths of from 15 cm to more than 45 cm. In areas used by vehicles desiccation polygons have diameters of 8–15 cm and fracture widths and depths of only 0.5–1 cm. Although the fracture pattern is more closely spaced in the vehicle-impacted soil, it is much shallower and does not provide avenues for extension of roots below a centimeter. This desiccation behavior is probably dependent on vehicle-induced changes in surface strength, moisture content, and bulk density.

Growth of plants and germination of seeds are also temperature sensitive (Shul'gin, 1957; Nielsen and Humphries, 1966; Luckenbach, 1975; Geological Society of America, 1977). Reestablishment of plants in vehicle-impacted areas will therefore be affected by the increased diurnal temperature range of those soils. The effects of changing the temperature of the soil include altering the time of germination (Shul'gin, 1957; Luckenbach, 1975; Geological Society of America, 1977), changing the concentration of soluble nutrients and salts in the soil, and changing the rates of chemical reactions (Nielsen and Humphries, 1966). All of these factors are also influenced by moisture content and porosity, so that the effect of changes in temperature cannot be isolated for the soils investigated here.

The effects of these changes on animal populations are equally profound. In addition to the direct impacts of vehicles and the adverse effects of loss of vegetation (Hicks and others, unpublished data 1976; Bureau of Land Management, 1975; Bury and others, 1977; Busack and Bury, 1974; Duck, 1977; Geological Society of America, 1977), the changes of soil properties described all directly affect the small animal populations. In turn, the loss of burrowing animals will extend the time required for revegetation. An excellent review of the ecologic impacts of

Table 5 Bulk Density and Soil Moisture

Locality	Depth (cm)	Bulk density (g/cm³)		Soil moisture (% dry weight)	
		In trail	Out of trail	In trail	Out of trail
Ellicott Stn.	0–10	1.87	1.37	3.7	8.3
	10–20	1.86	1.80	5.1	7.3
	20–30	1.71	2.11	5.3	6.0
	0–10	1.71	1.13	4.8	7.8
	10–20	1.74	1.85	6.2	5.2
	20–30	1.55	1.86	2.7	5.1
	0–10	1.72	1.18	4.4	11.7
	10–20	2.03	1.41	4.7	10.7
	20–30	1.74	1.94	6.2	9.9
	0–10	2.01	1.13	4.1	11.7
	10–20	1.89	1.71	6.3	7.4
	20–30	1.82	1.97	7.6	6.4
	0–10	1.93	1.26	2.3	10.0
	10–20	1.87	1.90	6.7	8.5
	20–30	1.85	1.98	6.8	9.7
	0–10	1.71	1.45	5.9	8.7
	10–20	1.73	1.70	6.6	8.8
	20–30	1.98	1.56	5.7	9.2
	0–10	1.37	1.12	6.4	9.0
	10–20	1.66	1.94	6.8	9.9
	20–30	1.78	2.15	6.7	9.8
	0–10	1.65	0.74	6.5	7.6
	10–20	1.89	1.24	6.3	10.2
	20–30	1.85	1.94	7.1	9.9
Montara	0–10	1.62	1.30	5.8	4.8
	10–20	1.98	1.98	9.1	6.7
	20–30	1.68	1.74	12.5	7.9
	0–10	1.76	1.47	7.5	4.3
	10–20	2.12	2.01	9.9	6.2
	20–30	1.84	2.04	10.1	6.1
	0–10	1.91	1.10	3.3	9.1
	10–20	2.06	1.88	8.4	7.3
	20–30	1.92	1.84	7.8	8.8
China Camp	0–10	1.47	1.10	10.7	14.8
	10–20	1.84	1.59	9.7	12.0
	20–30	1.70	1.81	8.5	10.9
	0–10	1.74	1.19	7.2	18.1
	10–20	1.94	1.93	9.3	16.8
	20–30	1.79	1.88	11.4	18.5
	0–10	1.72	1.21	4.8	15.1
	10–20	1.82	1.17	6.9	15.4
	20–30	1.32	1.28	3.6	16.0
	0–10	1.55	1.19	5.3	17.0
	10–20	1.87	1.50	5.8	13.6
	20–30	1.75	1.87	3.8	14.8
	0–10	1.55	1.30	2.4	14.5
	10–20	1.71	2.01	4.9	12.8
	20–30	1.78	1.91	5.0	12.3

Locality	Depth (cm)	Bulk density (g/cm³		Soil moisture (% dry weight)	
		In trail	Out of trail	In trail	Out of trail
Chabot Park	0–10	1.56	0.78	6.5	22.6
	10–20	1.21	1.27	7.5	20.5
	20–30	1.17	1.30	8.8	17.2
	0–10	1.59	0.87	6.0	19.6
	10–20	1.67	1.73	5.0	14.3
	20–30	1.55	1.35	7.0	12.9
	0–10	1.15	0.77	8.8	30.3
	10–20	1.11	1.19	12.4	19.2
	20–30	1.23	1.24	9.8	16.5
	0–10	1.44	0.54	5.3	23.3
	10–20	1.63	1.20	5.9	10.1
	20–30	1.31	1.61	6.5	11.6
	0–10	1.58	0.73	7.6	28.4
	10–20	1.55	1.08	7.0	18.3
	20–30	1.34	1.01	8.1	18.5
	0–10	1.67	1.62	11.0	15.2
	10–20	1.63	1.57	8.1	13.8
	20–30	1.69	0.79	7.9	15.0
Metcalf Rd.	0–10	1.82	1.55	8.7	14.7
	10–20	2.28	1.73	6.3	16.0
	20–30	No Sample	2.01	No Sample	12.5
	0–10	1.69	1.41	7.5	10.8
	10–20	1.57	1.75	9.6	11.0
	20–30	1.61	1.86	11.7	12.4
Rio Vista Sand	0–10	1.51	1.32	7.5	5.7
	10–20	1.82	1.71	7.6	5.9
	20–30	1.57	1.80	9.9	8.3
	0–10	1.21	1.20	5.7	2.6
	10–20	1.71	1.63	7.3	4.7
	20–30	1.84	1.76	6.8	7.1
	0–10	0.98	1.19	4.4	2.1
	10–20	1.55	1.43	5.8	3.8
	20–30	1.90	1.82	6.3	5.2
	0–10	1.34	1.26	6.1	4.0
	10–20	1.77	1.63	6.8	5.0
	20–30	1.80	1.68	5.3	5.7
	0–10	1.33	1.10	5.2	2.8
	10–20	1.66	1.75	6.1	3.4
	20–3-	1.76	1.85	5.4	4.5

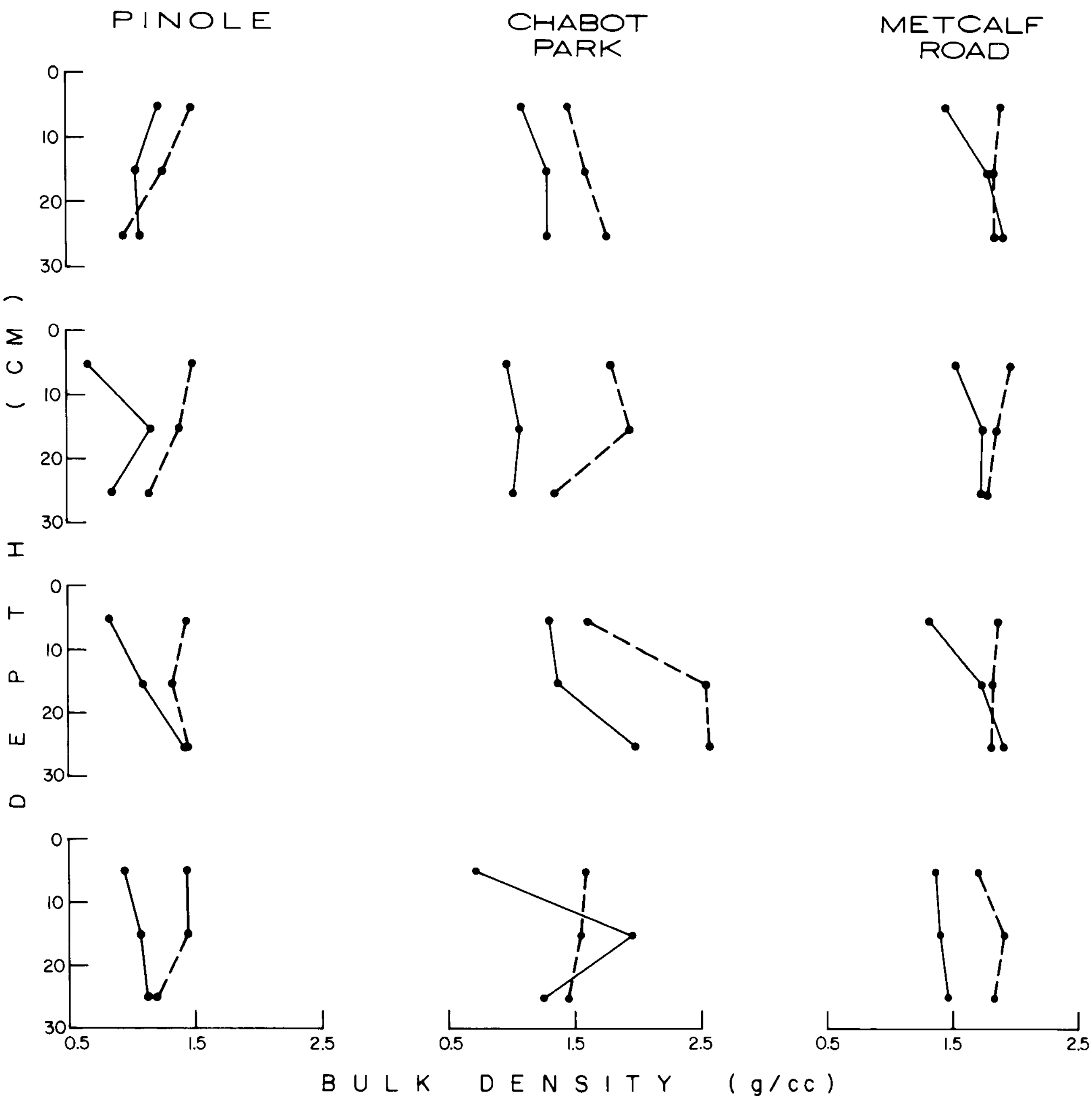

Figure 12. Examples of the variation of soil bulk density in (dashed line) and out (solid line) of trails to depths of 30 cm. Each set of paired samples was taken at a different elevation along a single trail; samples taken from bottom to top of the trail are represented from bottom to top of each column.

human trampling (Liddle, 1975) describes effects on small animal soil inhabitants that are likely to be similar to the effects of vehicular use.

The lower content of organic carbon in soils exposed in the vehicle-impacted areas compared to unused areas not only reflects a reduced infiltration and water-holding capacity; in the long term it slows the rate of weathering and soil production by reducing the amount of carbonic acid formed in the upper soil zone (Buckman and Brady, 1969). It is likely that the soil exposed in the vehicle-impacted areas has lower nutrient contents as reported by Webb and others (1977), which further inhibits plant growth.

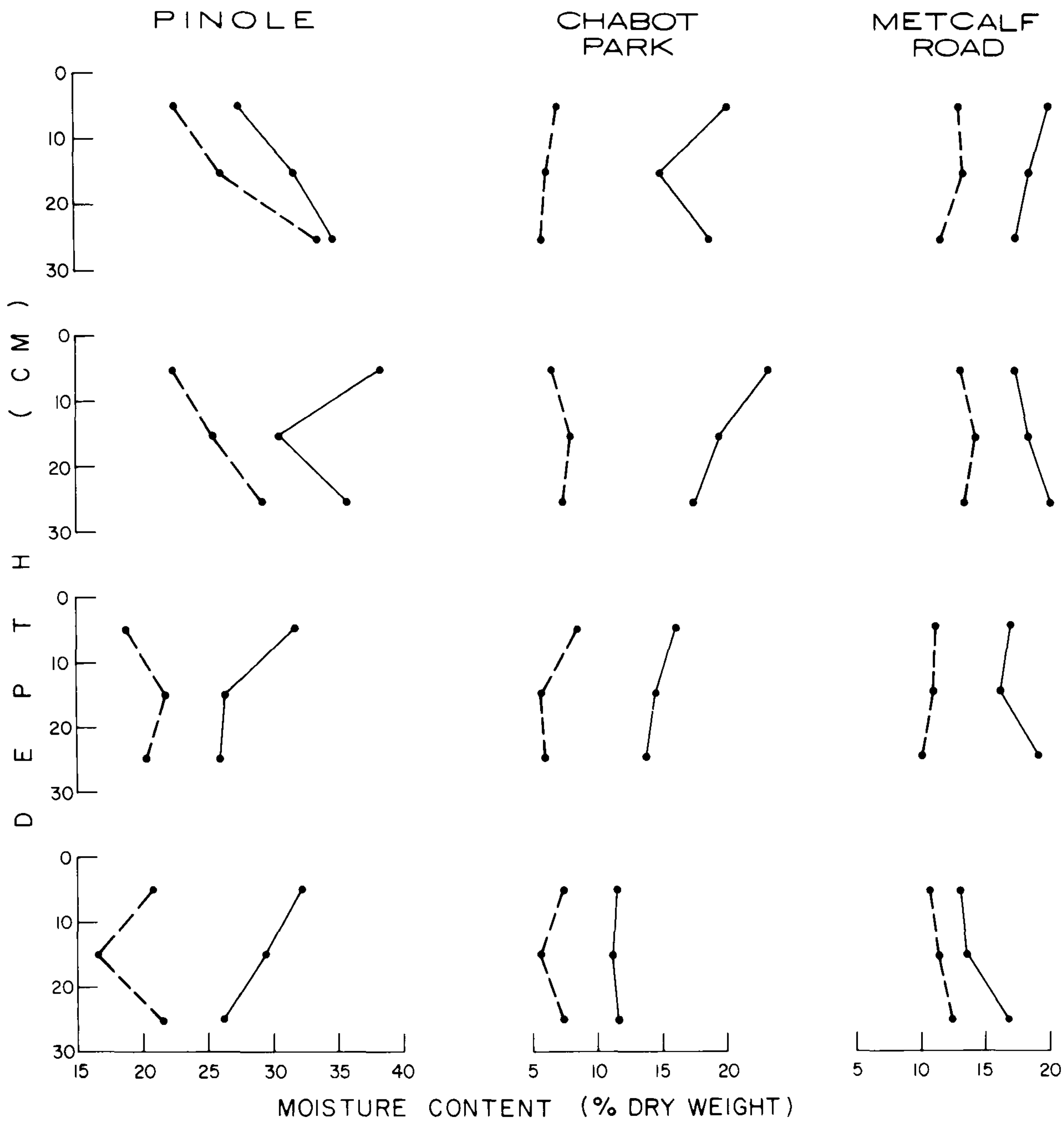

Figure 13. Examples of the variation of soil moisture in (dashed line) and out (solid line) of trails to depths of 30 cm. Each set of paired samples is taken at a different elevation along a single trail; samples taken from bottom to top of the trail are represented from bottom to top of each column.

Effects of vehicular impacts on erosion

The changes of soil properties described lead to reduced infiltration and increased runoff of rain water, which lead in turn to accelerated erosion (Snyder and others, 1976; Webb, 1977; Wilshire and Nakata, 1977a, b; Wilshire, 1977; Nakata and others, 1976; Webb and others, 1977). Moreover, the changes clearly inhibit the restoration of vegetation and activity of animals so that the increased erosion potential is perpetuated. To this must be added the very considerable effect of dry land recreational vehicles in directly eroding surface materials (Wilshire and Nakata, 1977b).

Erosional losses from individual trails were measured (see

Wilshire and Nakata, 1977a) and the results are reported in Table 8. Three to 30 profiles of individual trails were constructed by selecting sites where uniform original slopes could be reasonably inferred. A string was drawn taut between spikes set in the original ground surface on each side of a trail, and the distance between the string and the present ground surface was measured at 10 cm intervals. The area of the resultant profile (Fig. 16) was multiplied by the proportioned distance between profiles to obtain a total volume loss; mass loss was determined using the appropriate soil bulk density. Where erosion had penetrated bedrock (e.g., Fig. 16), the volumes of soil and bedrock losses were determined independently because the bulk densities differed substantially.

The losses of soil and substrate measured range from 17 to 370 kg/m^2 for sandy soils and from 7 to 1180 kg/m^2 for more silty and clayey soils. The loss per unit area varies widely, depending on such factors as duration of use, slope length, and trail gradient.

The rate of soil loss can only be estimated because the dates of opening of individual trails are not accurately known. However, minimum values are available by using the age of the facility in most cases, and others are conservatively estimated from the known minimum age of the site (Table 8). Where only a minimum age of the site is known, erosion rates are based on a duration of erosion twice or more the known minimum age.

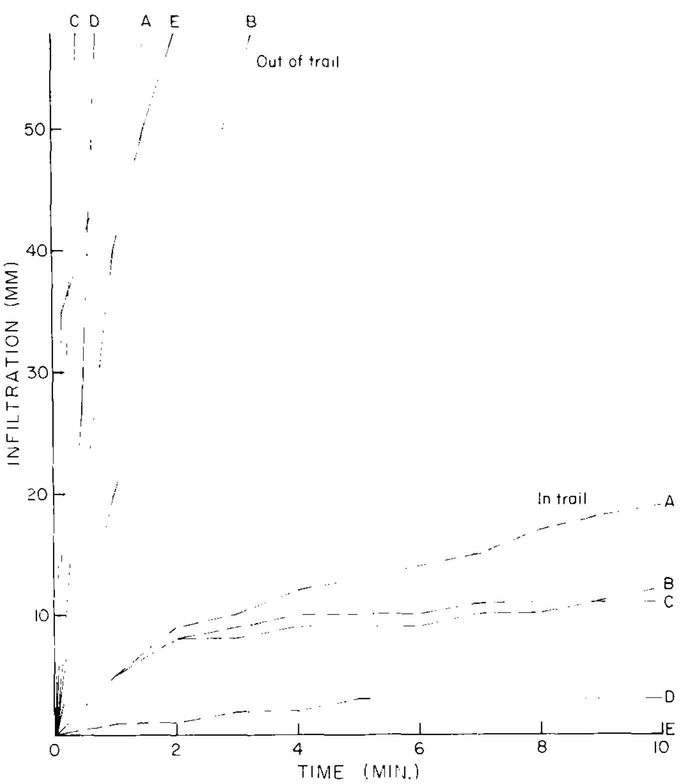

Figure 14. Rate of infiltration of water in five paired samples in and out of trails, Rio Vista.

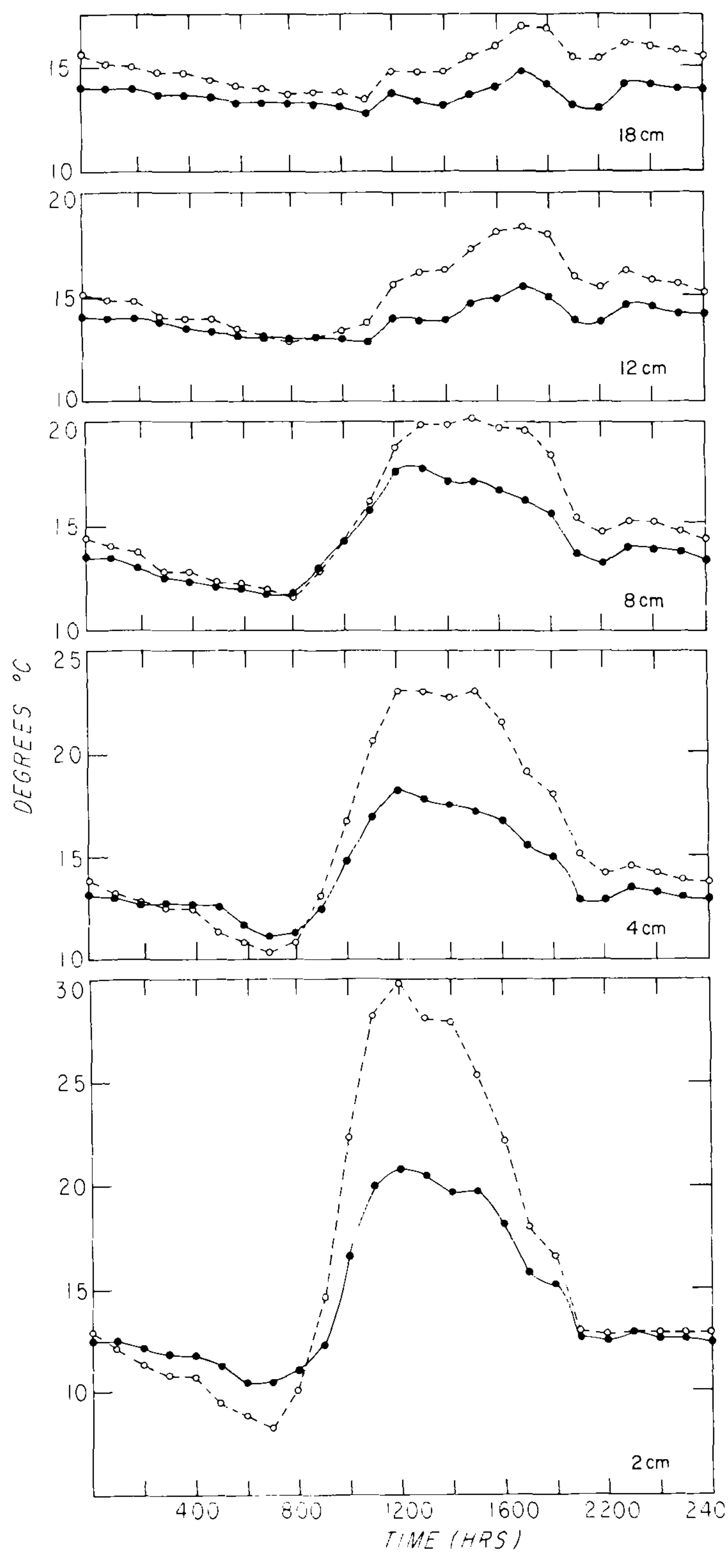

Figure 15. Diurnal temperature curves for five different depths in (dashed line) and out (solid line) of a trail, Rio Vista.

Table 6 Diurnal Temperature Fluctuation

Location: Ellicott Stn. Date: 3/30/71

	Depth and temperature (°C)									
	2 cm		4 cm		8 cm		12 cm		18 cm	
Time	In	Out	In	Out	In	Out	In	Out	In	Out
0	5.1	6.9	6.2	6.1	10.1	10.5	10.7	11.5	12.0	12.5
0100	4.9	6.4	6.0	5.9	9.7	10.2	10.1	11.1	11.8	12.5
0200	3.9	6.0	5.4	5.4	9.0	10.1	9.9	11.1	11.2	12.1
0300	3.8	5.9	5.0	5.1	8.2	9.6	9.1	10.5	11.0	11.8
0400	3.0	5.1	4.5	4.9	8.0	9.0	8.6	10.1	10.8	11.6
0500	2.2	4.5	3.8	4.2	7.5	8.9	8.0	10.0	10.1	11.1
0600	2.2	4.9	3.9	4.1	7.1	8.5	7.5	9.7	9.9	11.0
0700	2.9	5.1	3.9	4.7	6.8	8.1	7.2	9.1	9.5	10.8
0800	4.0	6.0	4.5	5.5	5.5	8.0	6.9	8.9	9.0	10.2
0900	14.2	13.2	13.8	12.5	10.2	10.0	10.1	10.1	11.2	11.8
1000	19.2	17.4	17.3	16.2	11.3	10.1	10.6	10.1	10.1	10.1
1100	18.9	17.1	17.4	16.2	12.1	10.1	10.2	9.0	9.0	9.0
1200	27.1	23.7	25.0	21.1	18.2	15.0	16.8	14.0	14.0	13.0
1300	28.9	23.9	27.2	20.2	20.2	15.8	18.7	14.5	15.0	12.9
1400	30.2	24.1	29.3	21.8	23.1	18.1	21.9	17.0	17.6	14.9
1500	19.5	16.3	22.0	15.9	21.8	17.8	21.9	17.4	18.9	15.5
1600	16.3	15.0	18.2	14.5	19.3	16.3	19.9	16.6	18.4	15.5
1700	14.5	13.1	16.1	12.5	17.2	15.2	18.1	15.7	17.2	15.0
1800	13.5	12.9	15.2	12.2	17.0	15.1	17.5	15.6	17.1	15.1
1900	11.1	11.1	13.0	10.9	15.9	14.9	16.5	15.2	16.5	15.2
2000	9.0	9.5	10.9	9.0	14.0	13.2	15.0	14.3	16.2	14.5
2100	7.9	8.9	9.5	8.1	13.0	12.6	13.9	13.5	14.7	14.0
2200	6.9	8.1	8.5	7.8	11.9	11.9	12.5	12.8	13.9	13.8
2300	6.0	7.2	7.5	6.9	11.0	11.2	11.9	12.2	13.1	13.1

Location: China Camp Date: 4/1/77

	Depth and temperature (°C)									
	2 cm		4 cm		8 cm		12 cm		18 cm	
Time	In	Out	In	Out	In	Out	In	Out	In	Out
0	8.2	9.1	9.0	11.3	9.0	11.5	11.3	11.5	13.8	11.4
0100	8.0	8.9	8.8	9.0	11.0	11.2	11.0	11.5	13.1	11.2
0200	7.0	7.9	7.8	8.0	10.3	11.0	10.5	11.1	12.8	11.0
0300	6.6	7.9	7.1	7.6	10.1	10.9	10.1	11.0	12.2	10.9
0400	5.9	7.0	6.9	7.0	9.9	11.0	9.9	11.0	12.1	11.0
0500	6.0	6.9	6.5	6.9	9.2	10.5	9.5	10.9	11.9	10.9
0600	6.0	7.2	6.2	7.0	8.8	10.2	9.0	10.5	11.4	10.8
0700	10.2	8.1	9.1	8.5	8.9	10.3	9.2	10.8	10.2	10.9
0800	14.0	9.6	12.5	9.5	10.0	10.8	10.0	10.7	11.0	10.8
0900	17.9	15.2	17.1	15.4	12.8	11.1	11.7	11.5	11.5	11.0
1000	20.0	16.7	19.1	16.8	14.1	11.2	12.2	11.8	11.8	10.9
1100	22.5	21.8	21.9	19.3	16.2	11.8	13.5	12.0	11.9	10.4
1200	24.9	21.4	24.8	20.9	19.1	12.9	15.3	13.0	13.1	10.9
1300	24.2	19.9	24.9	20.0	20.9	14.1	17.0	14.5	14.9	11.8
1400	24.7	18.5	25.0	19.9	21.2	14.0	17.6	14.3	15.1	11.5
1500	22.1	16.7	23.1	17.5	21.5	14.1	17.8	14.2	15.9	11.5
1600	18.8	14.9	20.2	15.0	20.5	14.0	17.7	13.9	16.3	11.5
1700	14.9	12.2	16.6	12.2	18.9	13.2	16.8	13.2	16.4	11.4
1800	12.8	11.3	14.9	11.5	17.3	12.9	16.0	12.9	16.2	11.3
1900	11.0	10.8	12.7	10.8	15.9	12.8	15.0	13.0	16.3	11.9
2000	10.1	10.8	11.5	10.4	14.8	12.2	14.0	12.9	15.9	11.9
2100	9.5	9.9	10.6	10.0	13.5	12.0	13.0	12.4	15.2	11.8
2200	9.2	10.0	10.0	9.8	12.8	11.7	12.5	12.2	14.6	11.8
2300	8.9	9.9	9.8	9.2	12.0	11.8	12.0	12.0	14.1	11.5

Table 6 Diurnal Temperature Fluctuation (*Continued*)

Location: Pinole Date: 3/10/77

	Depth and temperature (°C)									
	2 cm		4 cm		8 cm		12 cm		18 cm	
Time	In	Out	In	Out	In	Out	In	Out	In	Out
0	0	0	0.1	0.9	4.0	3.0	5.1	5.0	7.0	5.0
0100	0	1.0	1.0	1.5	4.0	3.0	6.5	5.0	7.8	6.0
0200	0	0.8	0	1.0	3.1	2.3	5.5	4.9	7.0	5.8
0300	0	0	0.7	1.0	3.0	2.3	5.0	4.7	6.3	5.3
0400	0	0	0	0.5	2.0	2.0	4.0	4.2	5.3	5.3
0500	0	0	0	0.1	1.9	2.0	4.1	4.5	5.0	5.0
0600	0	0	0	0	2.0	1.5	3.2	4.0	5.1	5.0
0700	0	0	0	0.2	1.5	2.0	3.2	4.5	4.9	5.8
0800	2.2	0.9	1.2	1.0	1.8	2.0	3.1	4.0	5.1	5.3
0900	10.0	3.0	5.2	2.8	3.5	3.0	3.4	4.5	5.0	5.6
1000	12.2	5.3	5.2	4.8	3.5	3.6	3.0	3.8	4.2	4.8
1100	18.0	9.0	13.1	8.0	7.8	5.1	4.2	3.8	4.3	5.2
1200	22.5	10.2	17.0	9.8	10.9	6.5	6.0	4.2	5.0	4.8
1300	24.2	12.0	19.2	12.0	13.5	8.8	8.5	5.4	6.3	5.2
1400	23.2	14.0	20.5	13.8	15.8	10.0	11.0	7.0	8.0	6.0
1500	24.9	14.0	21.1	13.5	16.0	10.1	11.1	6.5	8.0	5.5
1600	20.0	12.0	18.2	11.8	15.5	9.0	11.2	6.0	8.0	4.5
1700	16.0	10.5	16.1	11.0	16.0	9.5	13.0	7.1	10.0	6.0
1800	11.8	7.0	10.1	7.3	13.0	7.6	12.3	6.9	10.2	6.1
1900	4.8	4.0	6.0	4.9	6.9	5.9	7.9	6.8	10.0	6.8
2000	3.5	2.2	5.7	3.0	8.8	4.7	10.0	5.8	9.9	5.9
2100	1.3	2.8	3.7	2.2	7.0	4.2	8.9	4.9	5.6	4.9
2200	1.4	1.7	3.5	2.2	6.5	4.0	8.7	6.2	9.2	6.3
2300	0.1	0.8	2.1	1.8	5.2	3.9	7.8	6.0	8.1	6.0

Location: Metcalf Rd. Date: 2/6/77

	Depth and temperature(°C)									
	2 cm		4 cm		8 cm		12 cm		18 cm	
Time	In	Out	In	Out	In	Out	In	Out	In	Out
0	6.8	7.2	7.9	8.0	9.4	9.8	11.0	10.0	10.7	10.1
0100	7.0	7.1	8.1	7.8	8.9	8.9	10.5	9.6	10.7	9.8
0200	6.1	6.9	7.0	7.9	8.5	9.2	10.1	9.9	10.0	10.0
0300	5.9	6.5	7.2	7.1	8.1	8.5	9.8	9.0	10.0	9.1
0400	5.8	6.2	6.5	7.1	7.8	8.8	9.9	9.0	9.8	9.1
0500	5.9	6.2	6.9	7.0	7.3	8.1	9.0	8.5	9.5	9.0
0600	5.1	6.0	6.0	7.0	7.1	8.8	9.0	9.0	8.9	9.5
0700	5.2	5.9	6.2	6.7	7.0	7.9	8.2	8.0	8.9	8.9
0800	6.5	6.9	6.9	7.4	7.0	8.0	8.5	8.3	8.2	9.1
0900	9.0	8.9	8.9	8.2	7.8	8.1	8.1	8.2	8.5	8.8
1000	14.6	12.1	13.5	10.1	9.9	8.9	9.0	8.9	9.1	9.9
1100	17.1	15.7	17.3	13.0	12.5	9.8	10.1	9.2	9.9	9.8
1200	18.9	16.4	18.5	14.0	13.9	9.9	10.9	9.7	10.0	9.9
1300	20.4	17.9	20.8	15.2	16.0	10.1	11.8	9.9	10.1	9.5
1400	19.8	16.2	19.9	14.0	16.5	10.6	12.5	10.0	11.0	9.0
1500	19.7	16.1	20.3	14.5	17.8	11.3	14.2	11.0	12.0	9.9
1600	17.1	15.0	17.5	14.8	17.1	12.1	15.1	12.1	13.2	10.6
1700	14.1	12.0	15.0	11.9	16.0	11.5	15.0	11.8	13.0	10.1
1800	11.0	10.1	12.0	10.2	14.0	11.0	14.4	11.3	13.0	10.0
1900	10.0	9.5	11.1	9.6	13.0	10.8	13.9	11.2	12.9	10.1
2000	8.8	8.6	10.0	9.0	11.9	10.6	13.0	11.0	12.0	10.0
2100	7.9	7.9	9.5	8.5	11.0	10.1	12.4	10.8	12.0	10.0
2200	6.9	7.2	8.2	8.1	10.3	10.0	11.8	10.2	11.1	9.9
2300	6.8	7.0	8.5	8.0	9.8	9.5	11.2	10.0	11.3	9.9

Table 7 Organic and Inorganic Carbon and pH

Locality and soil type	Organic C (wt. %) In	Organic C (wt. %) Out	Inorganic C (wt. %) In	Inorganic C (wt. %) Out	pH In	pH Out
Ellicott Stn.	(0.35)*	(1.39)			6.5	6.2
	0.50	1.09	0.02	0.02		
	0.52	0.66	0.00	0.00	6.7	6.2
Montara	1.58	5.10	0.03	0.00		
	2.47	1.94	0.02	0.02	6.6	6.7
	0.78	2.74	0.03	0.00	6.8	6.5
China Camp	2.98	1.23	0.02	0.02		
	1.15	1.17	0.04	0.02		
	1.91	2.51	0.01	0.01	5.3	5.5
	1.74	3.50	0.03	0.01	5.4	6.0
	1.07	3.86	0.02	0.01		
Pinole	1.15	2.46	0.01	0.01	7.4	7.4
	1.05	2.15	0.02	0.00		
	1.24	1.12	0.02	0.01		
Rio Vista	0.27	0.24	0.03	0.00	6.7	6.4
Sand	0.08	0.16	0.00	0.00		
	0.15	0.55	0.00	0.00	6.9	6.9
Clay	(0.81)	(3.99)				
Chabot Park	0.76	3.96	0.02	0.00		
	0.86	1.46	0.00	0.00	6.3	6.1
	0.78	1.20	0.01	0.00	5.8	6.1
	0.89	4.54	0.01			
Metcalf Rd.	1.06	1.01	0.01	0.00	7.1	7.4
	1.39	1.95	0.00	0.00		
	1.01	1.25	0.00	0.00	6.9	6.7
	1.16	1.51	0.01	0.00		

*Parantheses indicate total organic and inorganic C.

Erosion rates from entire areas of use cannot be estimated except for the Chabot Park site because of the lack of good recent aerial photographs. Point counts of good quality aerial photographs of Chabot Park taken in 1965 indicate that 38% of the area was then exposed to erosion. With the minimum measured erosional losses from a single trail (Table 8), it is estimated that the area as a whole is eroding at a rate substantially greater than 11,500 tons/km^2/year.

The significance of this rate can be appreciated by comparison with the rate representing the onset of a serious problem of erosion (380 tons/km^2/year; United States Dept. of the Interior, 1975), the rate suggested by the United States Soil Conservation Service for soil-loss tolerance for soils of the types in question (250 tons/km^2/year; United States Dept. of Agriculture, 1975), and the average erosion rate in the San Francisco Bay area, itself an accelerated rate (690 tons/km^2/year; Brown and Jackson, 1973, 1974). The estimated rate of erosion at Chabot Park exceeds these values by factors of, respectively, 30, 46, and 17. Similar extremely accelerated erosion rates were determined for the State Vehicular Recreation Area at Hollister Hills, where the estimated average erosion rate for the entire area is 6400 tons km^2/year, which exceeds the comparison standards by factors of respectively, 17, 26, and 9. This is further supported by total area erosion rates measured by Panoche Hills (Snyder and others 1976), where the erosion rate in the vehicular use area exceeded 1580 tons/km^2/year cmpared to an amount too small to measure in the adjacent unused control area over a period of 2 years. Therefore, total area erosion rates at the sites studied are likely to far exceed all standards with which they are compared above.

The erosional losses that are likely to be incurred from off-road vehicular use of the land can be analyzed by application of

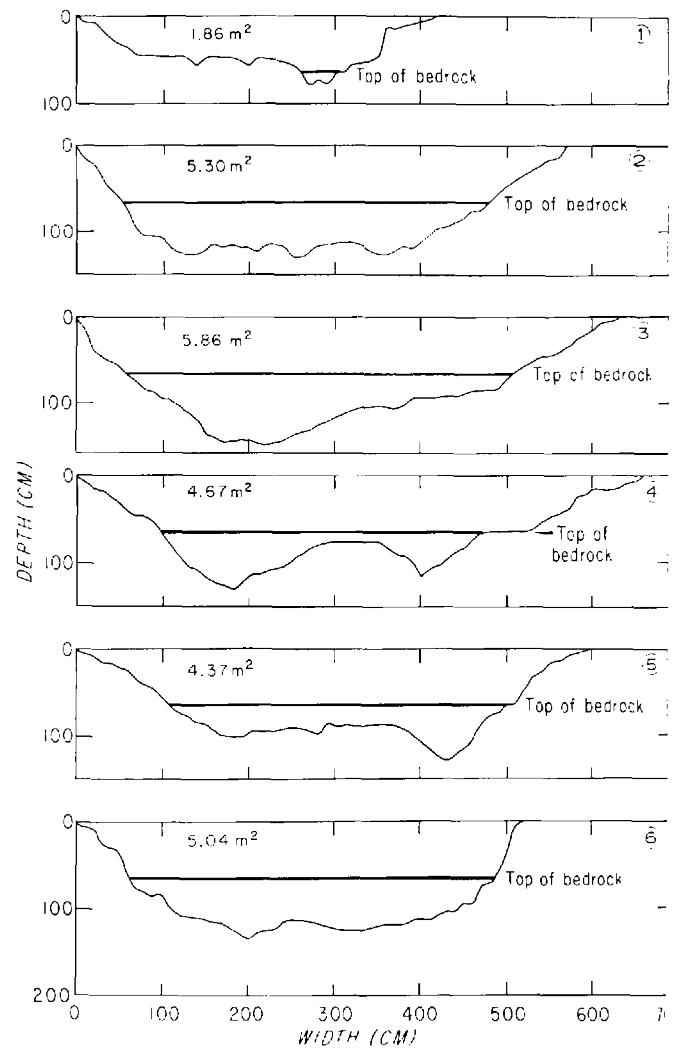

Figure 16. Soil loss profiles measured along a single trail at Chabot Park. Profile 6 in near the base, and profile 1 near the top of the trail. Total mass loss was computed at 623 metric tons, or 1180 kg/m^2.

Table 8 Erosional Loss from Individual Trails and Estimated Erosion Rates

Locailty	Date	Soil type	Trail length (m)	Trail slope (%)	Total mass loss (metric tons)	Mass loss per m² (kg/m²)	Estimated erosion rate (tons/km²/ year)*
Ellicott Stn.	11/7/75	Sand	79	30–40	278	51	8,500
	11/7/75	Sand	74	30–40	104	17	2,700
Montara	12/18/75	Gravelly sand loam	83	56	201	210	26,000
China	3/31/76	Gravelly sandy loam	58	33–40	35	120	
Camp	3/31/76	Gravelly sand loam	55+	51	59	210	20,800
	4/ 1/77	Gravelly sandy loam	195	20–51	228	180	
	4/ 1/77	Gravelly sandy loam	103	11–40	117	110	10,800
	4/ 1/77	Gravelly sandy loam	63	13–40	99	110	
Pinole	11/19/76	Silt loam	38	34	31	20	1,500
	12/26/76	Silt loam	60	55	14	7	580
Rio Vista	1/16/77	Silt loam and sand	68	78	95	540	54,000
	1/16/77	Silty loam and sand	46	78	170	370	
	12/29/76	Silty loam and sand	57	56	107	230	22,800
Chabot	4/ 9/76	Loamy sand	86+	22–60	295	860	
Park	3/ 5/77	Loamy sand	105	38–71	423	960	
	3/ 5/77	Loamy sand	97	29–56	219	629	
	3/ 5/77	Loamy sand	92	27–60	623	1,180	61,800
Metcalf	8/20/77	Sandy loam	74	11–33	19	50	25,000

*Figures used for duration of erosion: Ellicott St., 6 years; Montara, 8 years; China Camp, 10 years; Pinole, 12 years; Rio Vista, 10 years; Chabot Park, 19 years; and Metcalf Rd., 2 years.

the universal soil loss equation. The equation[1] was developed empirically to describe erosion rates of agricultural lands by sheet wash and rill erosion (Wischmeier, 1976). Although the parameters of this equation should be taken into account in planning for vehicular use of the land (Webb and others, 1977, have given an excellent discussion of this subject), the equation is likely to provide only minimum erosion rates without special modification, such as the following; In vehicular use areas gullying is commonplace and occurs even with trail maintenance; the effects of vehicular use on the soil erodibility must be evaluated; and the vehicles themselves cause substantial erosion without the intervention of running water. A dual program of evaluating these factors and direct measurement of erosion rates from vehicular use areas will allow more accurate application of the universal soil loss equation.

In the meantime, it is sufficient to point out that vehicular use for periods of less than 10 years at the sites examined has commonly resulted in total loss of the soil and nearly 20 years of use at Chabot Park has resulted in extensive exposure of bedrock in an area once covered by a 65 cm thick soil mantle. This reflects the importance of the slope length and gradient factors (L and S) of the soil loss equation, both of which are maximized to satisfy vehicular capability. These are the only factors of the equation that can be easily manipulated by management practice, which should include careful control of the steepness and length of uninterrupted trail segments. In almost all terrain–soil situations examined, reasonable soil conservation practice requires trail layouts that are far short of exploiting the machines' capability to negotiate difficult terrain. If this problem is not addressed, vehicular sites will continue to be the areas of uncontrolled erosion that most of them are today.

[1]The universal soil loss equation, $A = R * K * L * S * C * P$, is applied to sheet and rill erosion of agricultural lands. A is the rate of erosion, R is the rainfall factor. K is the erodibility of the soil, L is the slope length, S is the slope gradient, and C and P are cropping and management factors.

Off-site effects

It is not possible to modify the land as extensively as is typical of vehicular use areas without modifying surrounding land as well. The increased runoff caused by vegetation stripping and the physical changes in the soil contribute to increased sediment yield from accelerated erosion which must be accommodated by the drainage system. Except in very unusual circumstances, these new loads on the system are borne by properties downstream from the vehicular areas. Although no quantitative data are available, it is evident from field inspection that parts of stream courses heading in extensively bared areas of Chabot Park are undergoing erosion and others are heavily silted, in contrast to nearby stream courses that are not draining the vehicular use area. These effects, as well as adjustments of the hydrologic system to corrective actions, such as catchment dams, are appropriate management considerations and are likely to prove major cost items.

ACKNOWLEDGMENTS

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motorized or off-road vehicles. No person may use, drive, transport, park, let stand, or have charge or control over any motorized vehicle in the area located east of the closure signs and locked gate. Exemptions to this order may be granted to law enforcement and other emergency vehicles in the course of official duties and for other approved administrative activities performed by the Bureau of Land Management or U.S. Forest Service. Exemptions may also be granted to those persons involved with ranching activities associated with the Whitewolf Grazing Allotment.

EFFECTIVE DATE: This closure became effective Friday, March 7, 2003. The closure will remain in effect unless rescinded by the authorizing official. The permanent decision regarding motorized or off-road vehicle use in Furnace Creek will be determined through an amendment of the California Desert Conservation Area Plan, which is expected to occur by June 30, 2004. BLM will implement the purposed action soon after the effective closure date of March 7, 2003 without prior notice and opportunity for public comment because of the imminent need for regulatory authority to prevent damage to wetland and riparian resources.

FOR FURTHER INFORMATION CONTACT: Field Office Manager, Bureau of Land Management, Ridgecrest Field Office, 300 South Richmond Road, Ridgecrest CA 93555, (760) 384–5400.

SUPPLEMENTARY INFORMATION: The BLM has established national standards for the management and protection of riparian and wetland habitat on the Public Lands. Monitoring conducted during November 2002 and February 2003 indicates that the Furnace Creek fluvial system is not meeting the BLM's standards for a properly functioning riparian system. presently, portions of the Furnace Creek drainage are considered ''functional-at risk''. Riparian-wetland areas are considered ''functional-at risk'' when an existing soil, water, or vegetation condition makes them susceptible to degradation. Presently, there are seven locations in Furnace Creek where the existing vehicle route crosses the stream. Significant erosion and sedimentation of the stream are occurring at two stream crossings. Erosion in both locations is contributing excessive sediment to the adjacent riparian area. moreover, head-cutting is forming at both locations. Head-cuts are a fluvial geomorphic feature indicative of unstable conditions. The proposed closure order is consistent with protecting and restoring Furnace Creek to a properly functioning riparian system.

Bureau of Land Management's regulatory policy concerning the use of off-road vehicles on public lands is found in 43 CFR 8341. Whenever the authorized officer determines that OHV use will cause or is causing considerable adverse effects on resources (soil, vegetation, wildlife habitat, cultural, historic, scenic, recreation, or other resources), the area must be immediately closed to the type of use causing the adverse effects. The closure must remain in force only until the adverse effects are eliminated and measures to prevent their recurrency have been implemented (whichever occurs first). A considerable adverse environmental effect resulting from the use of off-road vehicles is defined in 43 CFR part 8341 as any environmental impact that causes:

(a) Significant damage to cultural or natural resources, including but not limited to historic, archaeological, soil, water, air, vegetation, scenic values; or

(b) Significant harassment of wildlife and/or significant disruption of wildlife habitats; . . . and is irreparable due to the impossibility or impracticality of performing corrective or remedial action.

Furnace Creek canyon will remain open for human use that does not entail the use of a motorized vehicle within the area closed by this order. Maps depicting the affected area are available by contacting the Ridgecrest Field Office, California Desert Conservation Area, Ridgecrest, CA. A gate will be erected at the closure points and the affected area will be posted with public notices and standard motorized vehicle closure signs.

Authority for this closure is found in 43 CFR 8364.1. Violations of this order may be subject to the penalties provided according to 43 CFR 8360.0–7.

Dated: March 11, 2003.

Hector A. Villalobos,

Ridgecrest Field Manager.

[FR Doc. 03–12522 Filed 5–19–03; 8:45 am]

BILLING CODE 4310–AG–M

DEPARTMENT OF THE INTERIOR

Bureau of Land Management

[ID–076–1220–BA]

Notice of Closure to Off-highway Vehicle Use

AGENCY: Bureau of Land Management, Interior.

ACTION: Notice of closure to off-highway vehicle use.

SUMMARY: With the publication of this notice, all existing trails and cross-country travel on certain lands administered by the Bureau of Land Management (BLM) Shoshone Field Office are closed to off-highway vehicle (OHV) use. The closure will remain in effect until the proposed Resource Management Plan (FY2005) can implement OHV designations, or until such time as the authorized officer of the Shoshone Field Office determines the closure may be lifted. The closure is in accordance with 43 CFR 9268.3(d)(1)(i–iii) and 43 CFR 8341.1(f)(4).

FOR FURTHER INFORMATION CONTACT: John Kurtz, Outdoor Recreation Planner, (208) 732–7296, BLM Shoshone Field Office, 400 West F Street, Shoshone, ID 83352.

SUPPLEMENTARY INFORMATION: The Blaine County Muldoon Summit Road dips where it passes through Bureau of Land Management administered land, resulting in limited visibility for 1/4 of a mile causing public safety issues. Within this section, All Terrain Vehicles (ATV), motorcycles and snowmobiles cross the county road to gain momentum to hill climb. These vehicles are climbing steep slopes resulting in ruts, vegetation damage, noxious weed spread and erosion. The vertical trails also cause visual scars for the surrounding residents and communities. These lands are also important winter wildlife habitat areas. This closure will protect these resources and reduce the potential for further noxious weed invasion. Private landowners adjacent to the lands have complained about the resource damage. This closure is in response to those complaints.

The area of closure includes BLM lands, specifically described wholly or partially:

Boise Meridian

T. 2 N., R. 19 E., Sec.31, N1/2NW1/4 (80 Acres).

Detailed maps of the area closed to OHV and recreational use are available at the BLM Shoshone Field Office, 400 West F Street, Shoshone, ID 83352.

Dated: April 3, 2003.

Rick VanderVoet,

Acting Shoshone Field Manager.

[FR Doc. 03–12516 Filed 5–19–03; 8:45 am]

BILLING CODE 4310–GG–P

Acute effects of suspended sediment angularity on juvenile coho salmon (*Oncorhynchus kisutch*)

Randal G. Lake and Scott G. Hinch

Abstract: To determine the roles of suspended sediment angularity and concentration as contributors to stress and mortality in salmonids, we exposed juvenile coho salmon (*Oncorhynchus kisutch*) to anthropogenically derived "extremely angular" and "round" silicate sediments over a range of concentrations in 96-h experiments. Stress responses (e.g., decreased leukocrit) were elicited by exposure to both sediment shapes when concentrations were >40 g·L^{-1}, corresponding to the minimum concentration at which physical gill damage was observed. Extremely angular sediments also caused stress responses (e.g., elevated hematocrit, decreased leukocrit) at concentrations <41 g·L^{-1}. However, we found no difference between sediment shapes in causing mortality at any sediment concentration. Further, mortalities were not observed until concentrations were about 100 g·L^{-1}, a value that is about an order of magnitude greater than high natural concentrations in salmonid rivers. Natural fluvial suspended sediments cause fish stress and mortality at much lower concentrations than we found with our anthropogenically derived suspended sediments.

Résumé : Pour déterminer les effets de l'angularité et de la concentration des sédiments en suspension en tant qu'agents de stress et facteurs de mortalité chez les salmonidés, nous avons exposé des saumons cohos (*Oncorhynchus kisutch*) juvéniles à des sédiments de silicate d'origine anthropique « extrêmement anguleux » et « arrondis » à des concentrations diverses dans des essais de 96 h. Un stress (p. ex., réduction du leucocrite) a été induit par l'exposition aux deux types de sédiments quand leurs concentrations étaient supérieures à 40 g·L^{-1}, concentration minimale à laquelle des dommages physiques aux branchies ont été observés. Les sédiments extrêmement anguleux ont aussi causé un stress (p. ex., hématocrite élevé, leucocrite réduit) aux concentrations inférieures à 41 g·L^{-1}. Cependant, les sédiments des deux types, à quelque concentration que ce soit, ne causaient pas une mortalité plus élevée l'un que l'autre. De plus, pour qu'il y ait mortalité, il fallait que les concentrations atteignent environ 100 g·L^{-1}, soit une valeur environ dix fois supérieure aux concentrations naturelles élevées observées dans les rivières à saumon. Les sédiments naturels en suspension dans les cours d'eau causent un stress et peuvent tuer des poissons à des concentrations beaucoup plus faibles que celles des sédiments en suspension d'origine anthropique utilisés dans nos essais.

[Traduit par la Rédaction]

Introduction

Laboratory and mesocosm studies have demonstrated that high concentrations of suspended sediment can cause physiological stress and mortality in juvenile fish (McLeay et al. 1987; Servizi and Martens 1987, 1991; reviewed in Newcombe and MacDonald 1991; reviewed in Waters 1995). Yet, juveniles of many fish species occur and thrive in rivers and estuaries that have naturally high concentrations of suspended sediments (e.g., Murphy et al. 1989; Northcote and Larkin 1989). This apparent paradox can be partly explained by the fact that high turbidity, caused by suspended sediments, can reduce the risk of capture by visual predators and thereby enhance survival of juvenile fish prey (Gregory 1993, 1994; Gregory and Northcote 1993; Gregory and Levings 1998). Sediment particle size is also an important factor: high concentrations of suspended sediments composed of relatively small particles can cause fewer deleterious effects to salmonids than those composed of larger particles (Servizi and Martens 1987). Duration of exposure to high concentrations of suspended sediments must also be considered (Newcombe and MacDonald 1991).

It has been frequently speculated that the shape of suspended sediment particles may affect physiological stress and mortality in fish (Noggle 1978; Langer 1980; Berg and Northcote 1985; Servizi and Martens 1987). It is conceivable that high concentrations of highly angular particles could cause significant gill damage and lead to mortality; however, no research has been conducted to examine the effects of suspended sediment particle shape on fish. The purpose of this study was to determine if suspended sediment angularity contributes to mortality and sublethal stress in juvenile coho salmon (*Oncorhynchus kisutch*). We accomplished this by exposing fish to extremely angular and nonangular (i.e., round) suspended sediments over a range of suspended sediment concentration levels in 96-h experi-

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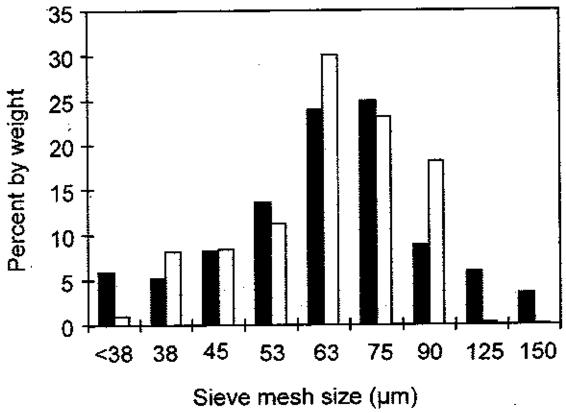
Fig. 1. Histograms showing the percent by weight of extremely angular (solid bars) and round (open bars) suspended sediments that were retained on U.S. Standard stacked sieves. The values represent the mesh size of each sieve. The value to the far left represents sediments not retained on any sieve.



ments and evaluated mortality rates, morphological body changes, and hematological stress responses.

Methods

Angular sediments (crushed silica) were obtained from OCL Industrial Materials Ltd., Surrey, B.C. Crushed silica is used in sand-blasting and can be purchased in graded sizes. The smallest size was used because its size distribution is similar to locally occurring suspended sediments from the Fraser River (see the particle size distribution given in Servizi and Martens (1987)). Round sediments (silica glass beads) were obtained from Canasphere Industries Ltd., Burnaby, B.C. Silica beads are manufactured for use in sand-blasting and for hardening metal surfaces (termed "peening"). We obtained a graded size distribution similar to that of the crushed silica. We determined the exact size distribution of both sediment types by screening particles through U.S. Standard screens (Fig. 1). A sediment extract was obtained from both angular and round silica by mixing 200 g silica·L water^{-1} and allowing the sediment to settle from suspension. Silica particles have residual amounts of hydroxyl ions on their surface as a by-product of their manufacturing; therefore, they were soaked in glacial acetic acid until the sediment extract approximated pH 7. These "treated" sediments were used in all experimental trials. X-ray diffraction revealed that both the round and angular sediments had the same mineral makeup, which was primarily quartz with some feldspar and traces of calcite, mica, and chlorite (L. Lavkulich, Institute for Resources and Environment, University of British Columbia, Vancouver, B.C., personal communication).

The aspect of particle shape that we were most interested in was "roundness", which refers to the sharpness of the corners and edges of the grain. A common method and the one that we used to estimate roundness is visual comparison of grains with standard images of grains of known roundness (Blatt et al. 1972). Natural sediments fall into one of six roundness categories ranging from very angular to well rounded. Our angular sediments had much sharper edges than those classified as very angular by Blatt et al. (1972); thus, we subjectively termed ours extremely angular. Our round sediments were similar in shape to the well-rounded sediments (Blatt et al. 1972). For comparative purposes, we refer readers to Servizi and Martens (1987) who provided photomicrographs of the sediments used in their bioassays. Based on the Blatt et al. (1972) scale, about 90% of Servizi and Marten's (1987) sediments would be classified as subangular to very angular whereas <5% of their sediments would be considered extremely angular. None of their sediments would be classified as well rounded.

Coho salmon eggs, obtained from Inch Creek, Mission, B.C., were fertilized and embryos were reared to the parr stage at the Department of Fisheries and Oceans' Cultus Lake Laboratory, Cultus Lake, B.C. Both facilities are situated near one another in the southern portion of the Fraser River watershed near Vancouver, B.C. Fish were reared under a natural photoperiod in troughs with Cultus Lake water at 6–8°C. Fish were fed a maintenance ration throughout the period of our experiments but were not fed 2 days prior to or during trials. Fish used in all trials ranged from 80 to 111 mm in length.

Our study was conducted using the same laboratory facilities, apparatus, protocol, and water supply as those of Servizi and Martens (1987, 1991). Their papers should be consulted for additional methodological details not presented herein. Eight 35-L inverted cone-shaped vessels, each containing 30 L of water, were used to expose juvenile coho to suspended sediments for a 96-h period that we termed a "trial". A circular cage, 30 cm diameter × 18 cm high, held five fish and was placed at a depth of 22 cm in each vessel. All vessels were partially submerged in a large circular tank to maintain constant temperatures. A 1-hp recirculation pump was externally mounted on each vessel to keep sediments in suspension. Sediments were pumped out a threaded hole in the narrow inverted tip of the vessel and back into the open top of the vessel.

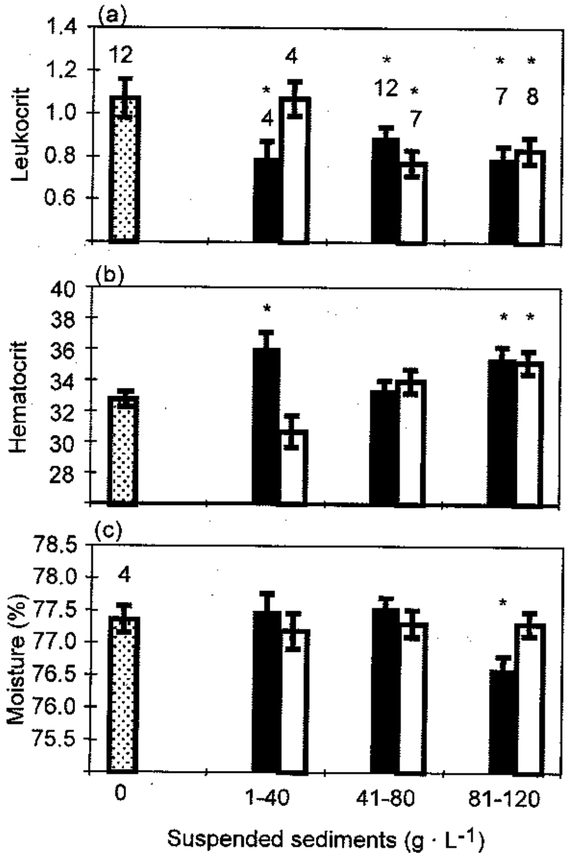
For each trial, four vessels were randomly assigned to contain round suspended sediments and the other four to contain angular suspended sediments. Trials were run from late November 1994 to early January 1995. In the first trial, no suspended sediment was added to the vessels. In each subsequent trial, concentration in the vessels were increased by 50 g·L^{-1} until the concentration was 200 g·L^{-1}, a concentration that preliminary work suggested would cause 100% mortality using either sediment shape. Suspended sediment concentrations were then decreased by 50 g·L^{-1} in each subsequent trial until the last trial, which contained no suspended sediments. In total, 10 trials were carried out. Due to mechanical problems with pumps, some vessels were excluded from some trials.

Due to among-pump variability in performance, the concentration of suspended sediment that fish encountered deviated from that intended. Twice daily over the 96-h trial, we sampled water from the top of each cage using an open scintillation vial that was capped before withdrawal. Each sample was vacuum filtered through preweighed Whatman No. 1 filter disks, dried at 105°C for 1 h, and reweighed to the nearest 0.01 mg. An average of these eight samples was used as the vessel-specific measure of sediment concentration encountered by fish.

Dead fish were removed daily from each vessel throughout the trials. At the end of each 96-h trial, live fish from vessels in which a minimum 60% survived (three out of five fish) were sacrificed by striking the back of the head and used for assessments of sublethal, physiological stress. We examined fish for hematocrit (percentage of red cells in the blood), leukocrit (percentage of white cells in the blood), and percent body moisture. Altered levels of these characteristics, whether they are an increase or decrease relative to baseline levels, suggest that physiological condition has been altered and are general indicators that salmonids are experiencing physiological stress (McLeay et al. 1987; Fagerlund et al. 1995).

Caudal peduncles were severed and blood collected with heparinized microhematocrit tubes. Blood was centrifuged at 11 500 rpm for 3 min. Calipers and a dissecting microscope were used to measure hematocrit and leukocrit as a percentage of the total blood sample volume. Samples were cooled on ice while the measures were made. Body moisture was estimated by removing a cross-sectional piece of the caudal peduncle, patting dry on both

Fig. 2. Average (±1 SE) (*a*) leukocrit, (*b*) hematocrit, and (*c*) percent body moisture for juvenile coho salmon exposed for 96 h to extremely angular (solid bars) and round (open bars) suspended sediments of differing concentrations. Stippled bars represent trials without suspended sediments (e.g., controls). Unless otherwise indicated, the number of trials (e.g., sample size) used in the calculation of the average for that particular sediment concentration is indicated on the top of the leukocrit bars. Asterisks indicate treatments that differed from the control ($P < 0.05$).



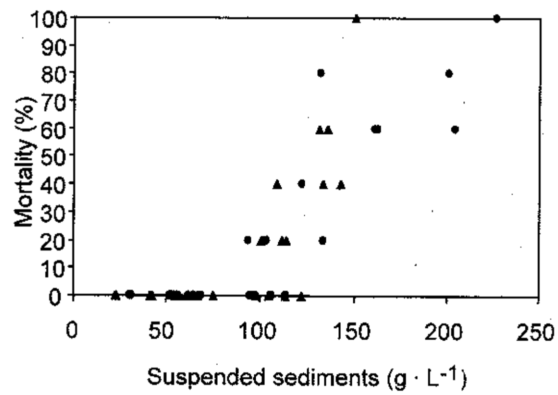
surfaces, and weighing to the nearest 0.01 mg. Cross sections from all fish were oven dried at 90° for 24 h and reweighed. Percent body moisture was calculated by dividing the difference in wet and dry weight by the wet weight and multiplying by 100. The responses of individual fish (e.g., leukocrit, hematocrit, moisture percentages) were averaged within vessel, and it is these mean values that were used as individual replicates in the statistical analyses of sublethal responses.

We qualitatively assessed physical damage to fish by dissecting the first left gill arch from each fish and examining it by wet mount under a microscope at 60× magnification for possible damage or abnormalities of filaments and lamella. We also examined fish for the presence of infection or parasites.

During the trials, dissolved oxygen concentrations were maintained at 11.7–12.3 $mg \cdot L^{-1}$ (99–103% saturation). Temperature ranged from 6 to 7.5°C. The pH ranged from 7.0 to 8.2. The lake water supply is regularly monitored and is relatively invariant with respect to turbidity (2 NTU), conductivity (168 $\mu S \cdot cm^{-1}$), and alkalinity (60 mg $CaCO_3 \cdot L^{-1}$) (Servizi and Martens 1991).

We used two-way ANOVA's to examine the relative roles of sediment concentration and shape on hematocrit, leukocrit, and percent body moisture. We used one-way ANOVA's to compare differences in these variables between controls (trials with no added suspended sediments) and treatments (trials with differing combinations of suspended sediment concentration and shape). The effects of sediment concentration and shape on fish mortality were examined by regressing percent mortality on suspended sediment concentration separately for each sediment shape. We then used ANCOVA to compare these linear regression relationships.

Fig. 3. Percent mortality of juvenile coho salmon after a 96-h exposure to a range of suspended concentrations of extremely angular (triangles) and round (circles) sediments. Twelve observations of 0% mortality are hidden.



Results

Vessel-specific survival rates of ≥60% were required for data to be included into the analyses on sublethal responses. However, few vessels with concentrations >120 $g \cdot L^{-1}$ met this criterion; therefore, we considered this our maximum value for the sublethal analyses and grouped vessels into the following three treatment categories: low (1–40 $g \cdot L^{-1}$), medium (41–80 $g \cdot L^{-1}$), and high (81–120 $g \cdot L^{-1}$). Variation in percent body moisture was weakly associated with variation in sediment concentration ($P = 0.051$); high concentrations produced relatively lower percent body moisture. There was no effect of sediment shape on body moisture ($P = 0.616$). The interaction between concentration and shape was significant ($P = 0.037$). High concentrations of angular sediment caused relatively low percent body moisture, and this category was the only one that significantly differed ($P = 0.032$) from the control category (Fig. 2). High sediment concentrations produced high hematocrit ($P = 0.039$) (Fig. 2) compared with low concentrations. There was no effect of sediment shape on hematocrit ($P = 0.198$). A significant interaction was present ($P = 0.006$), as low-concentration angular sediments produced a relatively high amount of hematocrit (Fig. 2). Hematocrit in the control category was lower than that observed in the high-concentration angular, high-concentration round, and low-concentration angular categories ($P \leq 0.03$ for each) (Fig. 2). Neither sediment concentration nor sediment shape, as main factors, explained significant variation in leukocrit ($P = 0.282$ and $P = 0.743$, respectively). The interaction was significant ($P = 0.027$), however; low concentration of round sediments caused relatively high leukocrit (Fig. 2). Leukocrit in the control category was higher than that observed in any of the other categories ($P \leq 0.03$ for each) (Fig. 2) but did not differ from that in the low-concentration round category ($P = 0.959$).

Mortality was not observed, for either sediment shape, until sediment concentration reached about 100 g·L^{-1}. At and above this concentration, mortality steadily increased with increasing suspended sediment concentration (Fig. 3). We found no differences in the slopes ($P = 0.470$) and intercepts ($P = 0.471$) of the two linear regression relationships, one for the round and one for the angular sediments, of percent mortality on suspended sediment concentration. The common regression equation, obtained by pooling round and angular data, was as follows: percent mortality = 0.046 × concentration – 25.169 ($P < 0.001$, $r^2 = 0.621$, $n = 59$). The LC_{50} estimated using this equation was 164.5 g·L^{-1}. No mortalities were recorded in the control trials. We were able to run five trials of round sediment at concentrations >150 g·L^{-1} but only one of angular sediment because at these high concentrations of angular sediment, pumps failed due to abrasion of their internal components. However, the slopes of the round and angular treatments did not differ from each other ($P > 0.05$) even when these high concentrations were excluded from the ANCOVA.

Visual examination of gills by wet mount revealed erosion at the distal end of the filament tips in suspended sediment concentrations >41 g·L^{-1} for both angular and round sediments. We could not visually distinguish any obvious differences in the level of erosion caused by round versus angular sediments, nor could we detect higher levels of erosion at the highest sediment concentrations. Because we had very few fish exposed to high concentrations of angular sediments, we were limited in our ability to detect effects at that combination of sediment shape and concentration. We observed no changes to gills for either sediment shape at concentrations ≤41 g·L^{-1}. No gills in live fish were observed to be clogged with suspended sediments, even at the highest concentrations. However, sediments were observed clogging the gills of morbid fish (e.g., fish that had loss equilibrium) and dead fish. Round suspended sediments were observed lined up in a single row between some gill filaments, but this was not observed for angular sediments. No other abnormalities, infections, or parasites were observed.

Discussion

The tolerance of salmonids to suspended sediments has been the subject of several studies (e.g., Noggle 1978; Newcombe and Flagg 1983; McLeay et al. 1987; Redding et al. 1987; Servizi and Martens 1987, 1991; Servizi and Gordon 1990), yet there have been no studies on effects of particle angularity on physiological stress and mortality. We found that stress responses (e.g., decreased leukocrit) were generally elicited by exposure to both sediment shapes when their concentrations were in excess of 40 g·L^{-1}, which corresponded to the minimum concentration at which physical gill damage was noted. Servizi and Martens (1991) suggested that tolerance to suspended sediments is related to oxygen content of the water, oxygen transfer across gill membranes, metabolic demands, and capacity to perform work. Although oxygen in our experiments was near saturation, the gill damage that we observed at high concentrations of suspended sediments could have caused anoxia and stress. In contrast with round sediments, angular sediments elicited stress responses (e.g., elevated hematocrit, decreased leukocrit) at relatively low sediment concentrations (<41 g·L^{-1}). Although we noted no damage to gills at these concentrations, it is possible that gills may have been irritated by angular sediments, resulting in increased mucus production and poorer oxygen transfer. Martens and Servizi (1993) found that juvenile coho salmon that were exposed to natural Fraser River suspended sediment for 96 h at concentrations of 16–41 g·L^{-1} had on average 1500 sediment particles per lamellae lodged intracellularly into gill epithelia. All particles were irregular and angular in shape.

It has been suggested that mortality rates of salmonids may increase with exposure to particles of increasing angularity (Noggle 1978; Langer 1980; Berg and Northcote 1985; Servizi and Martens 1987). Indeed, our sublethal measures demonstrated that extremely angular sediments placed fish under stress at lower concentrations than did round sediments. Therefore, we had expected that the concentration for initiation of mortality and the LC_{50} would be lower for extremely angular sediments. Instead, we found that both types of sediment caused similar mortality rates. The causes of mortality are not clear. Sediment did not appear to clog gills except in morbid or dead fish. Even at the highest sediment concentrations, oxygen remained saturated in the vessels, yet anoxia associated with gill impairment, along with other cumulative stressors (e.g., impaired osmoregulation, reduced metabolic capacity to clear sediment from gills), likely contributed to mortality (Servizi and Martens 1991).

Our relatively large LC_{50}, 164.5 g·L^{-1}, was very surprising for several reasons. First, in most natural systems, salmonids are unlikely to encounter the concentrations at which we first observed mortalities (about 100 g·L^{-1}). For instance, highly turbid large Canadian rivers such as the Red Deer and Peace have maximum daily suspended sediment concentrations of only 11–12 g·L^{-1} (Anonymous 1980), and glacial-fed alpine rivers experience occasional peaks of 14–15 g·L^{-1} (Gurnell and Clark 1987). The Fraser River's suspended sediment daily maximum rarely exceeds 1 g·L^{-1} and generally is <0.3 g·L^{-1} (Anonymous 1992). Second, Servizi and Martens (1991) exposed juvenile coho salmon to natural Fraser River suspended sediments using the identical test conditions and found a 96-h LC_{50} of only 22.7 g·L^{-1}. Their sediments had a very similar size distribution to ours but were less angular. Using the identical apparatus and natural sediments, juvenile sockeye salmon (*Oncorhynchus nerka*) had a 96-h LC_{50} of 17.6 g·L^{-1} (Servizi and Martens 1987) and juvenile chinook salmon (*Oncorhynchus tshawytscha*) 31 g·L^{-1} (Servizi and Gordon 1990).

Why natural sediments of intermediate shape would cause mortality at such relatively low suspended concentrations, compared with our values generated with anthropogenically derived sediments, is perplexing. Natural fluvial suspended sediments electrostatically sequester heavy metals and adsorb large organic molecules from solution to their surfaces (Allen 1986; McCallum 1995). Regardless of dissolved aqueous concentrations, these compounds can remain in high concentrations associated with sediments (Giesy and Hoke 1991). Upon contact with gills, electrostatically bound materials could be released from sediments and passively diffuse into epithelial mucus (Coombs 1980; Tessier et al.

1984). Sediment particles themselves, and their sorbed contaminants, can also be taken up via gill epithelia (Coombs 1980; Tessier et al. 1984; Martens and Servizi 1993). Martens and Servizi (1993) found that only a brief exposure of juvenile salmonids to suspended sediments was needed for small particles to be quickly phagocytosed by epithelial gill cells and then transferred into parenchyma and spleen tissues.

The sediments used by Servizi and Martens (1987, 1991) were taken from deposition zones in the lower Fraser River, an area influenced by non-point-source pollution that contains high levels of sediment-bound contaminants (Hall et al. 1991; K. Hall, Institute for Resources and Environment, University of British Columbia, Vancouver, B.C., personal communication). Although the sediments used by Servizi and Martens (1987, 1991) were repeatedly rinsed in clean water prior to use in their bioassays, compounds sorbed to particle surfaces would not have been removed by this approach. Our anthropogenically derived sediments, on the other hand, were quarried from deposits on land and were not exposed to natural riverine metals and organic compounds after they were crushed and created. Further, the acid bath that we gave to the silica sediments to remove hydroxyl compounds associated with the bead manufacturing process should have removed any residual metal, and possibly organic, compounds that were sorbed on particle surfaces. Thus, the likelihood of contaminant influences on our bioassays was minimal.

The only study that we are aware of that conducted bioassays using suspended sediments of anthropogenic origin and size distribution similar to ours was that of McLeay et al. (1987) who exposed juvenile Arctic grayling (*Thymallus arcticus*) to placer mine sediments. Such sediments are created by crushing gravel. Although not reported in their work, we suspect that these sediments would have been highly angular. They observed 20% mortality at a concentration of 100 $g \cdot L^{-1}$, a finding nearly identical to ours (see Fig. 3). The results of our bioassays with extremely angular sediments, along with this observation from McLeay et al. (1987), suggest that suspended sediment particle angularity may not be a main factor responsible for acute lethality in juvenile fish.

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Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival

E. Al Shaw and John S. Richardson

Abstract: Elevated concentrations of inorganic sediment supply in streams may impair many biological functions. However, the contribution of exposure duration to the observed impacts has not been previously considered. We evaluated the effects of sediment pulse duration using 14 streamside flow-through experimental channels, each of which contained a naturally colonised invertebrate assemblage and 10 rainbow trout (*Oncorhynchus mykiss*) fry. Channels were exposed to fine sediment pulses of constant concentration but varied pulse duration (ranging from 0 to 6 h) every second day over 19 days. Total abundance of benthic invertebrate and family richness declined as sediment pulse duration increased. Invertebrate drift total abundance increased as pulse duration increased; however, family richness of drift decreased. Trout length and mass gain over the 19-day period was negatively correlated with pulse duration. Path analysis suggests that the direct effects of fine sediment on trout (impaired vision leading to reduced prey capture success and (or) increased metabolic costs from physiological stress) are more important to trout growth than indirect effects (decreased drift and benthic invertebrate richness and drift abundance).

Résumé : Les concentrations accrues des apports de sédiments inorganiques dans les cours d'eau peuvent perturber de nombreux processus biologiques. Cependant, les impacts de la durée de l'exposition n'ont pas été examinés jusqu'à maintenant. Nous avons évalué les effets de la durée de l'afflux de sédiments au moyen de 14 canalisations expérimentales installées en bordure d'un cours d'eau, chacune colonisée naturellement par une communauté d'invertébrés et dix alevins de Truite-arc-en-ciel (*Oncorhynchus mykiss*). Les canaux ont été exposés à des afflux de sédiments fins à concentration constante, mais dont la durée variait de 0 à 6 heures tous les 2 jours sur une période de 19 jours. L'abondance totale des invertébrés benthiques diminuait en fonction de l'allongement de la durée de l'afflux de sédiments et la richesse en familles déclinait. La densité totale de la dérive des invertébrés augmentait avec la durée de l'afflux, mais la richesse en familles diminue. La croissance en longueur et en masse des truites pendant la période de 19 jours était en corrélation négative avec la durée de l'afflux. Une analyse des coefficients de direction laisse croire que les effets directs des sédiments fins sur la truite (une vision réduite qui diminue le succès de la capture des proies et/ou des coûts métaboliques accrus dus au stress physiologique) affectent plus la croissance que les effets indirects (diminution de la dérive, de la richesse des invertébrés benthiques et de la densité de la dérive).

[Traduit par la Rédaction]

Introduction

The effects of increased fine inorganic sediment loads on stream communities have been well documented (summaries in Newcombe and MacDonald 1991; Newcombe 1994; Waters 1995). Many studies have reported a decrease in invertebrate abundance and a change in community composition resulting from several sediment-mediated mechanisms. Culp et al. (1986) showed that invertebrates become dislodged into the water column by rolling or saltating particles. Many taxa rely on a filter-feeding apparatus to remove fine particulate organic matter from the water column, and these can become clogged by sediment, thereby reducing feeding efficiency. Also, the distribution of grazing invertebrates may be affected by the smothering of algal habitat and abrasion of cells by elevated sediment concentration (Vuori and Joensuu 1996). Regardless of the particular mechanism, in all cases an increase in fine sediment load generally causes invertebrates to enter the water column and drift.

Changes to the quantity and composition of available food resources (e.g., aquatic invertebrates) can directly influence the growth of resident fish. Sediment can also indirectly

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affect fish growth through modifications of behaviour and habitat. At relatively low suspended sediment concentrations, turbid conditions may be viewed as cover for foraging fish (Gregory and Northcote 1992), leading to increased foraging effort because of a perceived reduction of predation risk (Gregory and Levings 1996). In contrast with elevated feeding effort is a severe reduction in foraging efficiency with increased turbidity (Vogel and Beauchamp 1999). Direct effects of increased sediments on fish vary with suspended matter concentration (Newcombe 1994), degree of sediment deposition (Chapman 1988), particle size distribution and type of sediment (Lake and Hinch 1999), and life stage at which the fish is exposed (Servizi and Martens 1990). Sediment-induced mortality and decreased survival rates have been noted in only a few studies (Sigler et al. 1984; McLeay et al. 1986), although the concentration administered to cause mortality greatly exceeds elevated field levels and could only be achieved in experimental situations (Lake and Hinch 1999). Suspended sediments in streams can induce coughing or gill flaring (Berg and Northcote 1984) and decrease respiratory capabilities through gill abrasion (Herbert and Merkins 1961). Even relatively low amounts of deposited sediment can limit inter-gravel water exchange, reduce interstitial dissolved oxygen, and effectively smother developing eggs and alevins (Scrivener and Brownlee 1989). Early life stages tend to be more susceptible to sediment due to their limited movement and tolerance (Servizi and Martens 1990).

Newcombe and MacDonald (1991) proposed that sediment in streams be viewed as eliciting a dose-dependent response, incorporating both concentration and duration of exposure when predicting an organism's reaction. The dose model is used frequently by toxicologists describing biological responses to chemicals in the environment (Liber et al. 1992). With few exceptions (Newcombe 1994; Larkin et al. 1998), duration of exposure to sediments is rarely evaluated or even noted. Newcombe and MacDonald (1991) suggested a stress index to rank any potential impacts of sediment based on the dose–response model. When studies were reanalysed using estimates of duration, the new model suggested dose was a better indicator and predictor than concentration alone. Although many of the studies that were used to derive the stress index compared several concentrations and types of sediment, no study has exclusively tested the contribution of sediment pulse duration to changes in stream communities.

The purpose of this study was to address two key questions regarding the biological impacts of sediment in lotic ecosystems. First, we sought to determine the effects of fine sediment pulse duration on drifting and benthic invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) fry growth and mortality. The second objective was to quantify and compare the direct and indirect (mediated through the invertebrate food supply) effects of sediment exposure on fish growth.

Methods

Study site

This experiment was performed at Moffat Creek near Horsefly, British Columbia, approximately 80 km east of Williams Lake. Moffat Creek is a third-order tributary of the Horsefly River, which flows through Quesnel Lake into the Fraser River. The Fraser Plateau is covered by unconsolidated geologic materials in the form of fluvial, lacustrine, and colluvial deposits (Lord 1984), which tend to be highly mobile and become entrained and transported in the water column. Generally, there is little riparian canopy as much of the land has been cleared to the streambank for livestock grazing and hay crops.

Experimental design

To test the effects of fine sediment pulse duration on invertebrate assemblages and fish growth, an experiment was conducted by creating a gradient of duration treatments. Treatments consisted of seven sediment pulse durations ranging from 0 (control) to 6 h, pulsed every second day for 19 days (Table 1). All sediment pulses were initiated at 1000 h and continued for the predetermined duration. Each treatment was replicated twice. The dose regime was intended to mimic a typical disturbance activity, such as regular movement of cattle through riparian areas, riparian timber harvesting and frequently occurring thunderstorms which repeatedly introduce excess sediment pulses over an extended period of time. To exclusively test pulse duration, the concentration of each sediment pulse entering the stream channels was held at approximately 700 $mg \cdot L^{-1}$ throughout the dosing periods. This concentration has been shown to induce a response in fish and invertebrates (McLeay et al. 1986; Newcombe and MacDonald 1991). Also, a permanent B.C. Ministry of Forests water-quality monitoring station, situated downstream from our site, has often recorded this concentration and higher during rain events.

Experimental stream channels were constructed immediately adjacent to Moffat Creek. Water flow was redirected from the creek in polyvinyl chloride (PVC) pipe and partitioned into two 500-L headtanks. Each headtank supplied water to eight experimental channels, 16 in total, at approximately 0.5 $L \cdot s^{-1}$ each. Fourteen channels were used for sediment treatments and two channels were devoted to measuring invertebrate immigration. All channels had a surface area of 1.5 m^2 (7.5 m long × 0.2 m wide × 0.2 m deep) and were set to a 1% slope to mimic the local stream gradient. Substrate particles were sieved to fit the gravel–pebble range (6.4 mm to 20 mm) approximating the substrate of Moffat Creek. Substrate depth was approximately 8 cm throughout and no alteration of the substrate was initiated prior to the experiment. Small pool and riffle areas were created naturally around fish cover objects (10-cm section of PVC pipe cut along its length) placed in each of the channels (10 per channel). Water depth was approximately 2.5 cm at the upstream end and 10 cm at the outflow.

Invertebrates

Invertebrate assemblages were established in the channels through active addition of kick samples collected from Moffat Creek, and by natural drift through the intake pipe. Invertebrates were allowed to colonise for three weeks before the experiment began. Benthos-sampling baskets (surface area, 104 cm^2; volume, 500 mL; 12 per channel) were made from clear plastic tubs, which contained numerous holes over their entire surface, and were buried flush with the substrate surface to allow free movement of invertebrates. During each sampling event, three baskets were removed from the channels and the contents preserved with a 5% formaldehyde solution.

Drift nets (250-μm mesh; 1 per channel) were placed on the downstream end of the channels. During sampling events, drift nets were set at the beginning of the sediment pulse and remained in place to collect invertebrate drift for 24 h, thus including invertebrates that exhibit a delayed response to disturbance. All drift samples were washed into a labelled plastic jar and preserved with a 5% formaldehyde solution. In the laboratory, each invertebrate sample was rinsed through a 2-mm and 425-μm sieve. The invertebrates retained by these sieves were enumerated and identified using Merritt and Cummins (1996).

Table 1. Fine sediment treatments applied to experimental channels.

Pulse duration (h)	No. sediment pulses	Mean sediment concentration ($mg \cdot L^{-1}$)	Standard error	Number of replicate channels	Dose ($mg \cdot L^{-1} \cdot h^{-1}$)
0	N/A	N/A	N/A	2	0
0.5	10	695.0	15.8	2	3 475
1.0	10	699.0	14.7	2	6 990
3.0	10	701.5	17.9	2	21 045
4.0	10	704.5	12.1	2	28 180
5.0	10	702.0	13.8	2	35 100
6.0	10	705.0	13.7	2	42 300

Note: Each of the seven treatments was replicated twice giving 14 experimental units. Dose is given as the total administered over the duration of the 19-day experiment and is calculated as the product of concentration and duration (product of pulse length and number of pulses). N/A, not applicable.

Rainbow trout

To examine the effect of sediment pulse duration on rainbow trout we collected swim-up fry, using baited minnow traps and electrofishing gear, from the Horsefly River sockeye spawning channels in the town of Horsefly, B.C. (Quesnel Lake stock). Swim-up fry (n = 140) were anesthetized with MS222, mass and length measured, examined for signs of disease, and then randomly placed in the experimental channels five days prior to treatment. Each channel contained 10 fish, giving a density of 6.7 $fish \cdot m^{-2}$. This stocking density is similar to or less than other studies examining the effects of sediment on fish growth (Sigler et al. 1984; McLeay et al. 1986). A single pretreatment mortality was removed and replaced. The average size of trout prior to treatments was 45.8 mm and 1.04 g. Because of their small size, fry were not individually marked; mean mass and length within channels were used for statistical analysis (in all cases n = 14). At the conclusion of the experiment, fish were removed from the channel, immediately anesthetized, and mass and length recorded.

Sediment

Sediment particles used in all treatments were collected from an exposed streambank 2 km downstream from the experimental stream channel site. This material commonly enters Moffat Creek at several points along its length. Particles passing through a 425-μm sieve (medium sand – silt) were included in the experimental treatments. Prior to entering the channels sediment was combined with stream water in two 200-L tanks (approximately 30 $g \cdot L^{-1}$). Twenty airstones powered by a 90-W aquarium pump, combined with a mechanical stirring device ensured consistent mixing and prevented buildup of sediment in the release valves. When mixed with the incoming stream water, the concentration measured entering the stream channels averaged 704 $mg \cdot L^{-1}$ (Table 1). Sediment concentration was determined by filtering a 500-mL sample, collected where the stream water and sediment slurry combined, through a Whatman GF/C filter. Samples were collected randomly, from all channels, once during each sediment pulse. Sediment pulses were consistently initiated at 10:00 during each sediment event and continued for the designated duration.

To completely characterise sediments used in the experiment and facilitate comparison between studies, the particle size distribution and concentration–turbidity relationships were determined using a standard column of sieves ranging from 355 μm to <63 μm. The mean of 10 samples was calculated.

Sampling regime

Sediment was delivered to each channel, with the exception of the controls, every second day for a total of 10 pulses over 19 days. Invertebrates were collected on four occasions; two days before the first pulse and following the first (day 1), fifth (day 9), and tenth pulses (day 19). Both drift and benthic samples were collected at these times. Drift nets were set on the sampling days just prior to the start of the sediment pulse and were collected after 24 h. A total of 64 drift samples were collected during the experiment. Three benthic samples were randomly collected during each of the four sampling occasions, resulting in 168 samples. Sediment samples and discharge measures were collected during each sediment event. Temperature was constantly monitored with data logging temperature probes placed in the headtanks and the terminal end of two experimental channels.

Statistical analyses

We evaluated between treatment and between sampling date differences using repeated measures analysis of variance (ANOVA). To determine if a relationship exists between sediment pulse duration and invertebrate (drift and benthos) abundance and family richness, regression analysis was performed on each of the four sampling days. We used ANOVA, for each sampling day, to reveal which pulse duration achieved significantly different mean invertebrate abundance and richness measures as well as fish length and weight gain, in comparison to control.

Principal components analysis (PCA), using a correlation matrix, was used to ordinate drift and benthic invertebrates presence–absence data from the final day of the experiment. Families that were not present in more than one of the 14 channels were excluded from this analysis. Also Chironomidae, which appeared in each sample, would not contribute to a PCA based on presence–absence and were therefore excluded. Following the ordination, PCA scores for the primary axis were regressed against pulse duration to determine if the main pattern of invertebrate presence–absence was related to addition of sediment. This procedure was completed for both drift and benthic samples. Correlation coefficients (structure coefficients) were used to identify taxa contributing significantly to the observed assemblage.

Trout body condition was compared between treatments by establishing linear relationships between mass and length gain for each treatment and subsequently tested for similarity in slopes and intercepts using analysis of covariance (ANCOVA). Path analysis was performed using sediment pulse duration, trout mass gain, and invertebrate drift abundance data to quantify and compare direct versus indirect effects of sediment on fish growth.

Prior to conducting analyses, assumptions of normality and heteroscedasticity were tested and natural log transformations were applied where appropriate (Gauch 1982). Only trout mass required transformation. All statistical analyses were performed using the GLM and PRINCOMP procedures in PC SAS (ver. 6.12, SAS Institute Inc., Cary, N.C.) and utilised α = 0.05. Significance levels were adjusted for multiple simultaneous comparisons (10 comparisons, α = 0.005 for drift; 11 comparisons, α = 0.004 for benthos).

Results

There were no significant differences among experimen-

Fig. 1. Relationship between sediment treatments and drift total abundance or family richness at pretreatment (*a*), day +1 (*b*), day +9 (*c*), and day +19 (*d*). Solid lines represent least squared regression lines for total abundance (●) and broken lines for family richness (□). Regression statistics for each line are shown in the upper (abundance) and lower (richness) right corner. In each panel n = 14.

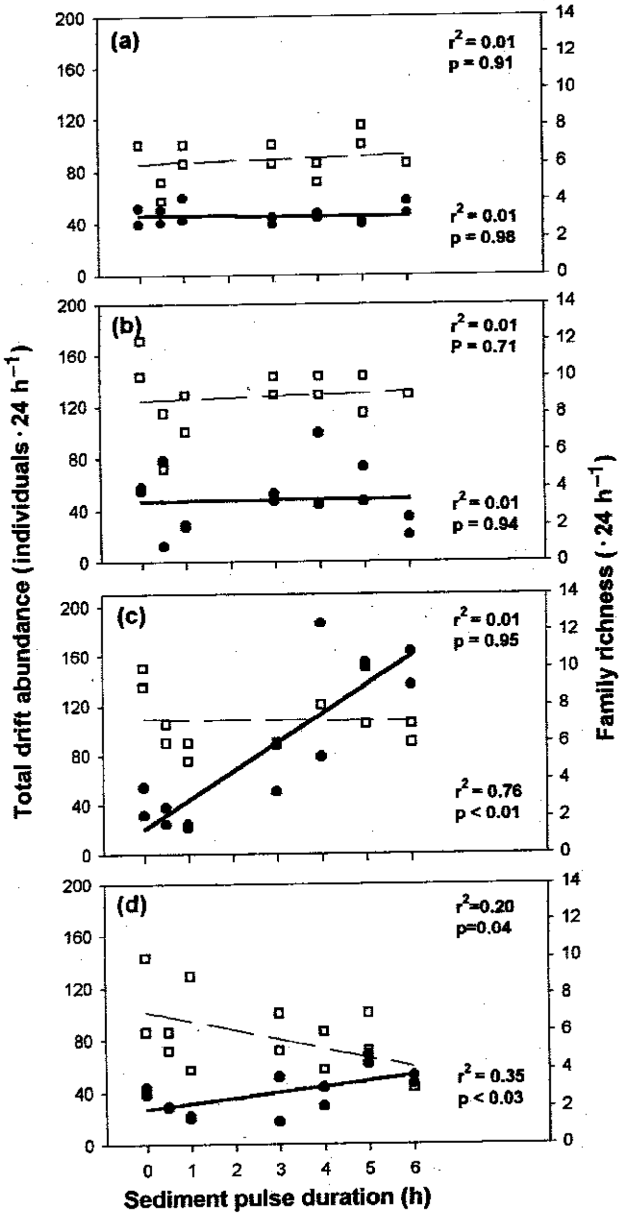
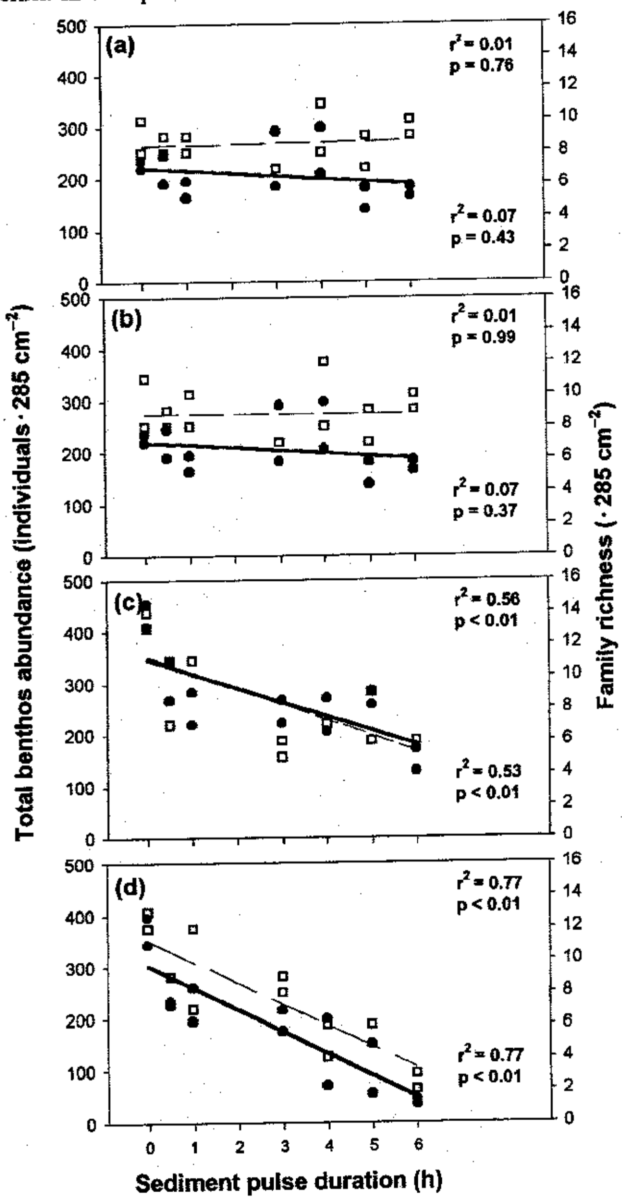


Fig. 2. Relationship between sediment treatments and benthos total abundance or family richness at pretreatment (*a*), day +1 (*b*), day +9 (*c*), and day +19 (*d*). Solid lines represent least squared regression lines for total abundance (●) and dashed lines for family richness (□). Regression statistics for each line are shown in the upper (abundance) and lower (richness) right corner. In each panel n = 14.



tal channels prior to the beginning of the experiment for stream water temperature, discharge, invertebrate abundance and richness (Figs. 1 and 2), or fish mass (ANOVA $F_{6,7}$ = 1.02, p > 0.43) and length (ANOVA $F_{6,7}$ = 1.6, p > 0.15). Water temperature (p = 0.96), discharge (p = 0.56), and sediment concentration (p = 0.53) did not vary among experimental channels for the duration of the experiment. Greater than 90% of the sediment particles, by weight, used in our treatments were <177 μm and 50% were <125 μm. Direct laboratory comparisons of sediment mass per volume and turbidity revealed that the mean sediment concentration administered in our experiment, 704 mg·L^{-1}, corresponds to 23 nephelometric turbidity units (NTU).

A total of 31 different families of invertebrates were identified in the drift and benthic samples. Chironomidae was the most abundant taxon and was the only family to occur in all samples. Baetidae and Limnephilidae were the next most abundant taxa, each accounting for 12% of all invertebrates collected. An average of 65 animals total were collected in each 24-h drift sample and 255 animals (8950 animals·m^{-2})

were collected in each benthic sample. For analysis, each 24-h drift sample was considered individually. The sum of the three benthic samples taken from each channel on each sampling day was used in analyses.

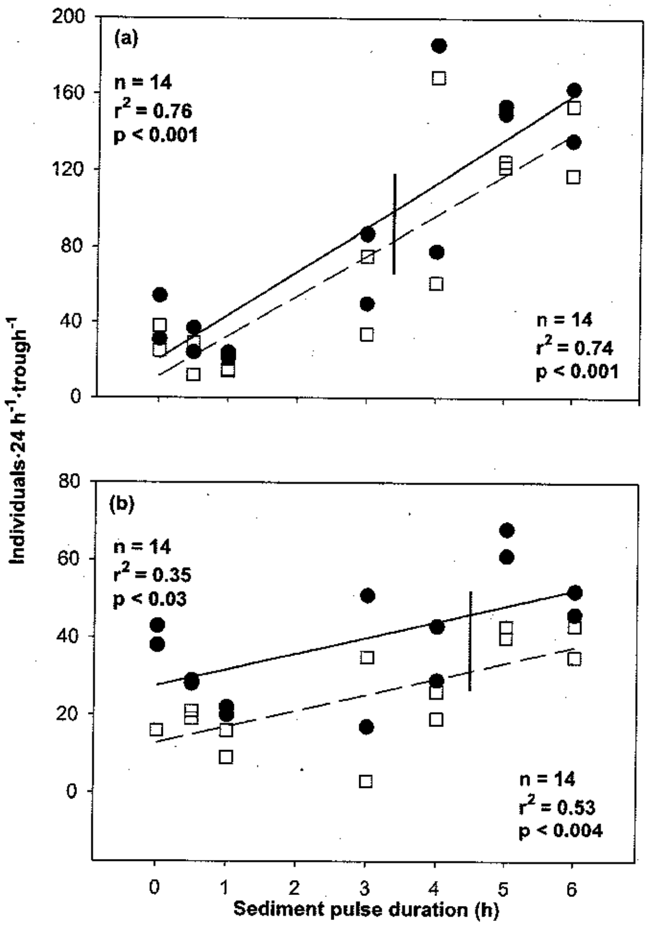
Repeated measures analysis revealed a significant sampling date effect (benthos $F = 9.92$, $p < 0.001$; drift $F = 4.79$, $p < 0.01$), a significant treatment effect (benthos $F = 13.41$, $p < 0.001$; drift $F = 9.4$, $p < 0.001$), and a significant interaction (benthos $F = 6.1$, $p = 0.001$; drift $F = 4.84$, $p = 0.001$). As there was a significant interaction effect, analysis continued by evaluating relationships within a single treatment over time, or across treatments on a single sampling day.

Linear regression analyses found significant relationships between the sediment pulse duration and total abundance for both drift (Fig. 1) and benthic invertebrates (Fig. 2) by day 9. No significant linear relationships, with respect to pulse duration, were found for drift or benthic samples collected prior to the initial pulse or after the first sediment pulse. However, following day 9, abundance and richness in both the drift and benthos showed significant relationships with sediment treatments. Treatment effects became stronger as the number of sediment pulses increased over the duration of the experiment. The only exception was drift abundance, which peaked on day 9. The relationship between the abundance of drifting invertebrates and sediment treatments was driven by the family Chironomidae. The abundance of Chironomidae in drift samples collected on day 9 and 19 showed significant linear relationships with sediment treatments (Fig. 3). Also, as exposure duration increased, the proportion of the drift samples composed of Chironomidae increased. Drift and benthos total abundance significantly varied from control abundance between the 3- and 4-h treatments on day 9 and between the 4- and 5-h treatments on day 19 (ANOVA and multiple range test).

For families other than Chironomidae, the change in abundance did not describe the patterns of variation as well as a measure of presence–absence, as some taxa disappeared as sediment pulse duration increased. PCA axis 1 for drifting insects summarized 27% (Fig. 4*a*) of the total variation and was dependent on variation in Simuliidae (larvae and pupae). PCA axis 1 for the benthic assemblage summarized 39% (Fig. 4*b*) of the variation and was dominated by the presence–absence of the families Elmidae, Nemouridae, Leptophlebiidae, Baetidae, and Heptageniidae. Significant negative relationships ($p < 0.01$) were found between both drift and benthic PCA axis 1 scores when regressed with sediment pulse duration (Figs. 4*a* and 4*b*).

Regression analysis of mass and length gain of rainbow trout fry revealed a linear relationship with sediment pulse duration (Figs. 5*a* and 5*b*). Following the experiment (19 days, 10 sediment pulses) there were significant negative impacts of sediment pulse duration on trout mass (ANOVA $F_{6,7} = 4.52$, $p < 0.03$) and length gained (ANOVA $F_{6,7} = 5.04$, $p < 0.03$) when treatments and controls were compared (Figs. 5*a* and 5*b*). Duncan's multiple range test revealed that significant differences occurred between the 4- and 5-h treatments for length gain, and the 5- and 6-h treatments for mass gain, when compared to the control (Figs. 5*a* and 5*b*). This value is similar to the results for changes in abundance and richness of invertebrates. Trout mortality was not influenced by the sediment treatments (ANOVA $F_{6,7} = 0.14$, $p = 0.71$; Fig. 5*c*).

Fig. 3. Total number of insects (solid line, ●) and total number of Chironomidae (broken line, □) in each drift sample, 9 and 19 days after sediment treatments began. Regression coefficients in the top left corner correspond to all insects (including Chironomidae) and bottom right refer to Chironomidae. Vertical lines indicate significant differences in total abundance and Chironomidae abundance in comparison to control (Duncan's multiple range test).



Trout body condition was evaluated as a change in the linear relationship between length and mass among treatments. When comparing the regression lines between treatments neither the slope ($F_{1,12} = 0.92$, $P = 0.49$) nor the intercept ($F_{1,12} = 1.11$, $P = 0.36$) was altered by sediment introduction.

Path analysis was performed using mean trout mass gain and invertebrate drift data. Path coefficients were calculated as the standardised partial regression coefficients of a multiple regression analysis with sediment pulse duration and drift abundance as the independent variables and trout mass gain ($\log_e$ transformed) as the dependent variable (J.T. Wooton, Department of Ecology and Evolution, University of Chicago, Chicago, Ill., personal communication). The path coefficient between sediment pulse duration and drift abundance is the correlation coefficient between the two variables. The relative strength of the indirect path is the product

Fig. 4. First principal component for invertebrate drift (*a*) and benthos (*b*) based on presence–absence regressed against sediment pulse duration. Regression statistics for each plot are located in top right corner. PC1 values for invertebrate drift increases with increasing presence of Simuliidae pupae and larvae, while PC1 values for benthos increase with the occurrence of Elmidae, Nemouridae, Baetidae, Leptophlebiidae, and Heptageniidae.

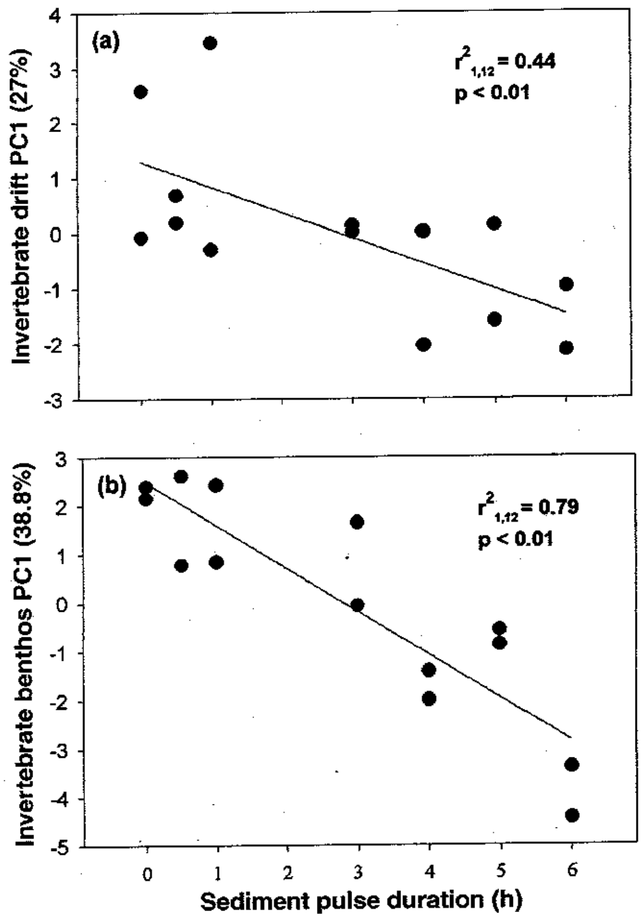
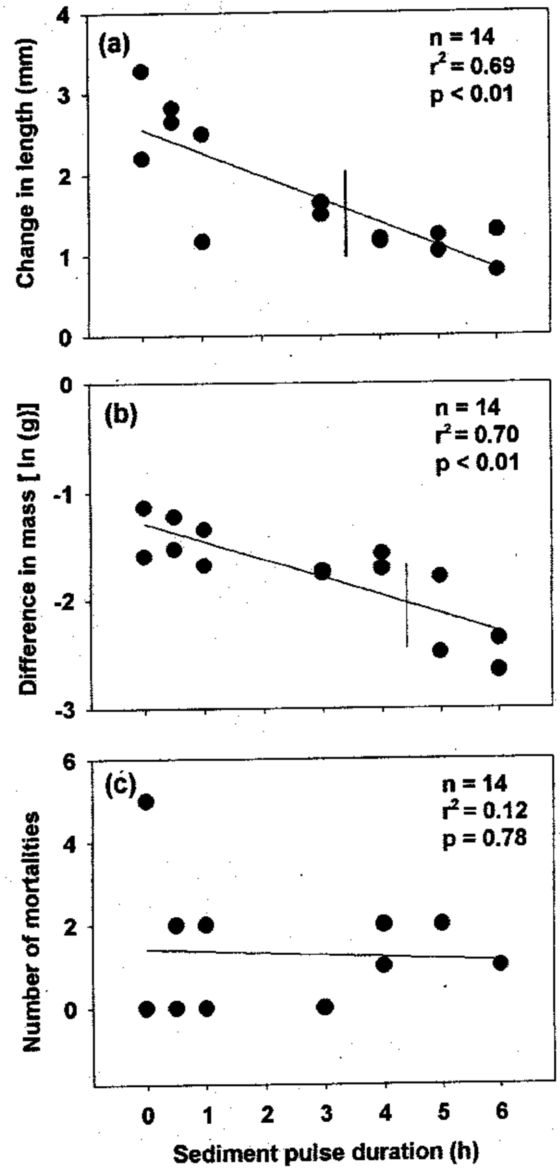


Fig. 5. Effect of pulse duration on (*a*) the change in length, (*b*) change in mass, and (*c*) mortality of trout, over the course of the experiment. Vertical lines indicate significant differences in (*a*) length and (*b*) mass in comparison to control (Duncan's multiple range test). Each point represents the mean value for each channel.



of the two coefficients deriving that path and is compared to the direct path, which has only one coefficient. The direct path, sediment pulse effects on change in trout mass, had a higher path coefficient than the complete indirect path coefficient, indicating greater importance (Fig. 6).

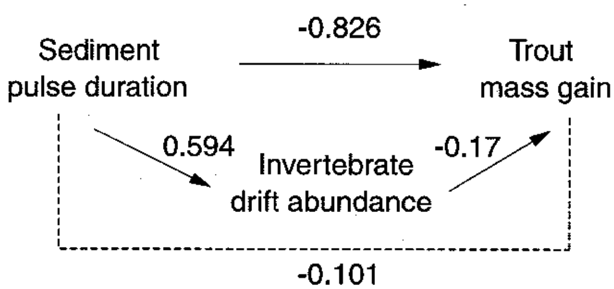
Discussion

By experimentally examining the effects of sediment pulse duration on invertebrate assemblages and trout growth and survival, we quantitatively tested a previously unconfirmed hypothesis and provided indirect evidence as to the mechanism of effect. In this experiment we have shown that the duration of a sediment pulse, given a constant concentration, had a negative effect on the richness and abundance of benthic invertebrates and the richness of drifting invertebrates. However, the total abundance of drifting invertebrates increased over the duration of the experiment. In addition, trout growth suffered as sediment pulse duration increased. Our study design was developed to compare the response of invertebrates and rainbow trout exposed to repeated pulsed-sediment episodes of contrasting duration. Unfortunately, it is not possible to discriminate between cumulative exposure over all pulses and the duration of individual pulses as being responsible for determining impacts on community structure and fish growth.

Benthic and drifting invertebrate assemblages

Duration of sediment exposure influenced the abundance and composition of invertebrate communities in the experimental channels relative to controls. However, the timing of the response was delayed. A dose-dependent response of invertebrate taxa was first noted following the fifth sediment

Fig. 6. Representation of the direct and indirect effects of fine sediment exposure on trout growth. Path analysis coefficients are given for each path. The complete indirect path coefficient is the product of the two coefficients making the path –0.101.



pulse (day 9). By this point, both drift and benthos abundance as well as benthos family richness were altered. Drift abundance rose with treatment severity while benthic abundance and family richness declined. Culp et al. (1986) also found no measurable short-term impact on five of the six numerically dominant taxa exposed to saltating sediment particles. The area surrounding our experimental site is mostly agricultural (both hay and livestock), suggesting a human-altered environment. In addition, exposed lacustrine sediment deposits located along the stream margin introduced particles during rain events (E.A. Shaw, personal observation). Invertebrates would have been previously exposed to sediment events of short duration and therefore may not immediately react to a pulse of sediment. Exposure to a constant source of sediment may also explain the relatively low diversity of disturbance-sensitive taxa (i.e., Ephemeroptera, Plecoptera, and Trichoptera). By day 19, abundance and family richness of both drift and benthic samples showed a response directly proportional to fine sediment exposure.

Ordination revealed a number of invertebrate taxa that were disproportionately affected by fine sediment addition. As sediment pulse duration increased, Simuliidae (larvae and pupae) were present in fewer drift samples. Similarly, Elmidae, Nemouridae, Baetidae, Leptophlebiidae, and Heptageniidae were at lower densities in benthic samples as exposure to fine sediment increased. Members of the families Baetidae (Culp et al. 1986; Vuori and Joensuu 1996), Simuliidae, and the order Plecoptera (Culp et al. 1986; Vuori and Joensuu 1996) have been shown to incur significant declines when exposed to elevated fine sediments. These taxa have been used as indicator organisms in various biomonitoring programs (Barton 1996; Somers et al. 1998) because of their sensitivity to disturbance and associations with clean, cool, running waters and substrates containing only small amounts of fine particles. Conversely, the family Chironomidae tends to increase in abundance or remain unchanged with fine sediment exposure (Culp et al. 1986; Vuori and Joensuu 1996). In our experiment, Chironomidae was the only taxon to increase in abundance with elevated sediment pulse duration.

Rainbow trout growth

The biological and ecological consequences of suspended sediment on fish have been the focus of many studies. Our results clearly indicate that a negative relationship exists between fish growth was and sediment pulse duration, despite a fixed concentration. The observed level of impairment was somewhat consistent with the predictions of Newcombe and Jensen (1996), who have developed a series of six model equations to predict the severity of ill effect (SEV) caused by fishes exposed to sediments in lotic ecosystems. The six models were developed for a range of fish species, life stages, and sediment particle sizes. Based on the concentration and duration of exposure to sediment, the model output is a series of expected impairment. Using the SEV model equation developed for juvenile and adult freshwater salmonids exposed to fine or coarse particles, trout in the four highest exposure doses of the present experiment would have been expected to show indications of major physiological stress; long-term reduction in feeding rate; long-term reduction in prey capture success; and poor condition. Trout in our experiment clearly showed reduced growth rates compared to control, suggesting similar findings to the model predictions. A lack of studies that describe early life stages exposed to sediments is a key shortcoming that will likely be remedied in future versions of this widely applied and useful fisheries management tool. The SEV model effectively predicted a reduction in feeding rate and prey capture success for trout in this experiment, but does not give any indication of the underlying mechanism.

The dose-dependent reduction in rainbow trout growth observed in this experiment can be explained by a number of mechanisms. First, suspended particles may come in direct contact with fish gills, causing abrasion, leading to decreased overall fitness and growth rates. Exposed gill tissue would also provide an entry for toxic chemicals, either bound to sediments or in solution, into the bloodstream and further impair growth. There is little support for this mechanism of action in our experiment, as visual inspection of each test fish revealed no gill tissue abrasion. Similarly, other studies that have exposed fish to an equivalent or higher dose of sediment reported no gill damage (Goldes et al. 1988), including highly angular, anthropogenically derived sediments (Lake and Hinch 1999).

A second mechanism that may explain a reduction in trout growth is that suspended sediments may directly alter behaviour or stress fish in a way that reduces foraging activity and (or) prey capture success. Gregory and Northcote (1992) reported a log-linear decline in reaction distance of chinook salmon to prey as turbidity increased from control to 810 NTU. Similar results have been established for bluegill (*Lepomis macrochirus*, Vinyard and O'Brien 1976), rainbow trout (Barrett et al. 1992), and lake trout (*Salvelinus namaycush*, Vogel and Beauchamp 1999). At moderate turbidity levels suspended material may also incur a perceived reduction in predation risk, allowing fish to increase foraging rates (Gregory 1994). During this experiment, trout were observed feeding on the drift in all channels prior to the addition of sediment. After the addition of sediment, trout in channels subject to a high sediment dose were noticeably more reactive, moving between cover objects and feeding areas more frequently and were exposed to potential predators for a longer duration than control fish. This observation suggests that foraging was not as successful or efficient for trout exposed to sediment pulses, providing support for this mechanism.

A third potential mechanism for growth reduction is that the invertebrate food resource was depleted or the composition altered as a result of sediment introduction, resulting in reduced growth rates. Many studies have also found that the density of benthic invertebrates is greatly reduced (Culp et al. 1986), and alterations to species composition (Culp et al. 1986; Newcombe and MacDonald 1991; Vuori and Joensuu 1996) occur following sediment events. Our study also indicated significant changes to the drifting insect assemblage composition.

Simultaneously exposing invertebrates and trout to sediment within a single experimental system allowed us to evaluate the two primary mechanisms causing the sediment impact. Path analysis suggested that the direct effect of sediment, acting to increase physiological stress and impair the vision of drift-feeding trout, is more important to trout growth than the indirect alteration to invertebrate abundance, which increased in our experiment. The increase in drift abundance was most strongly influenced by the family Chironomidae, which often make up a large portion of salmonid diet in both lotic and lentic environments (Pinder 1986) along with Trichoptera (caddisflies) and Ephemeroptera (mayflies) (Angradi and Griffith 1989). Unfortunately, our study did not distinguish between the abundance of drifting insects during the day versus night. As visual feeders, trout consume most of their prey during daylight hours. Often as a response to poor conditions, insects will drift to new habitats at night to avoid predation. A predominance of night drift caused by increased sediment pulse duration in our experiment would have suggested that the abundance of drifting invertebrate prey decreased, potentially altering the conclusions of our path analysis. Time may have also limited the response of drifting invertebrates. The final day of the experiment showed a significant decrease in the abundance of drifting invertebrates in the high sediment duration treatments, in comparison with the abundance of the same group following day 9. Allowing the experiment to proceed further might have resulted in high exposure treatments to arrive at a lower equilibrium abundance, also potentially altering the conclusions of our path analysis. Although the results of the path analysis seemed straightforward (i.e., the direct path coefficient was eight times higher than the indirect path) the results should be regarded as hypotheses to direct further testing (Wooton 1994).

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Operations on Saturated Soils

Or:

"To ensure that operations occur on a stable operating surface that does not produce sediment in quantities that violate Water Quality Requirements."

The Regional Water Quality Control Boards' Mission is to preserve and enhance the quality of California's water resources and ensure their proper allocation and efficient use for the benefit of present and future generations.

Background

- State and Regional Water Boards
- Implement State and Federal Water Law
- Basin Plan
- Non-Point Source Policy
- TMDLs
- WDRs/Waivers

Beneficial Uses of Water

HU/HA/ HSA	HYDROLOGIC UNIT/AREA/ SUBUNIT/DRAINAGE FEATURE	BENEFICIAL USES																										
		MUN	AGR	IND	PRO	GWR	FRSH	NAV	POW	REC1	REC2	COMM	WARM	COLD	ASBS	SAL	WILD	RARE	MAR	MIGR	SPWN	SHELL	EST	AQUA	CUL	FLD	WET	WQE
111.70	Middle Fork Eel River Hydrologic Area																											
111.71	Eden Valley Hydrologic Subarea	E	E	E	P		E	E	P	E	E	E	E	E			E	E		E	E			E				
111.72	Round Valley Hydrologic Subarea	E	E	E	P	E	E	E	P	E	E	E	P	E			E	E		E	E			E				
111.73	Black Butte River Hydrologic Subarea	E	E	E	P		E	E	E	E	E	E	E	E			E	E		E	E			P				
111.74	Wilderness Hydrologic Subarea	E	E	E	P		E	E	E	E	E	E	E	E			E	E		E	E			P				
112.00	**Cape Mendocino Hydrologic Unit**																											
112.10	Oil Creek Hydrologic Area	P	E	E	P		E		P	E	E	E		E			E	E		E	E		E	E	E			
112.20	Capetown Hydrologic Area	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E			P	E			
112.30	Mattole River Hydrologic Area	E	E	E	P	E	E	E	P	E	E	E	P	E			E	E		E	E		E	E				
113.00	**Mendocino Coast Hydrologic Unit**																											
113.10	Rockport Hydrologic Area	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.11	Usal Creek Hydrologic Subarea	E	P	P	P	E	E	E	P	E	E	E		E			E	E		E	E							
113.12	Wages Creek Hydrologic Subarea	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E							
113.13	Ten Mile River Hydrologic Subarea	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.20	Noyo River Hydrologic Area	E	E	E	P	E	E	E	E	E	E	E		E			E	E		E	E		E	E				
113.30	Big River Hydrologic Area	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.40	Albion River Hydrologic Area	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.50	Navarro River Hydrologic Area	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.60	Pt Arena Hydrologic Area																											
113.61	Greenwood Creek Hydrologic Subarea	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.62	Elk Creek Hydrologic Subarea	P	P	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.63	Alder Creek Hydrologic Subarea	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				
113.64	Brush Creek Hydrologic Subarea	E	E	E	P	E	E	E	P	E	E	E		E			E	E		E	E		E	P				

Timber-Related Sections of the Basin Plan for the North Coast

- Antidegradation policy
- Non-Point Source Policy
- TMDL Implementation Plans
- Sediment TMDL Implementation
- Action Plan for Logging, Construction, and Associated Activities

Antidegradation Policy

"Existing instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected."

Non-Point Source Policy

To "address all discharges of waste that could affect the quality of the waters of the State, including potential nonpoint sources of pollution."

TMDL Implementation Plans

- Garcia, Scott, and Shasta Rivers
- Each plan has Timber-specific section

Sediment TMDL Implementation

To control sediment waste discharges to impaired water bodies so that the TMDLs are met, sediment water quality objectives are attained, and beneficial uses are no longer adversely affected by sediment.

Action Plan for Logging, Construction, and Associated Activities

- **Prohibition against discharge** of soil, silt, bark, slash, sawdust, or other organic and earthen material from any logging, construction, or associated activity of whatever nature into any stream or watercourse.
- **Prohibition against placing or disposal** of soil, silt, bark, slash, sawdust, or other organic and earthen material from any logging, construction, or associated activity of whatever nature at locations where such material could pass into any stream or watercourse.
- **Turbidity shall not be increased** more than 20 percent above naturally occurring background levels.

“Significant” Discharge

- **All** unpermitted discharges are significant.
- A variety of activities result in some level of discharge.
- Waste Discharge Requirements (WDRs) are the mechanism to regulate waste discharges.
  - General WDR regulate discharges from Timber Activities.
  - Categorical Waiver where discharge can be reduced to below significant levels.

"Increase in Visible Turbidity"

An "increase in visible turbidity":

- Greatly exceeds the 20% threshold.
  - 20% threshold is biologically based, not observationally based
  - Visible increase is generally 100% to 500% or more
- Represents a violation of the Basin Plan.

Need for Enforceable Language in the Forest Practice Rules

- The current language, “may produce sediment,” is unenforceable.
- “quantities sufficient to cause a visible increase in turbidity of downstream waters” implies that saturated soil conditions exist only after a visible increase in turbidity is observed.

Proposed Road Use Language in the Forest Practice Rules

- Support substitute language to remove "~~quantities that may cause a visible increase in turbidity of downstream waters in receiving Class I, II, III or IV waters~~"
- Support retaining "quantities that violate Water Quality Requirements" language
- Support DFG Option 932.5(p)(4) and (5)
- Support CGS Option 923.5(ii)

Sediment affects
Multiple Beneficial Uses

North Coast Region
Contact Information
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Copy of the North Coast Basin Plan available at:
http://www.waterboards.ca.gov/northcoast/water_issues/programs/basin_plan/083105-bp/070605_Basin_Plan.pdf

Summary of Selected Information on Wet Weather Log Hauling and Impacts to Water Quality

Pete Cafferata
CAL FIRE
March 1, 2011

Outline

- ODF Wet Season Road Use Monitoring Study
- JDSF Hare Creek Wet Weather Hauling Study
- Sullivan PALCO Crossing Study
- HMP and MCR Monitoring Results
- Published Papers Summary
- Take Home Messages

FOREST PRACTICES MONITORING PROGRAM
TECHNICAL REPORT # 17

Oregon Department of Forestry
Wet Season Road Use Monitoring Project

June 2003

Keith Mills, Liz Dent, and Josh Robben

Study Goal

Study designed to identify factors that contribute to turbidity when roads are used during wet periods.

Road Surface and Hauling Impacts

Study Objectives

How do these factors affect turbidity?

- **Rainfall conditions that result in road runoff entering streams**

- **Road surface aggregate**

- **Road segment length draining to streams**

Study Sites

- Private industrial and state-managed forestlands in Western Oregon.
- Low elevation sites—not snow covered.
- Sites were located in both the OR Coast Range and the Cascades.
- Only roads with active winter timber hauling.
- All roads were gravel surfaced.
- Roads included primary haul roads and secondary roads accessing logging units.
- Roads cross small streams that flow all winter.
- 174 stream crossings were studied; 438 crossing pairs (upstream and downstream were analyzed for change in turbidity.

Study Design

- **Not random sampling design.**
- **Samples collected above and below crossings during periods of heavy rainfall.**
- **Results cannot be used to characterize average water quality conditions resulting from wet weather hauling.**
- **Study provides data on factors affecting stream turbidity during wet season road use.**

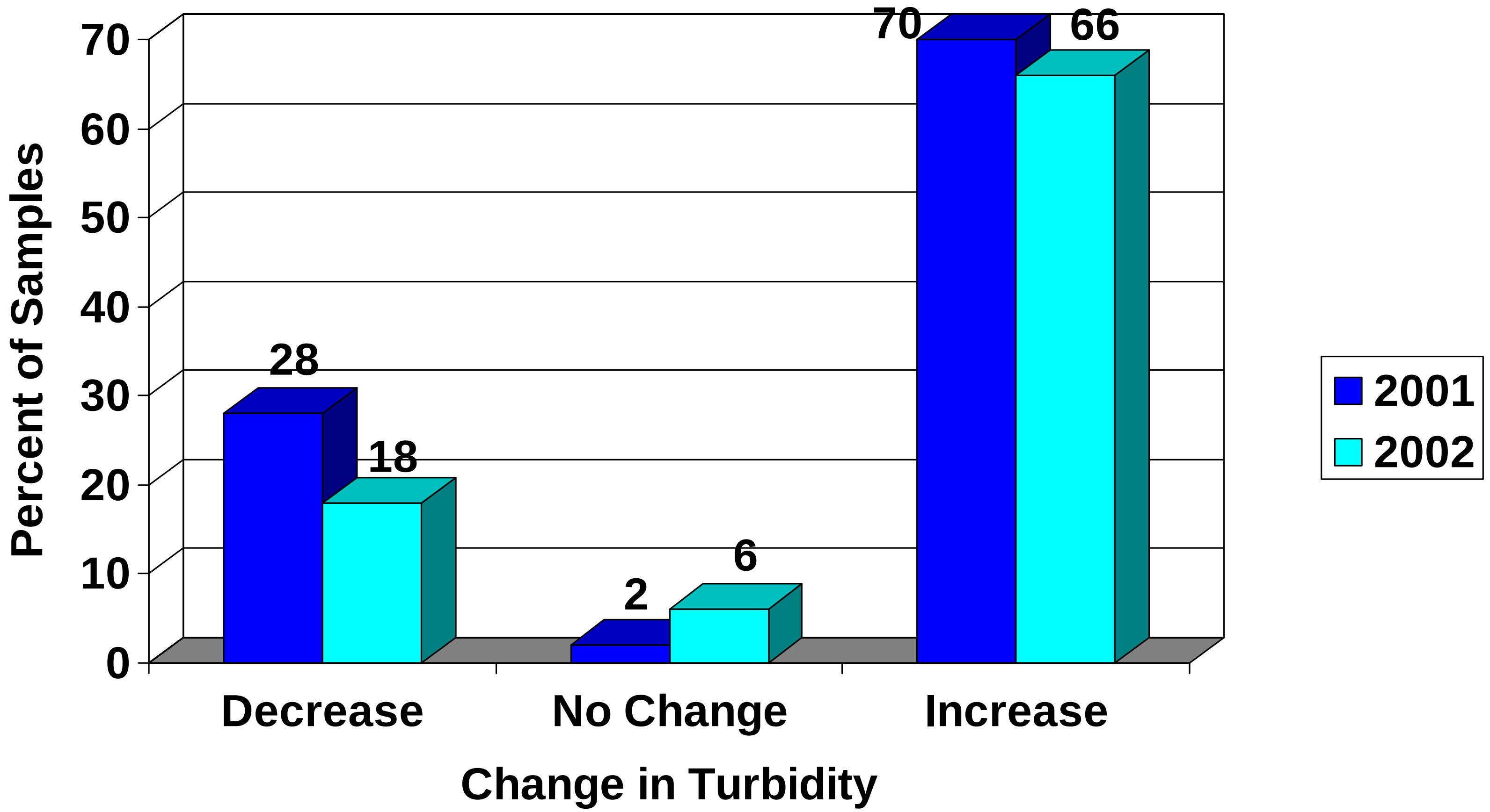
Road Segment Parameters

- **Average road gradient**
- **Ditch length draining to stream**
- **Depth of road surfacing material**
- **Sample of surfacing material**

Other Data Recorded

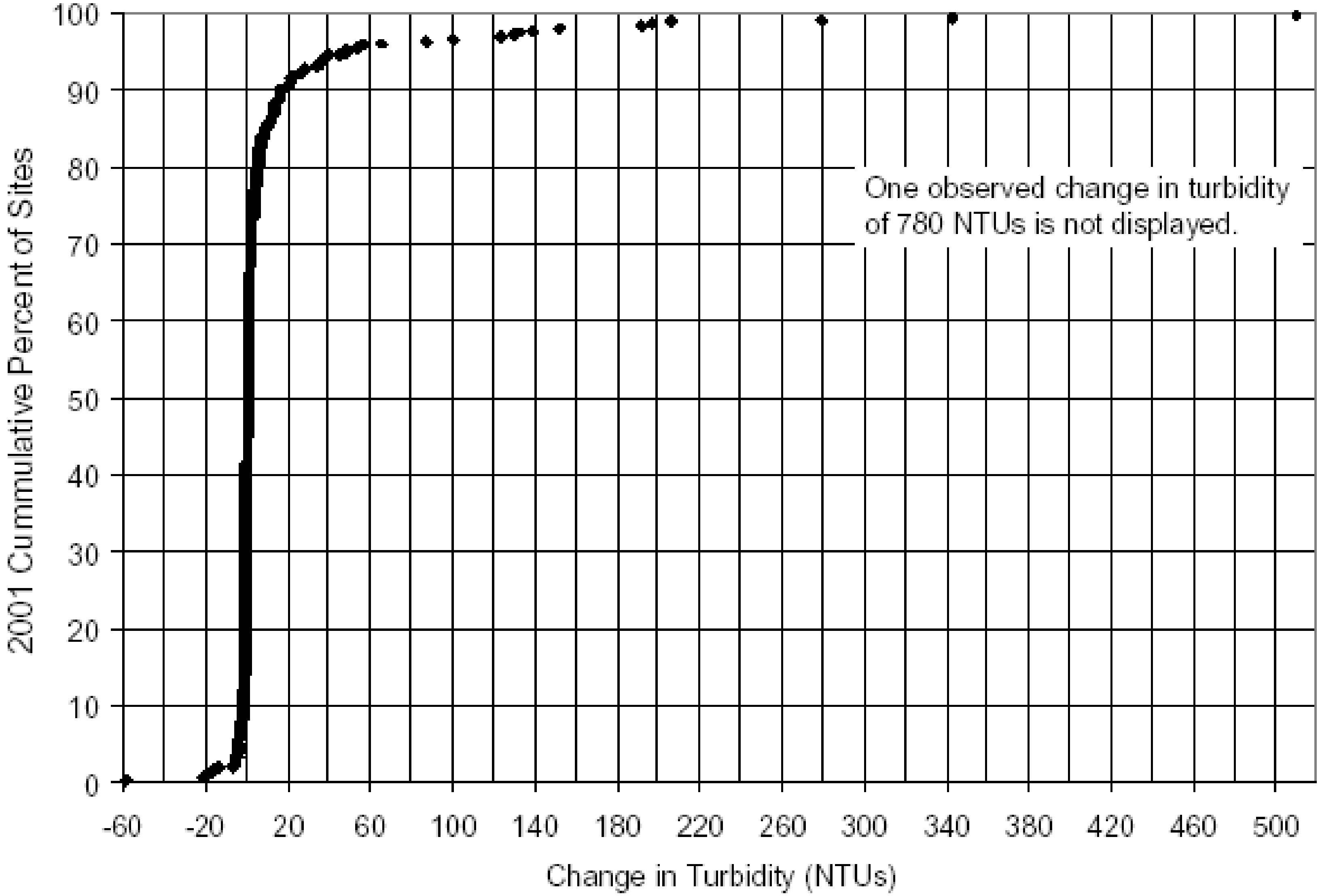
- Log truck traffic level
- Condition of road surface (rutting, mud depth)
- Recent maintenance activity
- Sources of sediment delivery to channel other than road surfacing
- Precipitation from nearest station
- Amount of ditchflow—estimated as % of streamflow

Crossing Turbidity Samples



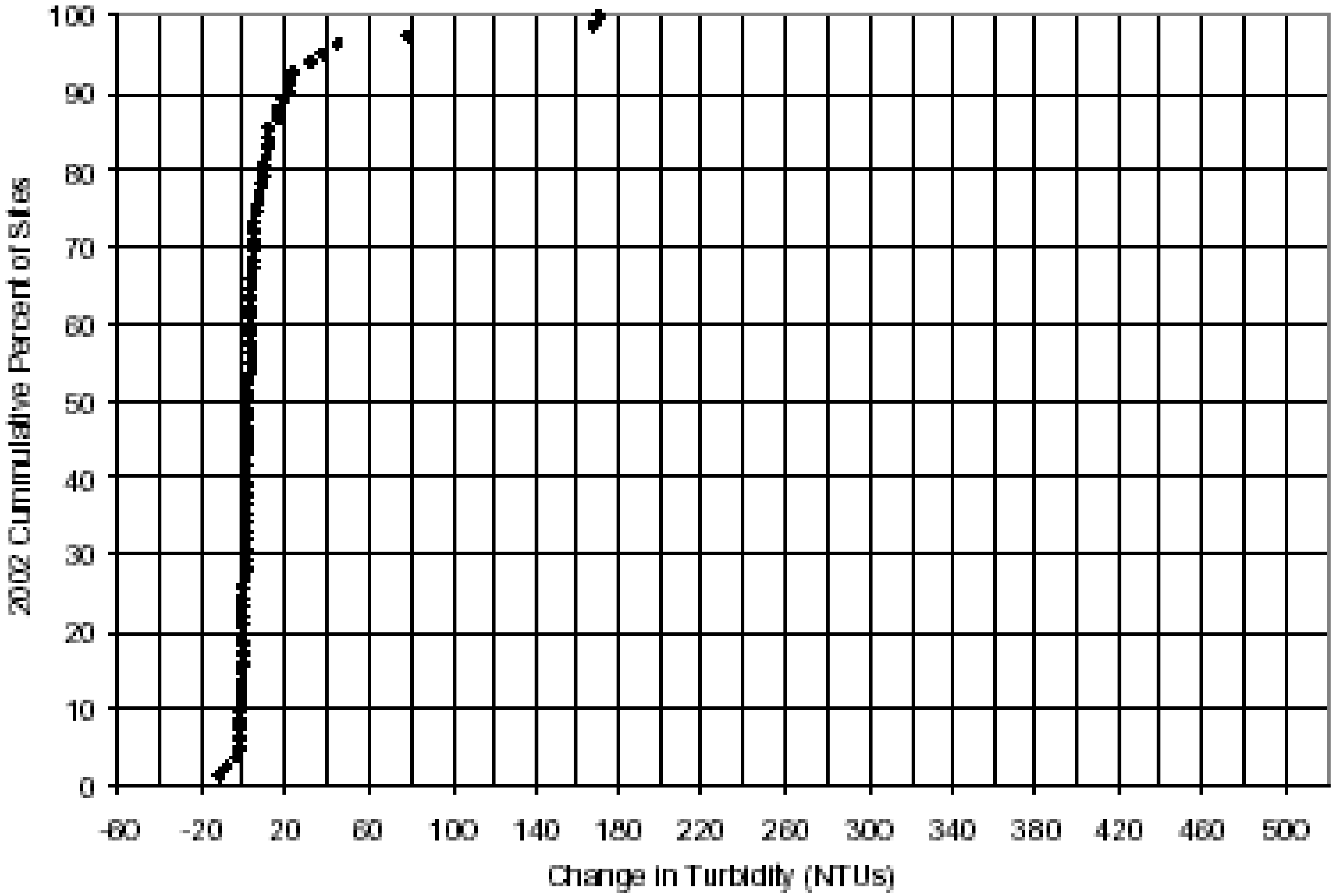
Road Surface Water Entering a Stream

Distribution of Turbidity Changes for 2001 Crossing Samples



A change of 20 NTUs or less was observed for ~90% of the sample pairs.

Distribution of Turbidity Changes for **2002** Crossing Samples



A change of 20 NTUs or less was observed for ~90% of the sample pairs.

Summary of Turbidity Changes

- **89-90% (2002 and 2001 seasons respectively) of the sample pairs showed a change of 20 NTUs or less.**

- **The remaining 11 to 10% ranged from an increase of 20 to 780 NTUs.**

Possible Reasons for Decreased Turbidity Below Crossings

- Changes <10 NTUs cannot be distinguished from measurement error.

- Changes >10 NTUs could be due to:
  - Settling of materials between sampling points
  - Poor mixing of suspended materials

Factors with Potential to Influence <u>Changes in Turbidity</u>

- Precipitation (3 day total)
- Depth of surfacing material
- Percent fines in surfacing material
- Durability of surfacing material
- Length of road ditch draining directly to the stream
- Traffic levels

2001 Multiple Linear Regression Model <u>for Observed Increases in Turbidity</u>

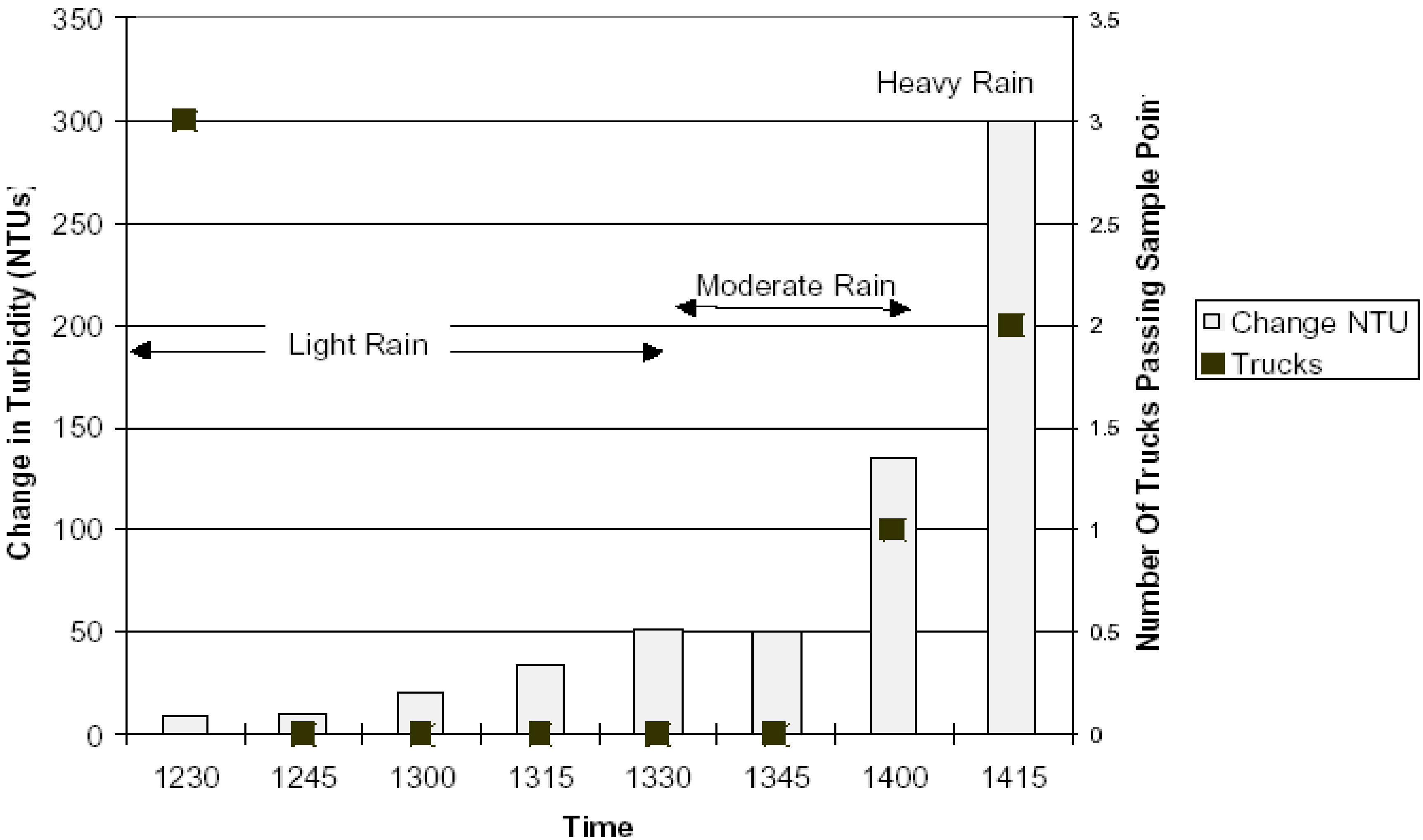
Log NTU = -3.474 + 5.9 (3-day precipitation) + 0.494 (ditch length)

Model is highly significant (p <0.001);

r^2 =0.66

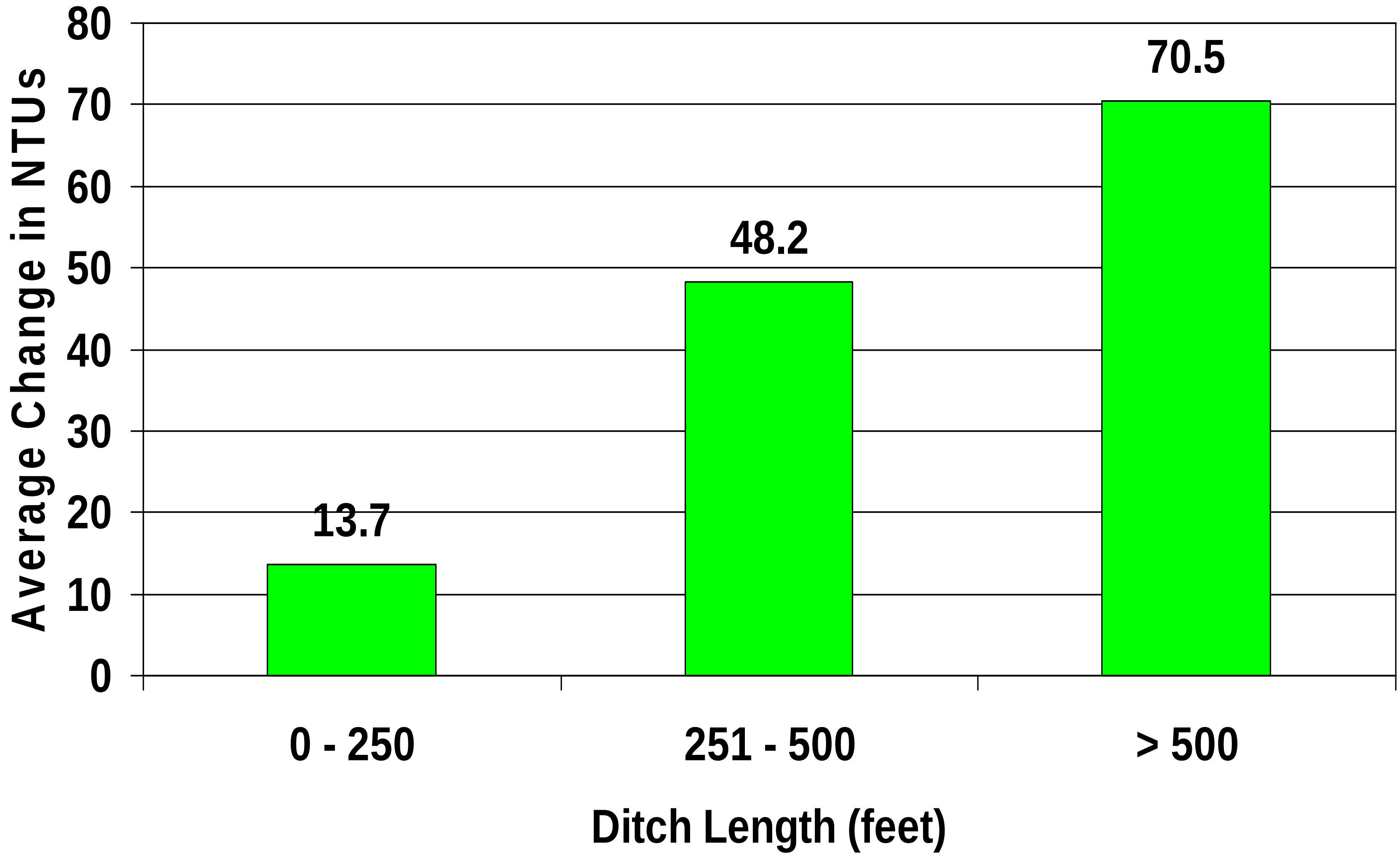
2002 model had a low r^2 value but indicated that percent fines (<0.075 mm or pass #200 sieve) in surface aggregate affected turbidity.

Changes in Turbidity Over Time at One Crossing Every 15 Minutes Over 2 Hours

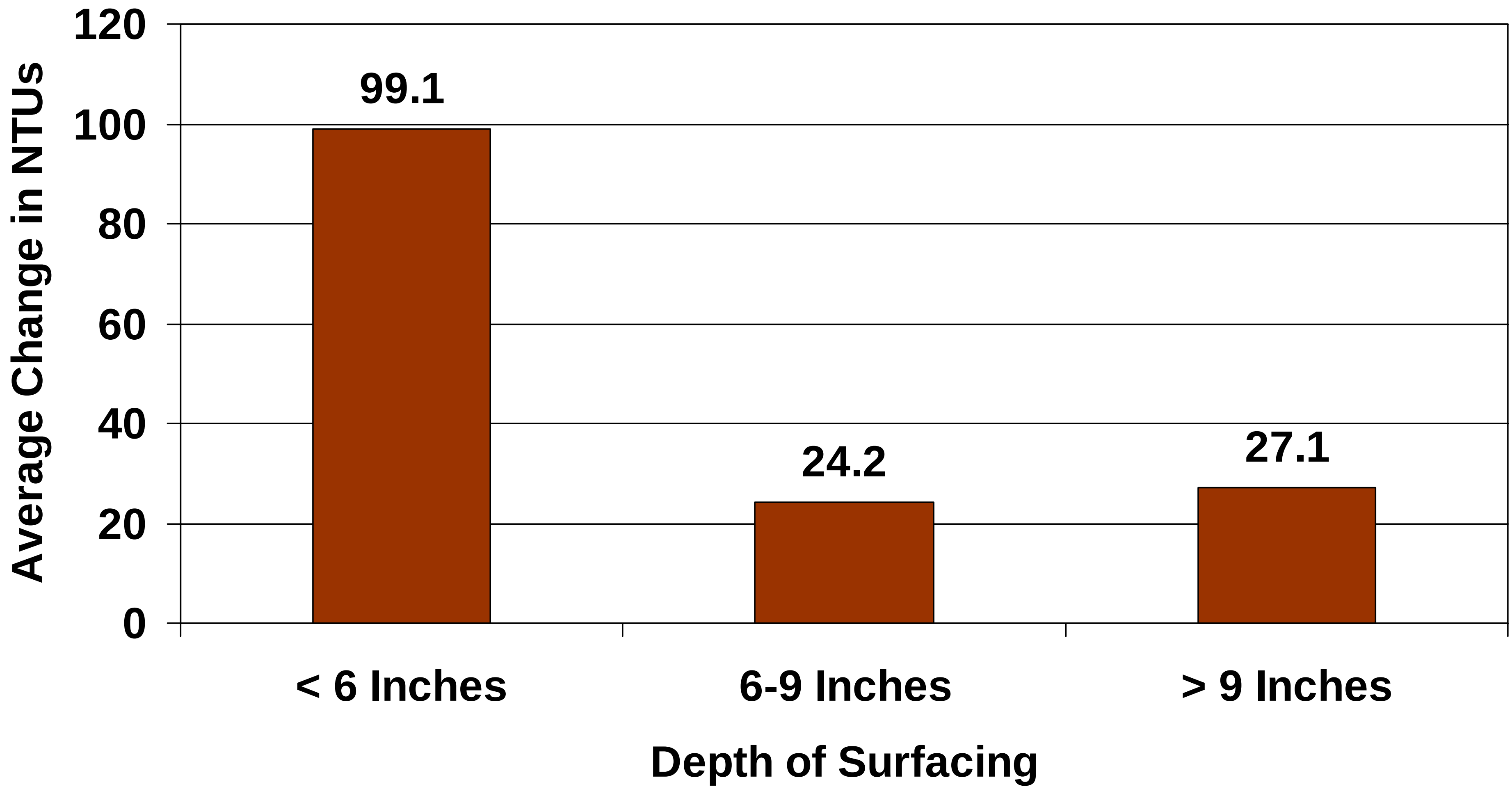


As truck traffic and rain increases, so does turbidity.

Average Change in Turbidity vs. Ditch Length Draining into Streams at Crossings



Average Change in Turbidity vs. Depth of Road Surfacing



Adequately Surfaced Road that Received 3 inches of Rain in 1 Day

Shows that a durable surface can limit or prevent turbidity entry into a watercourse.

Photo: K. Mills, ODF

Summary

- **Turbidity increases were associated with longer drainage ditches, fines in the aggregate, heavy truck traffic, and shallower rock surfacing.**
- **At crossings, 90% of sample pairs showed a change of 20 NTUs or less.**
- **The remaining 10% of the observations ranged from an increase in turbidity of 20 to 780 NTUs.**

Six Factors Identified as Most Important for Turbidity Increases

- **3 day precipitation between 1.5 – 3.0 in.**
- **Size distribution of road surfacing material**
- **Over 250 feet of ditchline draining to channel**
- **Depth of surfacing material <6 inches**
- **Durability of surfacing material of less than a 17 Los Angeles abrasion rating**
- **Traffic levels of 10 or more trucks per day.**

Recommendations

- **Use aggregate containing the minimum percentage of fines needed to bind, pack and seal the surfacing.**
- **Use at least 6 to 10 inches of sound aggregate (igneous or metamorphic rock).**
- **Reduce length of segments that deliver sediment to less than 250 feet by adding cross drains or other structures.**
- **Prioritize inspection of active winter operations during first moderate rainfalls to determine if immediate repairs are needed or ceasing road use is necessary.**

Results were used to develop new rules for wet season road use

ODF FPR 629-625-0700
Wet Weather Road Use

- Operators shall use durable surfacing or other effective measures that resist deep rutting or development of a layer of mud on top of the road surface on road segments that drain directly to streams on active roads that will be used for log hauling during wet periods.

- Operators shall cease active road use where the surface is deeply rutted or covered by a layer of mud and where runoff from that road segment is causing a visible increase in the turbidity of Type F or Type D streams as measured above and below the effects of the road.

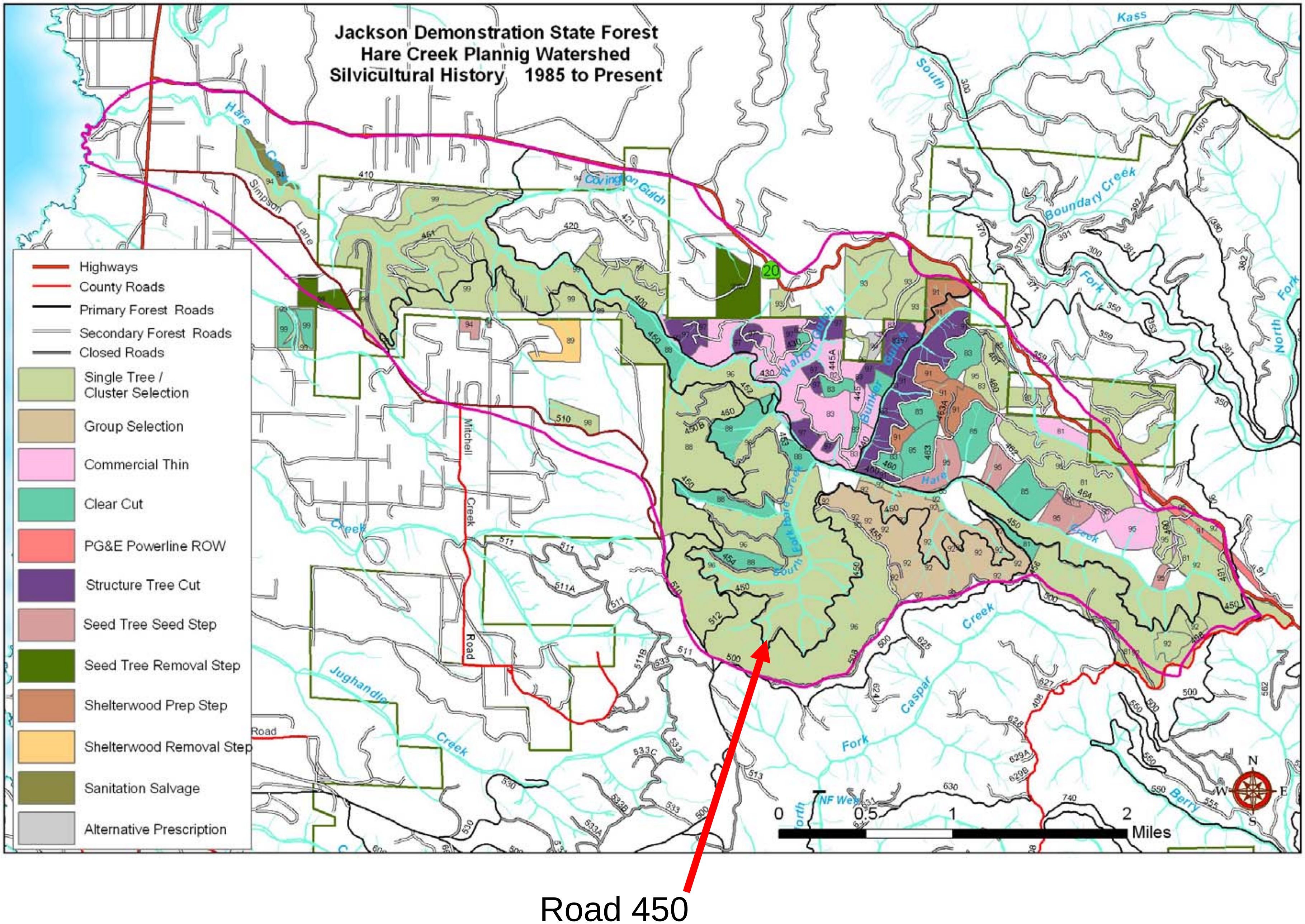
Turbid Water From Hauling Entering a Fish Stream


Hare Creek
Road Runoff
Turbidity Study

Hare Creek
Road Runoff Turbidity Study

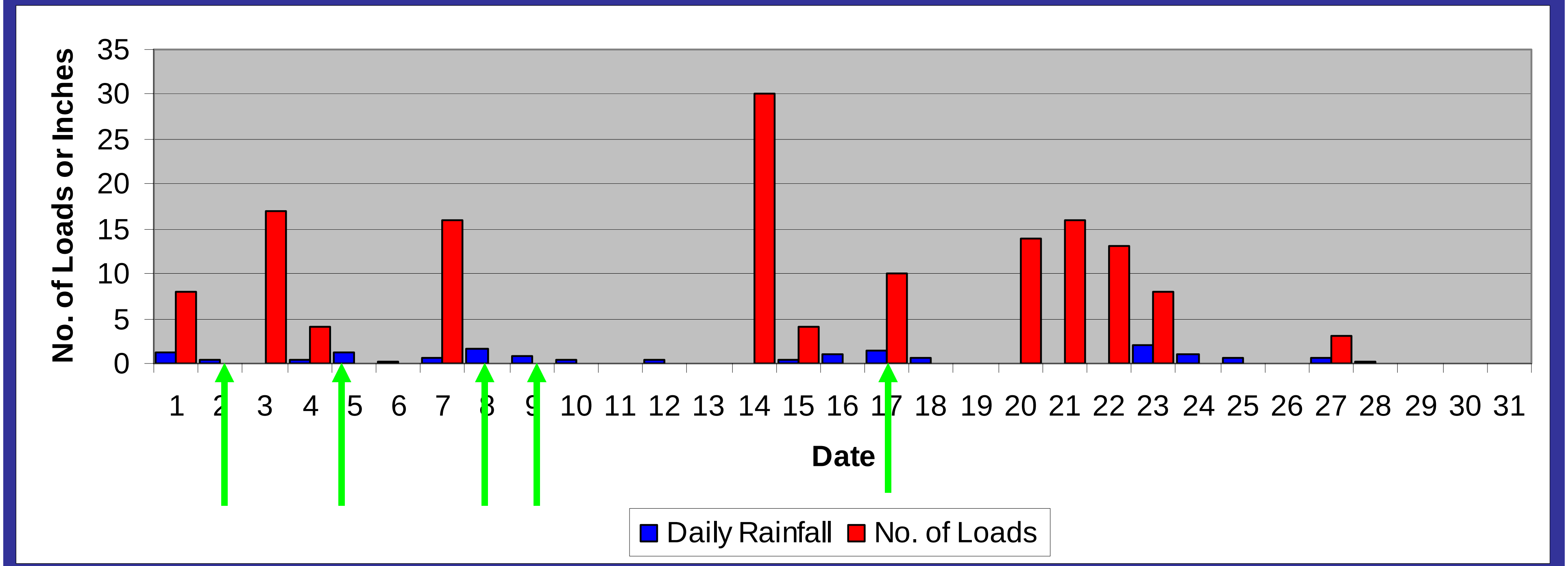
- Serious road runoff problems were noted as part of the Hare Creek 1988 Timber Sale.
- The rock which was applied to Road 450 failed to adequately surface the road.
- In many locations, the road bed was soft and rutted.
- Sampling stations were set up at locations where small streams crossed Road 450 through culverts (one station was a control out of the sale area).
- Storm events were sampled 5 times in March 1989, both above and below the stream crossings.

Hare Creek Planning Watershed



Hare Creek Road Runoff Turbidity Study

Hauling and Rainfall Data for March 1989

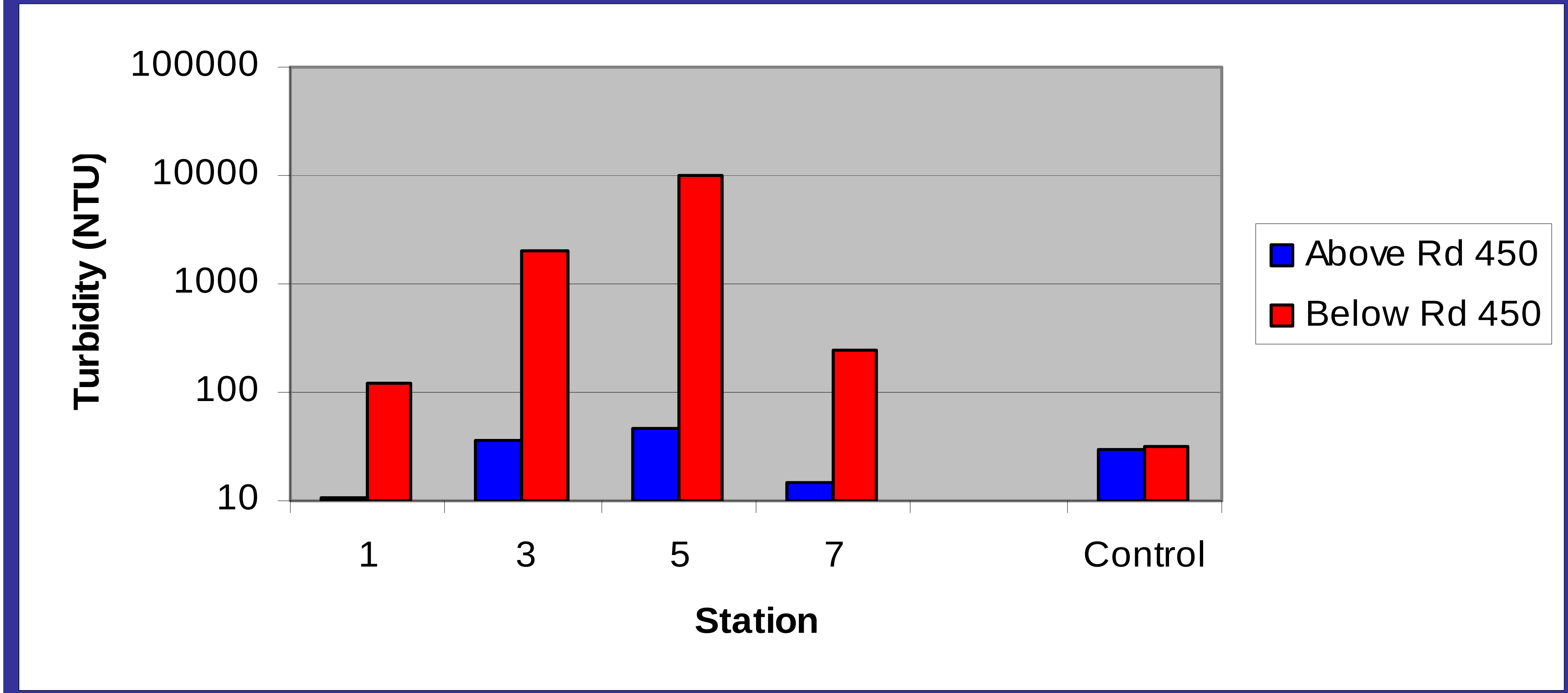


Green arrows indicate sampling periods

Data from
Cafferata 1989

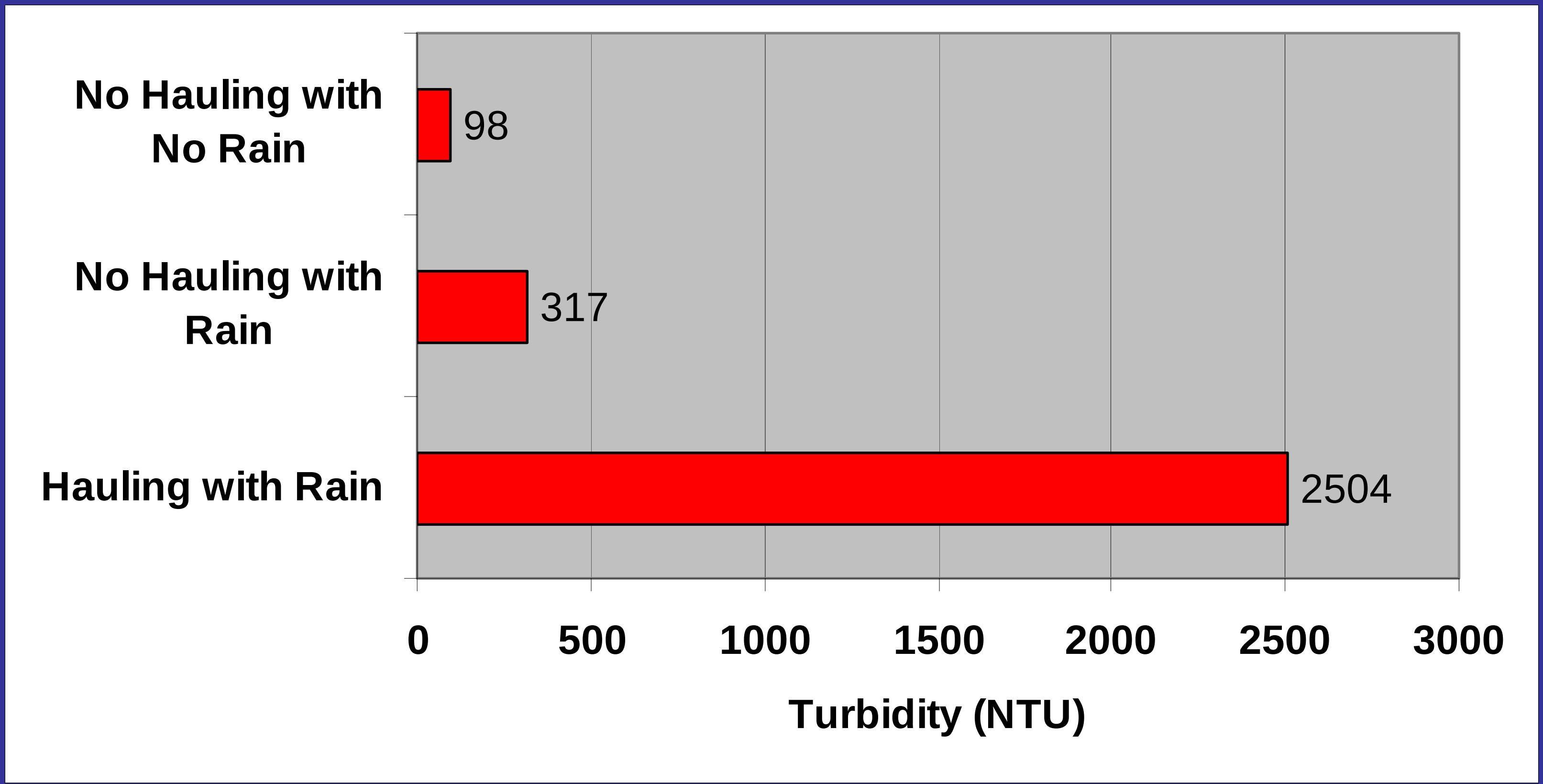
Hare Creek Road Runoff Turbidity Study

Sale area stations had an average turbidity of 115 NTUs above the road and 678 NTUs below the road



4 of the 5 stations had much higher turbidity <u>below crossings</u>

Hare Creek Road Runoff Turbidity Study



Higher turbidity below watercourse crossings demonstrated that log hauling and road practices needed modification. The results showed that by limiting log hauling during and for a short time following rainfall, damage to water quality can be substantially reduced.

Hare Creek Road Runoff Turbidity Study

- **Changes made (see JDSF Road Management Plan):**
  - **No log hauling will occur if greater than 0.25 inch of precipitation has fallen during the preceding 24 hour period.**
  - **Hauling can resume only after rain has ceased for 24 hours and no road-related turbid water is observed in inside ditches along the roads where hauling may occur.**
  - **Log hauling will not occur when “pumping” of fines from the road surface produces sediment that enters inside ditches and causes turbid water to flow in ditchlines with direct access to watercourses.**
  - **Only surfaced roads will be considered for wet weather log truck traffic. If road rock begins to significantly break down, wet weather use of that road will cease until the road is adequately repaired.**

Water quality grab sampling to find sediment sources during winter forest operations

Presentation by Dr. Kate Sullivan
PALCO (currently HRC)
Monitoring Study Group Meeting
Oct 16, 2003

Sediment Sources

- Intent of this sampling was to identify sediment sources
- Observe during storms
  - Approximately 1” in 24 hours
- Look at active THP units and road crossings

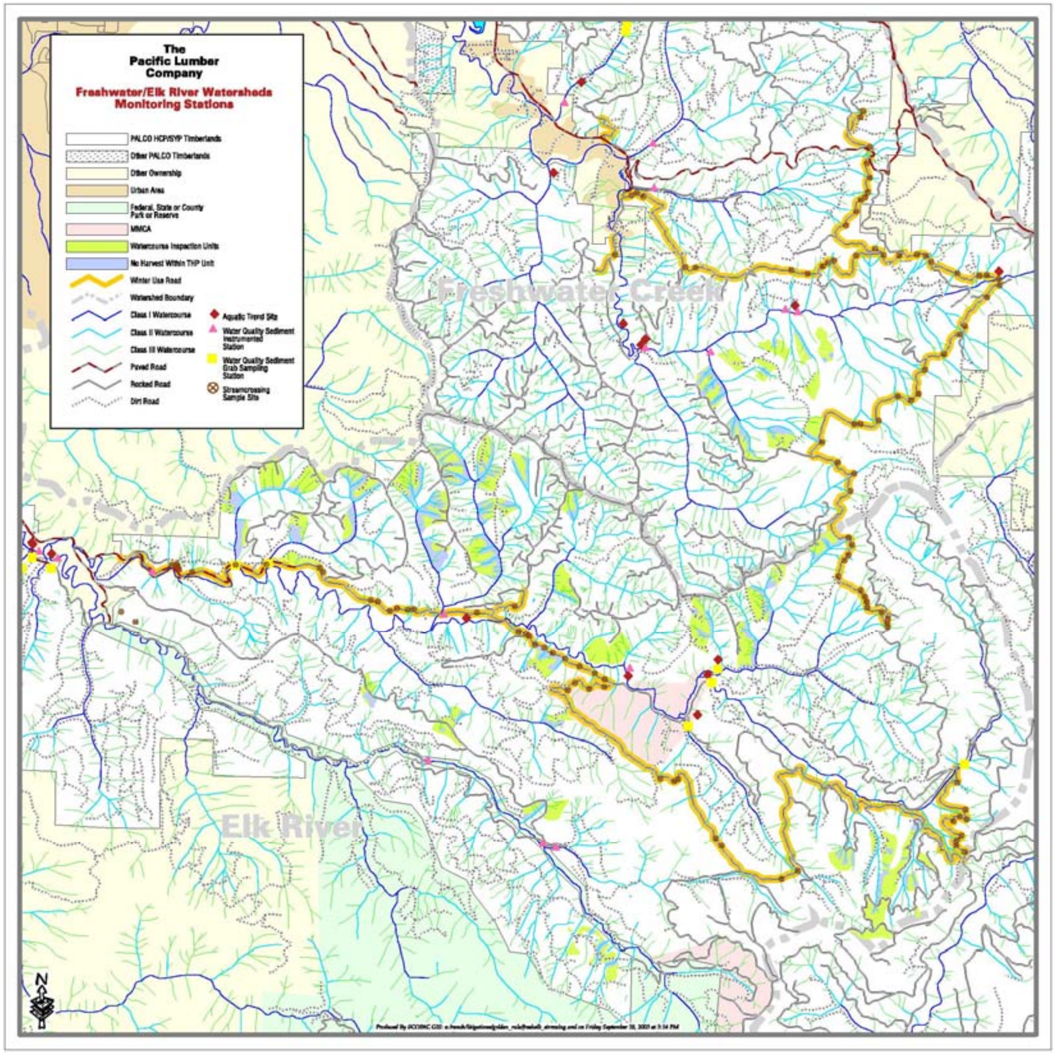
Effectiveness

Monitoring Road Runoff

Above and Below Grab Sampling-compare downstream to upstream

Image + photos: Dr. Kate Sullivan, HRC

Water Course Crossing Samples Locations



Freshwater Creek

- 57 crossings observed

Elk River

- 64 crossings observed

Over 400 samples collected

Image: Dr. Kate Sullivan, HRC

PALCO Above and Below Grab Sampling

- Most stations were sampled 10 times during the winter of 2002-2003.

- A threshold of greater than 20% above background was used to identify a significant difference for this work (i.e., downstream turbidity values more than 20% greater than upstream levels).

- It was common to have greatly elevated downstream turbidity for one sample period, while the other samples were approximately the same above and below the crossing.

Summary of Road Crossing Sampling Results—All samples Combined:

17% of 400+ samples collected at 121 crossings over the 2002-03 winter had downstream turbidity levels 20% greater than upstream levels

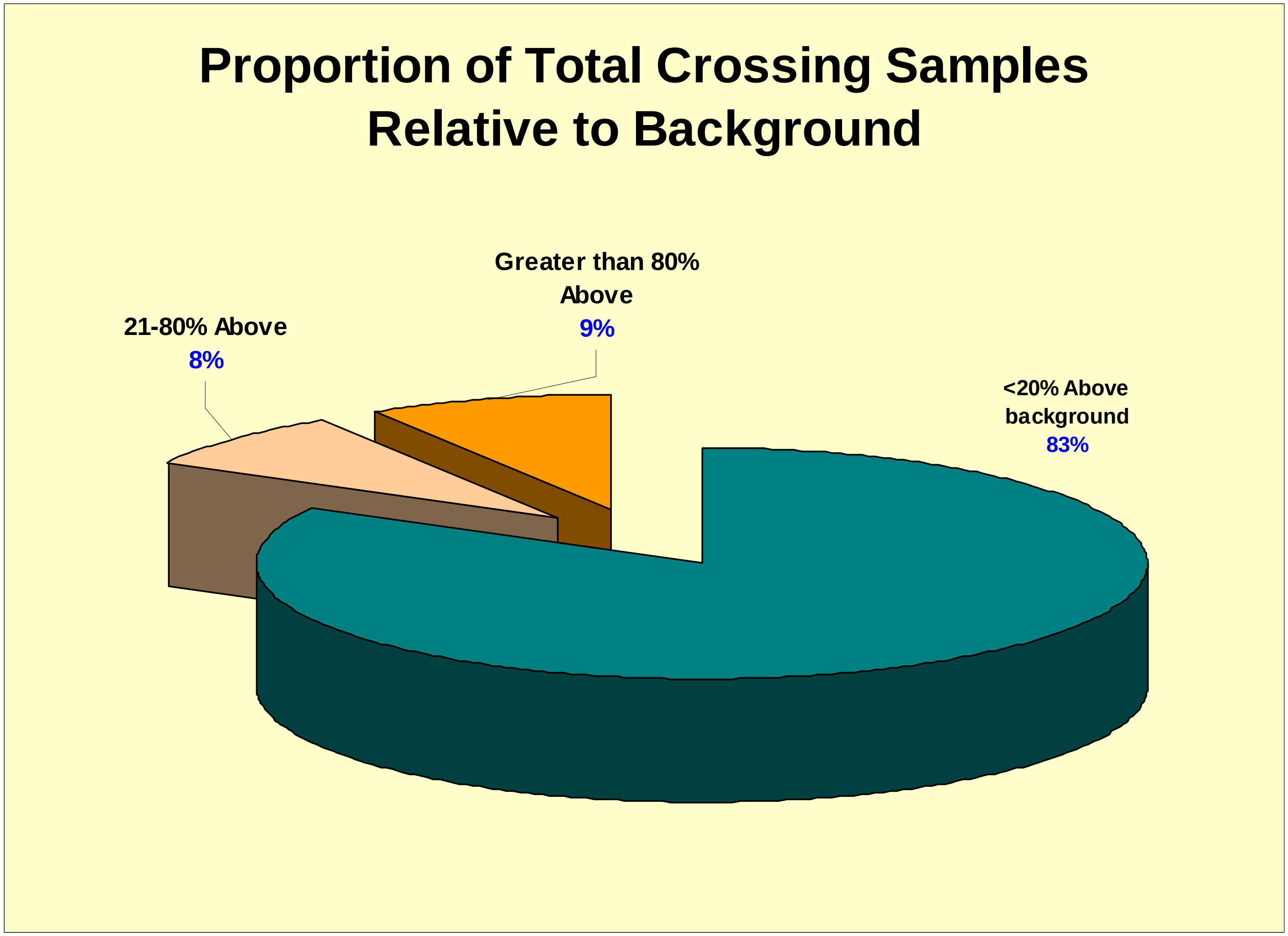


Image: Dr. Kate Sullivan, HRC

Roads Summary

- Many road crossings were always within 20%.
- Some were chronically high.
- Some occasionally exceeded.
- Some were repaired during the season and never showed up again.

PALCO Road Crossing Turbidity Data Published—Harris et al. 2007

- Elk River and Freshwater Creek watersheds; about **7%** of 2300 samples collected at 225 crossings over a 3-year period had downstream turbidity levels more than 20% greater than upstream levels.

- Potential impacts depend on erosion control measures implemented at the sites.

MONITORING STUDY GROUP
CALIFORNIA STATE BOARD OF FORESTRY AND FIRE PROTECTION

HILLSLOPE MONITORING PROGRAM

MONITORING RESULTS FROM 1996 THROUGH 2001

Andrea E. Tuttle
Director
Department of Forestry and Fire Protection

Mary D. Nichols
Secretary for Resources
The Resources Agency

Gray Davis
Governor
State of California

DECEMBER 2002
SACRAMENTO, CALIFORNIA
BOARD OF FORESTRY AND FIRE PROTECTION

Findings Related to ROAD APPROACHES

- The road surface cutoff drainage structure above the crossing allowed all or some of the water running down the road to reach the crossing at about 23 percent of the sample sites (~8% allowed all the water to drain to the crossing).

- Approximately 2-3% of road surfaces draining to crossings had significant rutting, rilling, and gullying.

MONITORING STUDY GROUP
CALIFORNIA STATE BOARD OF FORESTRY AND FIRE PROTECTION

Modified Completion Report
MONITORING PROGRAM

Implementation and Effectiveness of
Forest Practice Rules related to Water Quality Protection

MONITORING RESULTS FROM 2001 THROUGH 2004

Ruben Grijalva
Director
Department of Forestry and Fire Protection

Mike Chrisman
Secretary for Resources
The Resources Agency

Arnold Schwarzenegger
Governor
State of California

July 2006
SACRAMENTO, CALIFORNIA

Findings Related to ROAD APPROACHES

- The road surface cutoff drainage structure above the crossing had minor or major problems ~25% of time.
- Major problem 4% of the time.

- Approximately 6% of road surfaces draining to crossings had major and minor gullying.
- Major problem ~0.5% of the time.

- 16.5% of the road surfaces draining to crossings had major and minor rutting.
- Major problem ~1% of the time.

Information from Selected Papers

- **Reid and Dunne 1984**
  - Log truck traffic on forest roads during winter storms increased the yield of fine sediment during these storms by up to several orders of magnitude in western Washington.

- **Bilby, Sullivan, and Duncan 1989**
  - The amount of sediment produced was related to traffic rate.
  - Accumulated material flushed rapidly from the road surface during precipitation, leading to a decrease in sediment concentration in the ditch with time during a storm.

- **Luce and Black 2001**
  - Ditch pulling has more impact than traffic on a road with quality aggregate (i.e., no ruts).

- **Toman and Skaugset 2007**
  - To minimize wet weather sediment production, design aggregate surfacing to resist rutting.
  - Rut formation is a function of aggregate depth.
  - Study completed on GDRCO timberlands.

Take Home Messages

- Log truck traffic on forest roads during winter storms can increase the yield of fine sediment by up to several orders.
- Turbidity increases are associated with longer drainage ditches, fines in the aggregate, heavy truck traffic, and shallow rock surfacing.
- With current practices, ≤10% of samples taken above and below crossings show much higher turbidity levels downstream.
- In California, 4-8% of road cut off drainage structures above crossing allow all or most of the water to reach the crossing. Minor and major problems occur on ~25%.
- One way to minimize wet weather sediment production is to design aggregate surfacing to resist rutting by having an adequate aggregate depth.

Effects of Snowmobile Emissions on the Chemistry of Snowmelt Runoff in Yellowstone National Park

Final Report

Jeffrey L. Arnold and Todd M. Koel
Fisheries and Aquatic Sciences Section
Center for Resources
Yellowstone National Park
Wyoming 82190

YCR-2006-1
2006

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Yellowstone National Park
Wyoming 82190

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Executive Summary

Created in 1872, Yellowstone National Park forms the core of the Greater Yellowstone Ecosystem, and is arguably the largest intact naturally functioning ecosystem remaining in the lower 48 United States. The park was created to protect the unique geothermal features and headwaters of the Madison, Snake, and Yellowstone rivers, while providing for the enjoyment of this unique environment by visitors. Approximately 44,000 hectares of lakes and 4,300 kilometers of streams exist in Yellowstone National Park, all which are classified as Outstanding Natural Resource Waters (Class I), meaning they must receive a very high level of protection against degradation.

More than three million people visit the park each year to engage in a wide range of recreational activities. Throughout the winter season (December–March), most park roads are closed to vehicular travel and are groomed and maintained for oversnow transportation. As a result, many visitors during winter months travel by snowmobiles. A significant increase in use of these machines was first documented in the late 1980s, when the numbers had increased nearly tenfold over that in 1968. By the mid-1990s the number of snowmobiles entering the park had increased to nearly 75,000 per year. During this time most snowmobiles had two-stroke engines, known to burn fuel inefficiently. Consequently, the steady increase in snowmobile use within the park was a concern to resource managers because of the potential that the increase in fossil-fuel combustion could result in greater levels of emissions entering the pristine surface waters of the park.

During late March through mid-April of 2003 and 2004 snowmelt runoff samples were collected from four sites along the heavily used road corridor between Yellowstone National Park's West Entrance at West Yellowstone, Montana, and the Old Faithful visitors area. Three of these sites were located immediately adjacent to the roadway in the vicinity of the West Entrance, Madison Junction, and Old Faithful. The remaining site was used as a control, located near Madison Junction approximately 100 meters from the roadway and away from the effects of snowmobiles. Each site was visited on 9–10 different days during the spring sampling period, with visits dependent on having a daily temperature >5°C and good potential to obtain snowmelt runoff. *In situ* water quality measurements (i.e., water temperature, dissolved oxygen, pH, specific conductance, and turbidity) were collected. Snowmelt runoff samples were analyzed for nine volatile organic compounds (VOCs), including benzene, ethylbenzene, ethyl tert-butyl ether, isopropyl ether, *meta-* and *para-*xylene (m- and p-xylene), methyl tert-butyl ether, *ortho-*xylene (o-xylene), tert-pentyl methyl ether, and toluene. Of these nine compounds, only five were detected during any one sampling event. The detected compounds included benzene, ethylbenzene, m- and p-xylene, o-xylene, and toluene.

All *in situ* water quality measurements were within acceptable limits. The VOCs were most prevalent at the Old Faithful site, which receives extremely high use by snowmobiles each year. Fortunately, the concentrations of all VOCs detected each year were considerably below the U.S. Environmental Protection Agency's (USEPA) water quality criteria and guidelines for VOCs targeted in this study. During the course of the study, VOC concentrations of snowmelt runoff in Yellowstone National Park were below levels that would adversely impact aquatic systems. However, future research in Yellowstone National Park on snowmobile emissions should address the potential for another group of harmful chemicals known as the polycyclic aromatic hydrocarbons (PAH). The PAH tend to be more capable of persisting in the environment for longer periods than VOCs and are suspected at the Old Faithful site as it received runoff from a paved parking area.

Introduction

Over three million people visit Yellowstone National Park each year to engage in a variety of recreational activities, many of which include viewing wildlife, camping, hiking, fishing, and geyser watching. During the winter season, most roads are closed to wheeled vehicular travel, and are groomed and maintained for over-snow travel. Consequently, winter visitors can tour the interior of the park by snowmobiling, traveling in snowcoaches, cross-country skiing, and snowshoeing. A growing trend in the number of snowmobiles entering Yellowstone National Park was first documented in the late 1980s, during which time snowmobile numbers had increased nearly tenfold since 1968 (National Park Service 1990). By the mid-1990s the number of snowmobiles entering the park had increased to nearly 75,000 per year (National Park Service 2004). The burgeoning numbers of snowmobiles that enter the park each year are of particular concern because increases in fossil-fuel combustion could contribute to greater levels of emissions entering sensitive watersheds and animal habitats (Ingersoll 1999).

The increase in the number of snowmobiles entering the park prompted the National Park Service (NPS) to develop a winter use plan in 1990. In 1994, NPS and the U.S. Forest Service began work on a coordinated interagency report on winter visitor use management. In addition, the Greater Yellowstone Coordinating Committee, composed of National Park superintendents and National Forest supervisors within the Greater Yellowstone Area, met to specifically address the increasing number of snowmobiles that enter the park throughout the winter season (National Park Service 2002). In May 1997, the Fund for Animals and other organizations filed a law suit against the National Park Service alleging that the NPS failed to conduct adequate National Environmental Policy Act analysis when developing its winter use plan (National Park Service 2002). Under a Settlement Agreement, the NPS agreed to prepare an environmental impact statement (EIS) which was published in October 2000. Subsequently, a Winter Use Supplemental EIS (SEIS) was prepared in March 2002, to address new or additional information and data as provided by the public, cooperating agencies, and information regarding new snowmobile technologies (National Park Service 2002). This Winter Use SEIS identifies information needs as it relates to winter use activities and its impact on important park values such as air quality, soundscape, wildlife, aquatic resources, geothermal features, and visitor experience (National Park Service 2002).

In March 2003, a Winter Use Record of Decision (ROD) was completed to address future winter use activities in Yellowstone National Park. Two major components of the ROD were a reduction in the number of snowmobiles through daily limits and the use of snowmachines requiring best available technology (BAT). The SEIS and ROD led to changes in winter policies between the 2003 and 2004 winter seasons. During the 2003 winter season both two-stroke and four-stroke snowmachines were allowed in Yellowstone National Park with no limit to the number of snowmachines. By contrast, during the 2004 winter season a daily limit was placed on the number of snowmobiles that entered the park along with the requirement that all snowmobiles be BAT machines. The number of snowmobiles that entered the park during the 2003 and 2004 winter season were 47,799 and 22,423 respectively.

Obtaining baseline information on surface water quality as it relates to snowmelt runoff was identified as a priority in the Winter Use SEIS (National Park Service 2002). Specifically, snowmobile emissions could affect the overall surface water quality by changing pH, hydrogen, ammonium, calcium, sulfate, and nitrate levels and could also contribute harmful levels of volatile organic compounds (VOC). VOCs are hydrocarbons that are associated with crude oil and other petroleum products. They are of concern in Yellowstone National Park because they are produced by incomplete combustion of gasoline from two-stroke snow machines and can accumulate in snowpack (Ingersoll 1999). They are of particular interest because of their close association with gasoline exhaust and the possible adverse effects on human health and aquatic systems. In high concentrations, people exposed to VOCs can experience general symptoms such as headaches, dizziness, nausea, and throat and eye irritation (ATSDR 1997, ATSDR 1999); some VOCs, such as benzene, are known carcinogens (ATSDR 1997, USEPA 1980). VOCs can enter aquatic systems through precipitation or through snowmelt runoff. Concentrations of VOCs in snowmelt runoff and the effects they have on aquatic systems are poorly understood.

Over the past decade, many studies have been conducted to determine the occurrence of hydrocarbons in precipitation, surface water and ground water in urban settings (Bruce 1995, Delzar et al. 1996), in rural areas in the central and eastern parts of the United States (Fenelon and Moore 1996, Terracciano and O'Brien 1997, Reiser and O'Brien 1998), and within the Rocky Mountain region (Bruce 1995, Ingersoll 1999). Specific studies have been conducted in Yellowstone National Park to assess hydrocarbon concentrations in snowpack associated with snowmachine use (Ingersoll 1999, Tyler et al. 2001). These studies concluded that snowpack from roadways used by snowmachines contained detectable concentrations of several VOCs (i.e., benzene, methyl tert-butyl ether, m- and p-xylene, o-xylene, and toluene) while snowpack from off-road locations contained only trace amounts of toluene. In addition to snowpack chemistry, Ingersoll (1999) also collected samples (i.e., one sample visit) of snowmelt runoff during May 1998 where detectable concentrations of VOCs were limited to toluene.

The goal of the present study was to determine potential impacts of snowmobiles on the water quality of surface waters near roads in Yellowstone National Park. Specific objectives were to (1) examine snowmelt runoff for the presence of specific VOCs, (2) determine if concentrations of any VOCs exceed safe drinking water criteria, and (3) predict the potential for impacts by VOCs on the fauna of streams near roads heavily used by snowmobiles in the park.

Study Area

Yellowstone National Park encompasses approximately 898,321 hectares of pristine landscape in the northwest corner of Wyoming and portions of southwest Montana and eastern Idaho. It was created in 1872 as the nation's first national park, primarily to protect the unique geothermal features. Yellowstone National Park and adjacent wilderness areas form the headwaters of three major drainages, including the Snake River, the upper Missouri River system (Gallatin and Madison rivers), and the Yellowstone River. The park contains four large, high-elevation lakes (Yellowstone, Lewis, Shoshone, and Heart lakes) and more than 2,000 smaller lakes and ponds. In addition, Yellowstone National Park contains >10,000 geothermal features, which include geysers, hot springs, fumaroles and mud pots. All water bodies within the park are designated as Outstanding Natural Resource Waters (Class 1) and are given the highest level of protection possible (Wyoming Department of Environmental Quality 2001). As a result, preventing degradation of water quality and maintaining high water quality standards are a high priority for park resource managers. Roads receiving the greatest amount of use by snowmobiles each year and of greatest concern to managers were those between West Entrance and Old Faithful, in the west-central region of Yellowstone National Park (Figure 1).

Methods

To characterize snowmelt runoff in areas of heavy use by snowmobiles in Yellowstone National Park, we established four sites along the heavily used road corridor between West Entrance and Old Faithful (Madison/Firehole river drainage; Figure 1). Three sites were located adjacent to the roadway in the vicinity of West Entrance, Madison Junction, and Old Faithful. A fourth site was used as a control and located approximately 100 meters from the snowmobile traffic and the roadway near Madison Junction, (Figure 1). Samples from three sites (Madison, Old Faithful, and control) were collected from flowing water directly related to snowmelt runoff for both sample years. Conversely, soils near the West Entrance sample site are comprised of surficial glaciofluvial alluvium deposits which consist primarily of sands and gravels and a slope of less than 5 percent (Rodman et al. 1996). Consequently, flowing water near the West Entrance site was absent during both sampling periods. This resulted in snowmelt water being collected from pooled areas on or adjacent to the roadway. In 2003 the sample location at West Entrance was located approximately 5 meters from the main roadway within a group of pine trees. In 2004 the sample location was relocated to an area on the road surface near the West Entrance gate where water accumulated adjacent to a berm of snow. Distance between the 2003 and 2004 snowmelt sites was approximately 20 meters. Sample water from the Madison site was collected as it egressed from a culvert and before it entered the Madison River. In addition to snowmelt runoff from groomed road surfaces, water from the Madison site captured

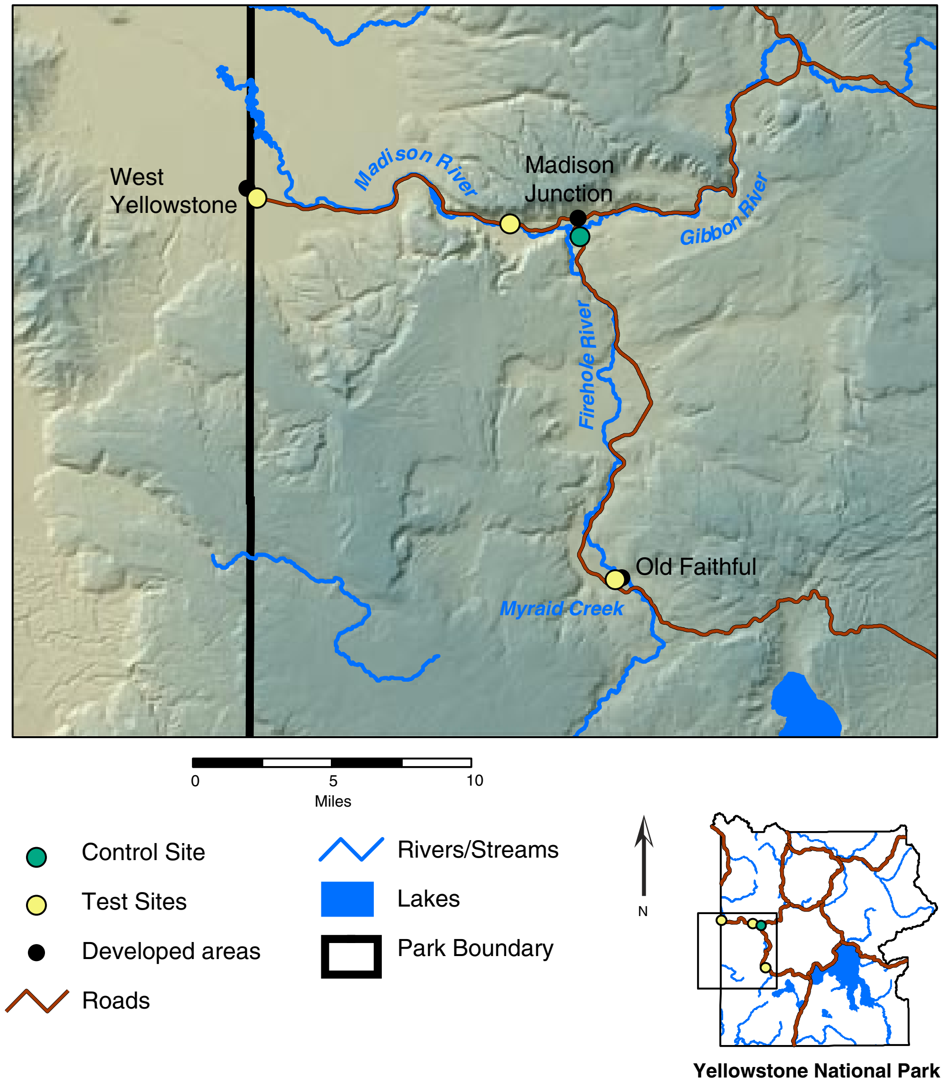

Figure 1. Study area within Yellowstone National Park with locations of sites where snowmelt runoff was collected in 2003 and 2004.

considerable amounts of snowmelt water from the adjacent hillside on the north side of the road. Samples from the Old Faithful area were collected from water flowing through a culvert that captured melt water drained from a large portion of a paved parking lot near the Old Faithful Visitor Center. This culvert directs the melt water into Myriad Creek whose confluence with the Firehole River is approximately 400 meters north of the sample site. In contrast to the Madison and Old Faithful locations, the control site near Madison Junction was located on a small intermittent stream that flowed through forested area that had been burned during the 1988 wildfires. All of the vegetation in this area was impacted by the 1988 fires and as a result, the site was characterized by abundant grasses, dead timber, and young lodgepole pine, *Pinus contorta*, approximately 3 meters in height.

During 2003 and 2004, sample collection began in mid-March after the end of the winter season when roads had opened for administrative travel by wheeled vehicles. Sampling generally took place when air temperatures were above 5°C with the sequence of site visits selected in random order. *In situ* measurements of water temperature, dissolved oxygen, pH, and specific conductance were made using a Hydrolab datasonde 4a multiparameter probe (Hach Environmental, Loveland, Colo.). In addition, a HACH 2100P (Hach Environmental, Loveland, Colo.) was used to measure turbidity. Instruments were calibrated twice daily (i.e., once prior to sampling and once after sampling was completed) following manufacturer instructions for calibration procedures.

Snowmelt runoff for VOC analysis was collected using USGS standard methods for determination of VOC compounds as described by Connor and others (1998). Water depth was too shallow to completely fill the sample vials; therefore a 2-liter opaque plastic bottle was used to collect snowmelt water from each site. Prior to collection, the bottle was rinsed with both deionized and sample water. The bottle was then submerged until approximately 500 ml of water was collected. Sample water was poured into four, 40-milliliter borosilicate U.S. Environmental Protection Agency (EPA) approved glass vials until a meniscus formed above the lip of each vial and all air bubbles were expelled. Each vial was immediately preserved by adding concentrated 1:1 solution of hydrochloric acid until a pH to 2 standard units (SU) was achieved. All vials were inspected for air bubbles and sealed with a cap having a Teflon-faced silicone septa. Each vial was labeled with a unique number that corresponded to the site data sheet.

In addition to sample vials, a set of two trip blanks were used to provide quality assurance/quality control information regarding sampling procedures and protocols. The trip blanks were provided by the U.S. Geological Survey (USGS), National Water Quality Laboratory, Denver, Colorado, and contained purified water. Trip blanks accompanied the field crew on randomly chosen sample dates. Analysis was conducted on these blanks to assure that no VOC contamination occurred during travel to and from the sample sites.

All sample vials and field blanks were shipped on ice to the USGS National Water Quality Laboratory, Denver, Colorado, for analysis. A gas chromatograph was used for VOC analysis (Connor et al. 1998). VOCs include a large list of hydrocarbons of which nine compounds were analyzed for this study including benzene, ethylbenzene, ethyl tert-butyl ether, isopropyl ether, *meta-* and *para*-xylene (m- and p-xylene), methyl tert-butyl ether, *ortho*-xylene (o-xylene), tert-pentyl methyl ether and toluene.

Results

Snowmelt Runoff in Spring 2003

In 2003, sampling began on 15 March and continued through 15 April. During this period, West Entrance, Old Faithful, and control sites were sampled nine days while the Madison site was sampled eight days. Summary statistics for all *in situ* water quality measurements are presented in Table 1. Water depth was quite shallow for all sites and ranged between 0.04 m and 0.13 m. Surface water temperatures ranged between 1.0°C and 11.8°C. The lowest mean water temperature of 3.94°C (range 1.3–7.1°C) was recorded at the West Entrance location; the highest mean water temperature of 7.09°C (range 5.2–9.4°C) was recorded for the control site. Individual measurements for pH ranged from 5.3 to 7.4 for all samples. The lowest mean pH value of 5.78 (range 5.3–7.0) was recorded at West Entrance and the highest mean pH value of 6.97 (range 6.8–7.1) at the control site. Snowmelt runoff typically has low specific conductance. Specific conductance was generally low for the three test sites which had ranges between 13 and 55 µSiemens cm^{-1} (µS) for all sample days combined. The lowest mean specific conductance of 21.44 µS (range 13–36 µS) occurred at West Entrance. Unlike the test areas, the control site exhibited the highest mean specific conductance of 144.78 µS (range 130–158 µS). Turbidity values, recorded in nephlometric turbidity units (NTU), were generally low for the West Entrance, Madison, and control locations with means of 14.01 NTU, 2.36 NTU, and 5.00 NTU respectively. The highest mean turbidity of 23.22 NTU (range 3.30–37.20 NTU) was recorded for Old Faithful (Table 1).

Of the nine VOC chemicals analyzed in snowmelt runoff, only five compounds were detected at least once during the 2003 sample season: benzene, ethylbenzene, m- and p-xylene, o-xylene, and toluene (Table 2). West Entrance and Old Faithful were the only two sites where all five compounds were detected during at least one sample event. Concentrations of benzene (0.0325 µg/L), ethylbenzene (0.0476 µg/L) and o-xylene (0.116 µg/L) were detected at West Entrance during one sample visit on 15 March 2003 (Figure 2). The presence of m- and p-xylene was detected during two visits to this site at concentrations of 0.1860 µg/L and 0.0105 µg/L, respectively, while toluene was detected in the snowmelt water collected from all nine visits (range 0.0243–0.1860 µg/L). In contrast, VOCs at the Madison site were not above the analytical detection limit (i.e., not detected) in any water samples collected during the sampling period (Table 2).

Volatile organic carbon compounds were most frequently detected at Old Faithful. The VOC

Table 1. Sample statistics for basic water quality measurements collected during the spring snowmelt runoff period of 2003.

[m, meter; °C, degrees celsius; mg/L, milligrams per liter; SU, standard units; µS, microSiemens; NTU, nephlometric turbidity units]

Site name	Sample statistic	Water depth (m)	Water temp. (°C)	Dissolved oxygen (mg/L)	pH (SU)	Specific cond. (µS)	Turbidity (NTU)
West Entrance	Mean	0.05	3.94	-	5.78	21.44	14.01
	Median	0.05	3.90	-	5.50	21.00	14.45
	Standard Error	0.00	0.63	-	0.63	2.34	1.82
	Std. dev.	0.01	1.90	-	0.53	7.02	5.14
	Range	0.01	5.80	-	1.70	23.00	15.70
	Minimum	0.05	1.30	-	5.30	13.00	6.80
	Maximum	0.06	7.10	-	7.00	36.00	22.50
	Num obs.	3	9	-	9	9	8
Madison	Mean	0.05	6.91	-	6.96	58.69	2.36
	Median	0.05	7.00	-	6.90	59.50	1.60
	Standard Error	0.00	0.39	-	0.05	1.17	0.47
	Std. dev.	0.01	1.09	-	0.14	3.31	1.33
	Range	0.01	3.50	-	0.40	8.50	3.20
	Minimum	0.04	4.80	-	6.90	55.00	1.20
	Maximum	0.05	8.30	-	7.30	63.50	4.40
	Num obs.	3	8	-	8	8	8
Old Faithful	Mean	0.06	7.00	8.17	6.86	27.44	23.22
	Median	0.05	8.00	7.80	7.00	24.00	26.70
	Standard Error	0.01	1.32	0.37	0.12	4.49	4.09
	Std. dev.	0.01	3.97	0.64	0.37	13.48	12.28
	Range	0.02	10.80	1.10	1.20	42.00	33.90
	Minimum	0.05	1.00	7.80	6.20	13.00	3.30
	Maximum	0.07	11.80	8.90	7.40	55.00	37.20
	Num obs.	3	9	3	9	9	9
Control	Mean	0.10	7.09	7.56	6.97	144.78	5.00
	Median	0.09	6.90	7.70	7.00	145.00	5.20
	Standard Error	0.01	0.53	0.11	0.03	3.07	0.58
	Std. dev.	0.02	1.60	0.25	0.10	9.22	1.73
	Range	0.05	4.20	0.60	0.30	28.00	5.80
	Minimum	0.08	5.20	7.20	6.80	130.00	2.60
	Maximum	0.13	9.40	7.80	7.10	158.00	8.40
	Num obs.	5	9	5	9	9	9

concentrations from this site were highest during a three-day sample period between 02 April and 10 April (Figure 3). During this time, concentrations of benzene ranged between 0.00992 µg/L and 0.0314 µg/L (Table 2). Highest concentrations of ethylbenzene (0.2740 µg/L), m- and p-xylene (1.4500 µg/L) and o-xylene (0.6920 µg/L) were detected during this period on 02 April 2003 (Table 2). Toluene was identified in the water from all nine site visits with a range between 0.0242 µg/L and 0.5890 µg/L. Snowmelt runoff from the control site, located approximately 100 meters from the roadway, surprisingly contained trace amounts of toluene during six of nine sample visits with a range from not detected to 0.0406 µg/L.

Snowmelt Runoff in Spring 2004

In 2004, snowmelt runoff samples were collected over a two-week period between 20 March and 03 April. Madison, Old Faithful, and control sites were sampled on ten days, and the West Entrance site was sampled on six days. Summary statistics for all *in situ* snowmelt water quality measurements are presented in Table 3. Water depth for all sites and sample dates combined ranged between 0.02 m and 0.10 m. Water temperatures ranged between 0.5°C and 11.6°C. The lowest mean water temperature of 4.12°C (range 1.0–9.3°C) occurred

Table 2. Summary of VOC concentrations at the four sample locations during the 2003 spring snowmelt runoff sample season.

[µg/L, micrograms per liter; e, estimated; <, less than reporting limit]

Site Name	Sample Date	Benzene (µg/L)	Ethylbenzene (µg/L)	m- and p-xylene (µg/L)	o-xylene (µg/L)	Toluene (µg/L)
West Entrance	03/15/03	e0.0325	e0.0476	e0.1860	0.116	0.1860
	03/30/03	<0.07	<0.06	<0.12	<0.14	e0.0243
	03/31/03	<0.07	<0.06	<0.12	<0.14	e0.0343
	04/01/03	<0.07	<0.06	<0.12	<0.14	e0.0557
	04/02/03	<0.07	<0.06	<0.12	<0.14	e0.0348
	04/09/03	<0.07	<0.06	<0.12	<0.14	e0.0797
	04/10/03	<0.035	<0.03	e0.0105	<0.07	e0.0566
	04/11/03	<0.07	<0.06	<0.12	<0.14	e0.0715
	04/15/03	<0.035	<0.03	<0.06	<0.07	e0.0259
Madison	03/31/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/01/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/02/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/09/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/10/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/11/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/14/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/15/03	<0.035	<0.03	<0.06	<0.07	<0.05
Old Faithful	03/15/03	<0.035	e0.0179	e0.0747	e0.0578	e0.0810
	03/30/03	<0.035	e0.0291	e0.1390	e0.0811	e0.0381
	03/31/03	<0.035	e0.0234	e0.1280	e0.0751	e0.0305
	04/01/03	<0.035	<0.03	<0.06	<0.07	e0.0242
	04/02/03	e0.0161	0.2740	1.4500	0.6920	0.6100
	04/09/03	e0.0314	0.1850	0.9840	0.5340	0.5890
	04/10/03	e0.00992	e0.0220	e0.0925	e0.0608	e0.0663
	04/11/03	<0.035	e0.0146	e0.0801	e0.0661	e0.0483
	04/14/03	<0.035	<0.03	<0.06	<0.07	e0.0283
Control	03/30/03	<0.035	<0.03	<0.06	<0.07	<0.05
	03/31/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/01/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/02/03	<0.035	<0.03	<0.06	<0.07	e0.0120
	04/09/03	<0.035	<0.03	<0.06	<0.07	e0.00997
	04/10/03	<0.035	<0.03	<0.06	<0.07	e0.0152
	04/11/03	<0.035	<0.03	<0.06	<0.07	e0.0150
	04/14/03	<0.035	<0.03	<0.06	<0.07	e0.0406
	04/15/03	<0.035	<0.03	<0.06	<0.07	e0.0212
Trip Blank	03/15/03	<0.035	<0.03	<0.06	<0.07	<0.05
	04/11/03	<0.035	<0.03	<0.06	<0.07	<0.05

at West Entrance; the highest mean water temperature of 7.89°C (range 5.3–10.4°C) occurred at Madison. Values for pH ranged from 5.3 to 7.4 for all sites combined and sample dates combined. The lowest mean pH of 6.49 (range 5.7–7.6) occurred at Old Faithful; the highest mean pH of 7.07 (range 6.3–7.4) occurred at West Entrance. The lowest mean specific conductance 16.5 µS (range 13–26 µS) occurred at West Entrance; the highest mean specific conductance value of 165.2 µS (range 156–174 µS) occurred at the control site. The lowest mean turbidity values were associated with West Entrance (5.3 NTU), Madison (2.75 NTU), and control sites (3.14 NTU) while the highest mean turbidity value of 35.21 NTU (range 5.3–250 NTU) was collected from the Old Faithful site.

Overall, presence of VOCs in snowmelt runoff illustrated similar patterns between the 2003 and 2004 sample years. Snowmelt runoff VOC concentrations

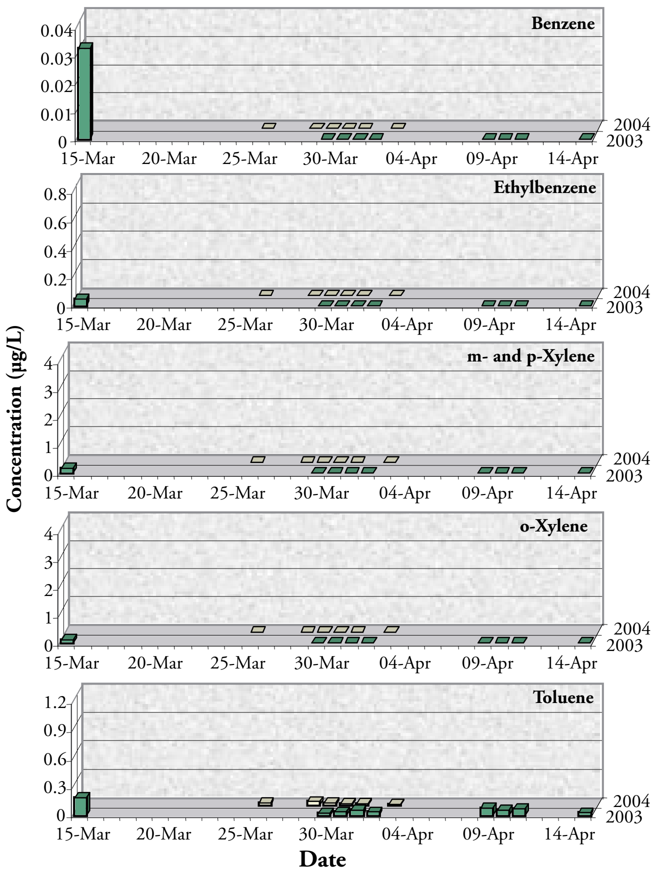

Figure 2. Concentration levels for five VOCs detected in snowmelt run-off at West Entrance, Yellowstone National Park, during spring 2003 and 2004 sample period. Note different concentration values for each compound. Values reported as zero were below detection limits.

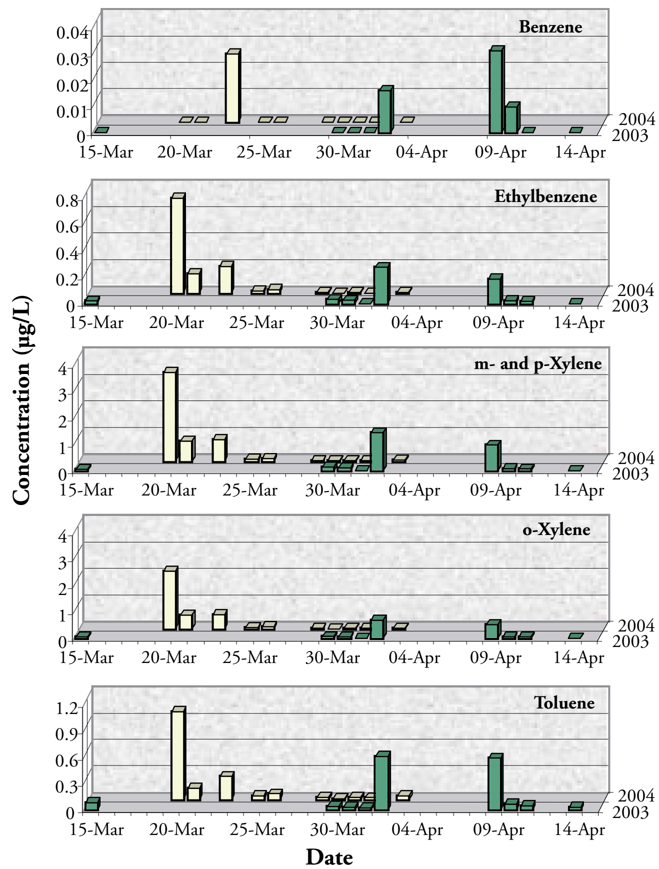

Figure 3. Concentration levels for five VOCs detected in snowmelt run-off at Old Faithful, Yellowstone National Park, during spring 2003 and 2004 sample period. Note different concentration values for each compound. Values reported as zero were below detection limits.

from spring 2004 are presented in Table 4. Six samples were collected from West Entrance and benzene, ethylbenzene, and o-xylene were not detected in any of the samples. Concentrations of m- and p-xylene were detected in two samples with concentrations of 0.007615 µg/L and 0.007854 µg/L collected on 31 March and 01 April, 2004, respectively (Table 4). Toluene was detected in water from all six site visits and ranged between 0.01183 µg/L and 0.03744 µg/L. The highest concentration was collected on 29 March with successive samples showing a general decreasing trend in toluene concentrations (Figure 2). Similar to the 2003 results, VOCs in water collected from the test site near Madison were below the analytical detection limit.

Samples from Old Faithful contained all five VOCs during the 2004 sample period. Benzene was detected from one sample with a concentration of 0.02631 µg/L on 23 March. At Old Faithful, the remaining four VOCs were collected at a greater frequency during 2004 compared to 2003 with the highest concentrations collected during the first three site visits (Figure 3). Ethylbenzene was found in eight of ten water samples, ranging from ND to 0.7202 µg/L; m- and p-xylene were found in ten samples, ranging from 0.01463 µg/L to 3.3650 µg/L; o-xylene was detected in nine water samples, ranging from ND to 2.1830 µg/L; and toluene was found in ten water samples, ranging from 0.01032 µg/L to 1.0080 µg/L (Table 4). Snowmelt runoff obtained at the control site once again contained trace amounts of toluene, but at a lesser frequency than the 2003 samples. Toluene concentrations ranged between ND and 0.01929 µg/L at the control site and were encountered early in the sample season during three sites visits (30 March and 02 April, 2004).

Table 3. Sample statistics for basic water quality measurements collected during the spring snowmelt runoff period of 2004.

[m, meter; °C, degrees Celsius; mg/L, milligrams per liter; SU, standard units; µS, microSiemens; NTU, nephlometric turbidity units]

Site name	Sample statistic	Water depth (m)	Water temp. (°C)	Dissolved oxygen (mg/L)	pH (SU)	Specific cond. (µS)	Turbidity (NTU)
West Entrance	Mean	0.08	4.12	-	7.07	16.50	5.30
	Median	0.08	3.45	-	7.15	14.50	3.15
	Standard Err.	0.01	1.32	-	0.16	2.05	1.78
	Std. dev.	0.02	3.24	-	0.39	5.01	4.36
	Range	0.06	8.30	-	1.10	13.00	10.70
	Minimum	0.04	1.00	-	6.30	13.00	1.80
	Maximum	0.10	9.30	-	7.40	26.00	12.50
	Num obs.	6	6	-	6	6	6
Madison	Mean	0.03	7.89	-	6.81	58.50	2.75
	Median	0.02	7.65	-	6.95	58.50	2.65
	Standard Err.	0.00	0.54	-	0.12	0.50	0.30
	Std. dev.	0.01	1.69	-	0.39	1.58	0.96
	Range	0.02	5.10	-	1.10	6.00	2.50
	Minimum	0.02	5.30	-	6.10	56.00	1.40
	Maximum	0.04	10.40	-	7.20	62.00	3.90
	Num obs.	10	10	-	10	10	10
Old Faithful	Mean	0.05	4.49	-	6.49	19.20	35.21
	Median	0.06	4.15	-	6.60	17.50	11.05
	Standard Err.	0.01	0.86	-	0.24	1.72	23.89
	Std. dev.	0.02	2.72	-	0.75	5.45	75.56
	Range	0.06	9.50	-	1.90	19.00	244.70
	Minimum	0.02	0.50	-	5.70	13.00	5.30
	Maximum	0.08	10.00	-	7.60	32.00	250.00
	Num obs.	10	10	-	10	10	10
Control	Mean	0.07	6.82	-	6.60	165.20	3.14
	Median	0.08	5.80	-	6.60	164.50	3.15
	Standard Err.	0.00	0.74	-	0.16	1.92	0.32
	Std. dev.	0.01	2.33	-	0.51	6.07	1.00
	Range	0.02	7.00	-	1.30	18.00	3.10
	Minimum	0.06	4.60	-	5.90	156.00	1.40
	Maximum	0.08	11.60	-	7.20	174.00	4.50
	Num obs.	10	10	-	10	10	10

Discussion

In general, most of the *in situ* water quality measurements are comparable between the two sample years and are within limits expected during a snowmelt runoff period on the Firehole and Madison rivers of Yellowstone National Park. Two exceptions however, were the lower pH values measured at West Entrance during spring 2003 and the high turbidity seen at Old Faithful during both sample years. Typically, snowmelt water is expected to have a pH near 7.0. The low mean pH for West Entrance (mean pH, 5.78) was most likely attributed to the physical area in which the water was obtained and not necessarily a characteristic of the chemical composition of the snowmelt run-off itself. Sample water from this site was collected from an off-road location within a group of lodgepole pine trees. Soils within a pine forest are naturally more acidic due to leeching of hydrogen ions from pine needles. This likely contributed to the lower pH values recorded at the West Entrance. By comparison, water samples collected from this area in 2004 were obtained directly from the paved road surface. The resulting mean pH value, when not in contact with forest soils, was 7.07.

High turbidities for the Old Faithful area are alarming because snowmelt run-off flows into Myriad Creek, a small tributary of the Firehole River. Sample water from this site reflects run-off that drains a paved

Table 4. Summary of VOC concentrations at the four sample locations during the 2004 spring snowmelt runoff sample season.

[μg/L, micrograms per liter; e, estimated; <, less than reporting limit]

Site Name	Sample Date	Benzene (μg/L)	Ethylbenzene (μg/L)	m- and p-xylene (μg/L)	o-xylene (μg/L)	Toluene (μg/L)
West	03/26/04	<0.021	<0.03	<0.06	<0.038	e0.02390
Entrance	03/29/04	<0.021	<0.03	<0.06	<0.038	e0.03744
	03/30/04	<0.021	<0.03	<0.06	<0.038	e0.02247
	03/31/04	<0.021	<0.03	e0.007615	<0.038	e0.01788
	04/01/04	<0.021	<0.03	e0.007854	<0.038	e0.01813
	04/03/04	<0.021	<0.03	<0.06	<0.038	e0.01183
Madison	03/20/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/21/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/23/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/25/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/26/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/29/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/30/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/31/04	<0.021	<0.03	<0.06	<0.038	<0.05
	04/01/04	<0.021	<0.03	<0.06	<0.038	<0.05
	04/03/04	<0.021	<0.03	<0.06	<0.038	<0.05
Old	03/20/04	<0.021	0.7202	3.3650	2.1830	1.0080
Faithful	03/21/04	<0.021	0.1493	0.7653	0.5019	0.1272
	03/23/04	e0.02631	0.2062	0.8113	0.5463	0.2634
	03/25/04	<0.021	e0.01822	e0.06662	e0.04316	e0.03967
	03/26/04	<0.021	e0.02782	e0.1087	e0.06731	e0.06515
	03/290/4	<0.021	e0.008128	e0.02824	e0.02128	e0.02670
	03/30/04	<0.021	<0.03	e0.01463	<0.038	e0.01032
	03/31/04	<0.021	e0.005851	e0.02332	e0.01493	e0.02351
	04/01/04	<0.021	<0.03	e0.01821	e0.01116	e0.02169
	04/03/04	<0.021	e0.008393	e0.03157	e0.01910	e0.04145
Control	03/20/04	<0.021	<0.03	<0.06	<0.038	e0.01538
	03/21/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/23/04	<0.021	<0.03	<0.06	<0.038	e0.01929
	03/25/04	<0.021	<0.03	<0.06	<0.038	e0.01143
	03/26/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/29/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/30/04	<0.021	<0.03	<0.06	<0.038	<0.05
	03/31/04	<0.021	<0.03	<0.06	<0.038	<0.05
	04/01/04	<0.021	<0.03	<0.06	<0.038	<0.05
	04/03/04	<0.021	<0.03	<0.06	<0.038	<0.05
Trip	03/23/04	<0.021	<0.03	<0.06	<0.038	<0.05
Blank	04/03/04	<0.021	<0.03	<0.06	<0.038	<0.05

parking lot with no soils or vegetation cover. As a result, runoff contained fine sand. This suspended material could have been produced from mechanical wear of the blacktop material in the parking lot. The material, as it enters the stream and is deposited, could have adverse affects on the chemical and physical properties of the water, as well as have harmful affects on the biological communities. During site sampling, a considerable amount of petroleum-like film was observed on the water's surface as it flowed from the culvert. This was probably a result of vehicular oil and petroleum products left on the parking area during the summer season and could have also been a factor with the higher turbidity measurements at this site.

At the Madison site no VOCs were detected in sample water during the two-year study period. Generally, snow from the plowed road surfaces in this area appeared to contribute minimal amounts of snowmelt to the overall sample. Most of the water volume appeared to originate from numerous freshwater springs flowing from the hillside adjacent to the roadway. If VOCs are introduced into the snowpack on groomed road surfaces, they are most likely diluted by the flux of water from these springs.

At West Entrance and Old Faithful, five VOC compounds were identified at least once during the two-year sample period. The highest concentration for benzene (0.0325 µg/L) was recorded at West Entrance on 15 March 2003, while Old Faithful had the highest frequency of occurrence for all VOCs with highest concentrations of ethylbenzene (0.2062 µg/L), m- and p-xylene (0.8113 µg/L), o-xylene (0.5463 µg/L), and toluene (0.2634 µg/L) on 23 March 2004. Although VOC concentrations were associated with areas having high volumes of snowmobile traffic, measurable amounts of VOCs remained considerably below EPA recommendation for VOCs in freshwater systems. The recommended freshwater acute criteria for benzene is 5,300 µg/L; for ethylbenzene, 32,000 µg/L; and for toluene, 17,500 µg/L (Rowe et al. 1997). Furthermore, aquatic toxicity information (Table 5) provides support that concentrations of VOCs measured during the 2003 and 2004 would have minimal, if any, affects on organisms within aquatic systems (Rowe et al. 1997).

The presence of toluene at our control site was not unusual and this VOC has been previously documented in the snowpack of off-road sites in Yellowstone National Park (Ingersoll 1999). The VOCs, including toluene, are produced as a byproduct of forest fires (Levine 1999). One possible explanation for the presence of low toluene levels at this our control site is that residual amounts of toluene exist within the burned timber that is abundant in the area around the control site. If toluene is contained within the cinders and charred wood, it could be released during the spring snowmelt period, producing detectable concentrations within the sample water. Additional monitoring needs to be conducted to confirm the persistence of toluene in snowmelt runoff at similar off-road locations.

Conclusion

Detectable concentrations of the five VOCs (i.e., benzene, ethylbenzene, m- and p-xylene, o-xylene, and toluene) were found in snowmelt runoff at the West Entrance and Old Faithful areas of Yellowstone National Park during 2003 and 2004. Both the West Entrance and Old Faithful areas receive high numbers of snowmobile use during the winter season. Additionally, trace amounts of toluene were detected at the control site, located away for roads and snowmobile traffic at Madison Junction. The highest frequency of VOCs occurred at the Old Faithful sample site however, these low VOC concentrations found within the snowmelt runoff would become more diluted as they entered a larger body of water such as the Firehole or Madison rivers, further limiting potential impacts to aquatic organisms. All detectable VOC concentrations were well below the EPA's recommended freshwater acute criteria for benzene, ethylbenzene, and toluene. Additional information regarding the toxicity of these five compounds on aquatic organisms provided further evidence that any impacts of VOCs found in snowmelt runoff on Yellowstone National Park's aquatic systems are likely negligible.

Although VOCs did not appear to be in high enough concentration to negatively impact aquatic systems, a concern arose during the study regarding the large amounts of petroleum based products that originated from snowmelt water observed at the Old Faithful site. These products could contain a different group of hydrocarbons, known as polycyclic aromatic hydrocarbons (PAH), which are much more persistent in the environment than VOCs. The PAHs do not easily dissolve in water and readily settle on the bottom of lakes and streams adhering to sediment particles (ATSDR 1995). In addition, PAHs can also accumulate in plant and animal tissues. Further studies are needed to identify concentrations of PAHs in effluent draining the Old Faithful area to determine possible affects on the aquatic environment there.

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Table 5. Aquatic toxicity information for selected volatile organic compounds measured by the U.S. Geological Survey (Rowe et. al. 1997).

[µg/L, micorgrams per liter; LC50, lethal concentration value; effective mean concentration]

Compound name	Taxonomic classification	Genus, species/ common name	Duration hours	Measures of toxicity (µg/L) LC50	EC50
Benzene	crustacean	*Gammarus fossarum*	120	66,007	
		Scud	96	68,283	
	crustacean	*Gammarus pulex*	48	42,000	
		Scud			
	crustacean	*Daphnia cucullata*	48	356,000	
		Water flea	48	390,000	
	crustacean	*Daphnia pulex*	96	15,000	
		Water flea	48	265,000	
	crustacean	*Daphnia magna*	1		6,300
		Water flea	24		10,000
	fish	*Salmo trutta*	1	12,000	
		Brown trout			
	fish	*Oncorhynchus mykiss*	96	5,300	
		Rainbow trout	96	5,900	
	fish	*Thymallus articus*	96	12,926	
		Artic grayling			
	insects	*Chironomus thummi*	48	100,000	
		Midge			
	invertebrates, misc	*Dugesia lugubris*	48	74,000	
		Turbellarian, flatworm			
o-Xylene	crustacean	*Daphnia magna*	24		1,000
		Water flea	48		3,185
	fish	*Oncorhynchus mykiss*	*96*	7,600	
		Rainbow trout	96	8,050	
m-Xylene	crustacean	*Daphnia magna*	24		4,700
		Water flea	48		9,556
	fish	*Oncorhynchus mykiss*	96	8,400	
		Rainbow trout			
p-Xylene	crustacean	*Daphnia magna*	24		3,600
		Water flea	48		8,494
	fish	*Oncorhynchus mykiss*	96	2,600	
		Rainbow trout			
Ethylbenzene	crustacean	*Daphnia magna*	48	75,000	
		Water flea	24	77,000	
	crustacean	*Daphnia magna*	24		1,810
		Water flea	24		1,930
	fish	*Oncorhynchus mykiss*	96	4,200	
		Rainbow trout	96	14,000	
Toluene	crustacean	*Daphnia magna*	24	310,000	
		Water flea	48	310,000	
	crustacean	*Daphnia magna*	1		
		Water flea	48		
	fish	*Oncorhynchus mykiss*	96	5,800	3,600
		Rainbow trout	96	24,000	6,000

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Effects of Lead and Hydrocarbons from Snowmobile Exhaust on Brook Trout (Salvelinus fontinalis)

Transactions of the American Fisheries Society
Volume 104, Issue 2, 1975
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Evelyn S. Adams[a]
pages 363-373

Abstract

Lead and hydrocarbons from snowmobile exhaust were found in the water at high levels during the week following ice-out in a Maine pond. Fingerling brook trout (Salvelinus fontinalis) held in fish cages in the pond showed lead and hydrocarbon uptake. These contaminants accumulated during the previous winter when snowmobile operation on the pond was equivalent to one snowmobile burning 250 liters of fuel per season on a .405 hectare pond with average depth of 1 m. Lead content of the water rose from 4.1 ppb before snowmobiling to 135 ppb at ice-out; exposed trout contained 9 to 16 times more lead than controls. Hydrocarbon levels undetectable prior to snowmobiling reached 10 ppm in the water and 1 ppm in exposed fish.

Trout held in aquaria for 3 weeks in melted snow containing three different concentrations of snowmobile exhaust also showed lead and hydrocarbon uptake. Their digestive tract tissue contained the most lead. (2 ppm) and gills the least (0.2 ppm).

Stamina, as measured by the ability to swim against a current, was significantly less in trout exposed to snowmobile exhaust than in control fish.

Science, Service, Stewardship

Summary of Selected Information on the effects of fine sediment on anadromous salmonids.

Dan Wilson, Fishery Biologist
Protected Resources Division
Southwest Region

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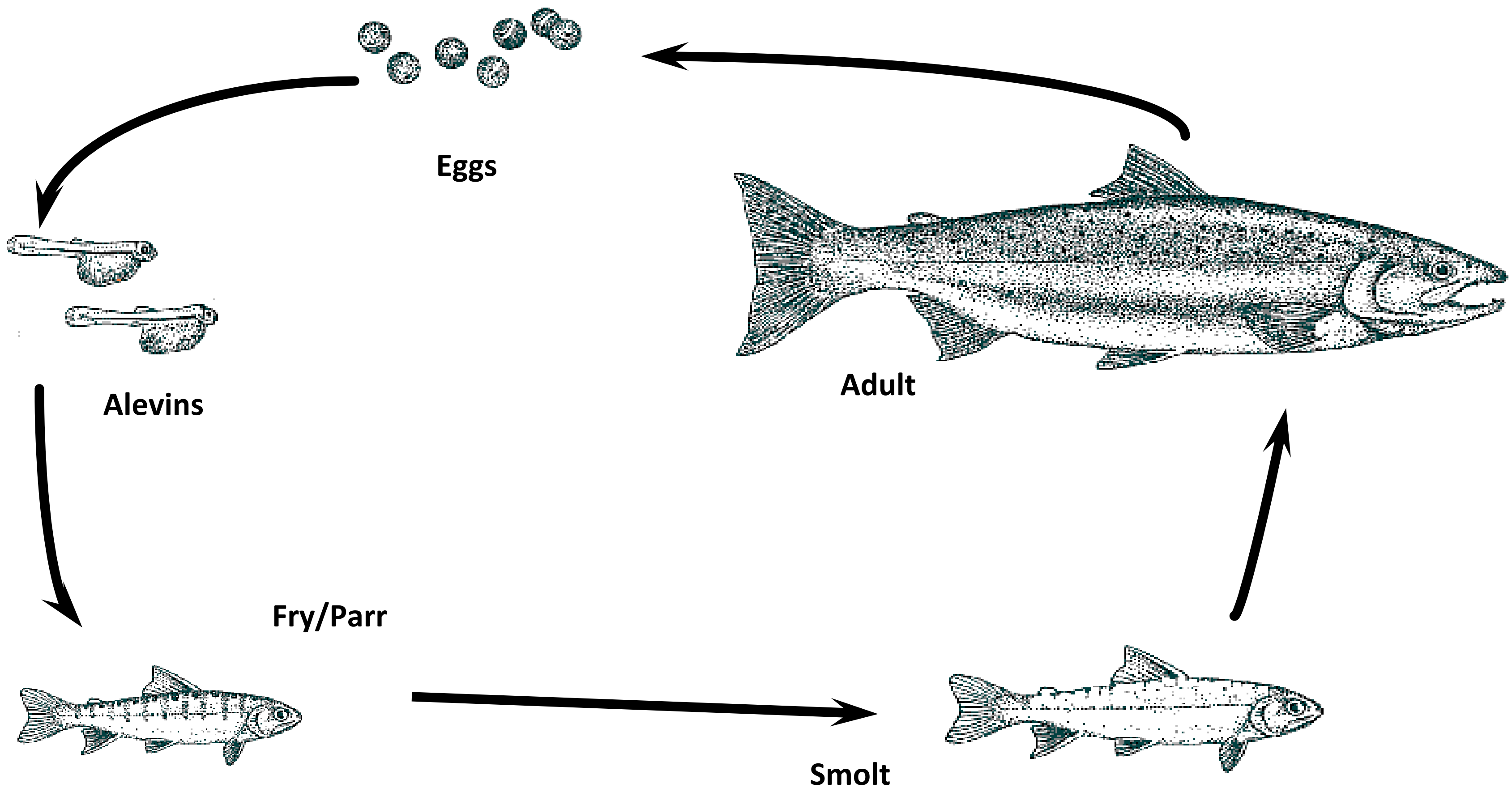
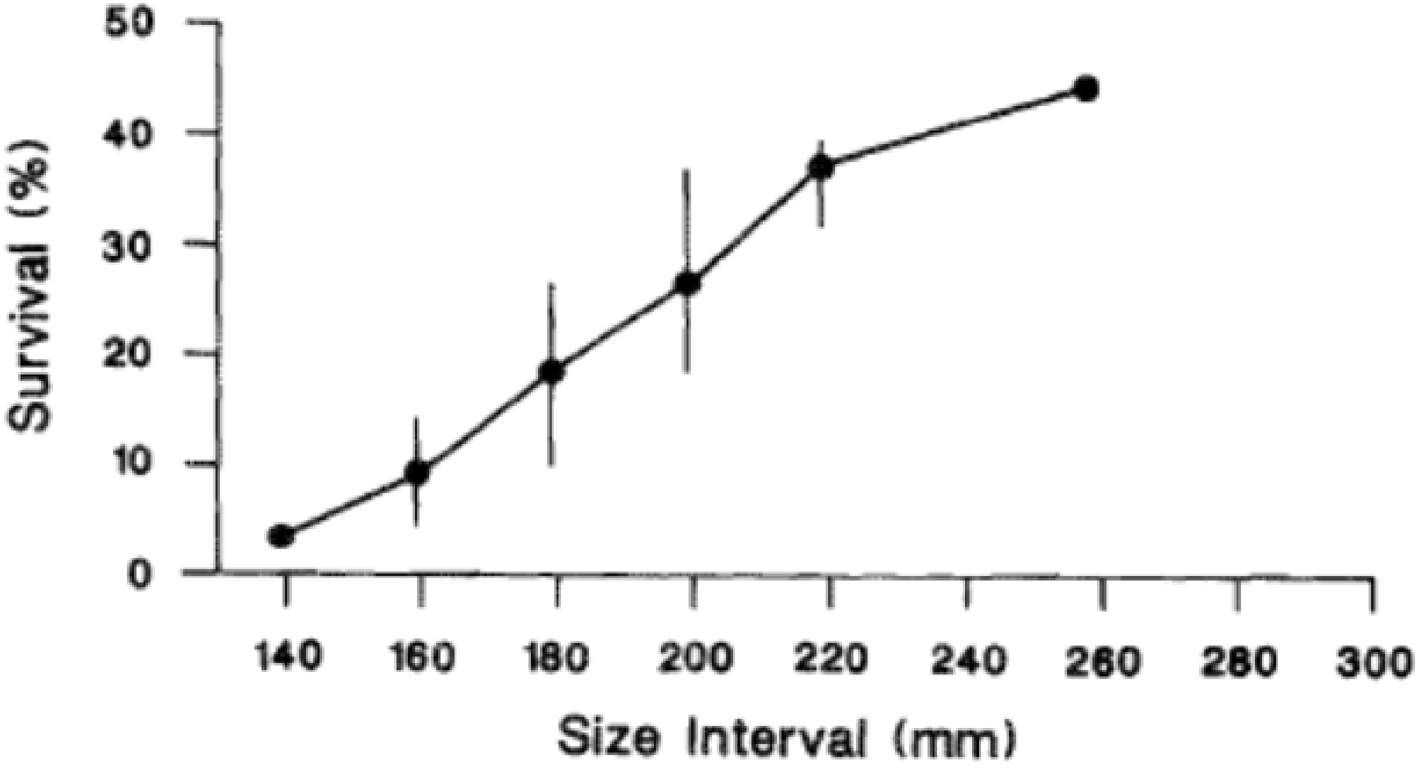
Salmonid Life Cycle



Ward et. al 1989



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Secret to Success

$$I - M - E = G$$

I = ingested food energy

M = metabolism

G = growth

E = energy excreted

(modified from Moyle and Cech 2004)

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Fine Sediment

- Particle sizes analyzed in the literature are generally grain sizes that are <2mm.

- Particle sizes in this range may be periodically transported as suspended load, bed load or deposited channel or on the floodplain.

- All three of these outcomes may have important consequences for stream biota

Suspended Load vs. Bedload

Suspended Load

- The load whose weight is supported and carried by the column of water within the interstices of the bed grains.

Bed Load

- The moving grain load whose immersed weight is carried by intermittent contact with the immobile bed.

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Effects of Fine Sediment on Salmonids

- Waters (1995) identified 3 relevant categories:
  - Direct effect of suspended load
  - Effects of salmonid reproduction success in redds (i.e. egg to fry survival).
  - Effects of deposited sediment on fry and juvenile salmonid habitat.

Effects of Suspended Load

- Stress index described by Newcombe and MacDonald (1991) is a function of the concentration of suspended sediment and duration of exposure.

- Shaw and Richardson (2001) tested the direct and indirect effects of the duration of exposure on rainbow trout.

- Lake and Hitch (1999) tested the acute effects of suspended sediment on juvenile coho salmon.

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Newcombe and MacDonald 1991

- Literature review of 70 sources.
- Developed a stress index based on sediment concentration and duration of exposure.

TABLE 1.—Ranking of effects of suspended sediments on fish and aquatic life.

Rank	Description of effect
14	>80 to 100% mortality
13	>60 to 80% mortality
12	>40 to 60% mortality, severe habitat degradation
11	>20 to 40% mortality
10	0 to 20% mortality
9	Reduction in growth rates
8	Physiological stress and histological changes
7	Moderate habitat degradation
6	Poor condition of organism
5	Impaired homing
4	Reduction in feeding rates
3	Avoidance response, abandonment of cover
2	Alarm reaction, avoidance reaction
1	Increased coughing rate

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Newcombe and MacDonald 1991

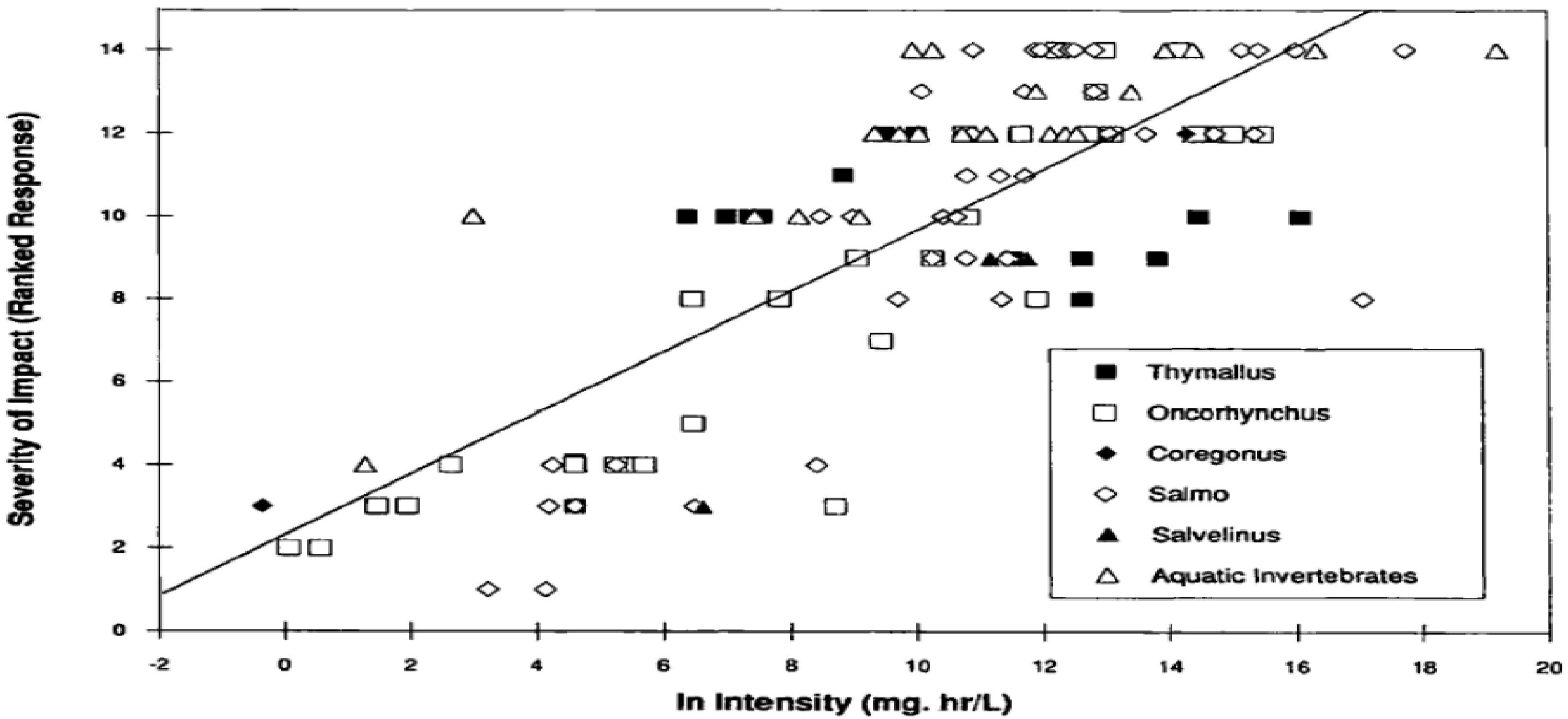

FIGURE 2.—Relationship between $\log_e$ (ln) of suspended sediment intensity and severity of effects on salmonid fishes and aquatic invertebrates. Severity of effect = 0.738 $\log_e$ intensity + 2.179; $r^2 = 0.638$, $N = 120$. Intensity is concentration (mg/L) times duration of exposure (h).

Shaw and Richardson 2001

- Field laboratory experiment testing acute effects of exposure duration to constant concentration of suspended sediment on RBT.
- 0-6 hour pulses released every 2nd day for 19 days.

Table 1. Fine sediment treatments applied to experimental channels.

Pulse duration (h)	No. sediment pulses	Mean sediment concentration ($mg \cdot L^{-1}$)	Standard error	Number of replicate channels	Dose ($mg \cdot L^{-1} \cdot h^{-1}$)
0	N/A	N/A	N/A	2	0
0.5	10	695.0	15.8	2	3 475
1.0	10	699.0	14.7	2	6 990
3.0	10	701.5	17.9	2	21 045
4.0	10	704.5	12.1	2	28 180
5.0	10	702.0	13.8	2	35 100
6.0	10	705.0	13.7	2	42 300

Note: Each of the seven treatments was replicated twice giving 14 experimental units. Dose is given as the total administered over the duration of the 19-day experiment and is calculated as the product of concentration and duration (product of pulse length and number of pulses). N/A, not applicable.

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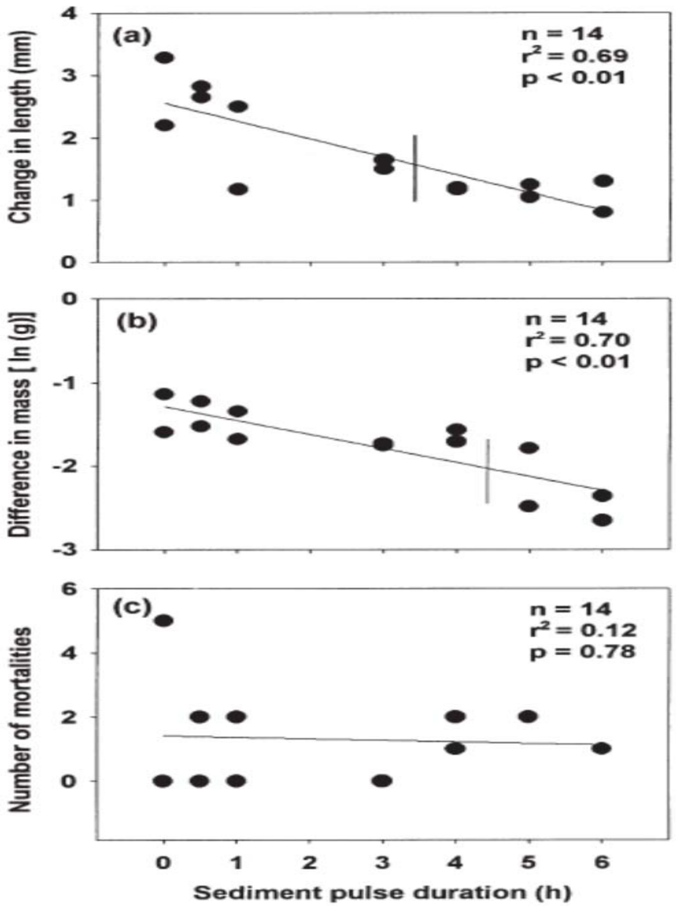
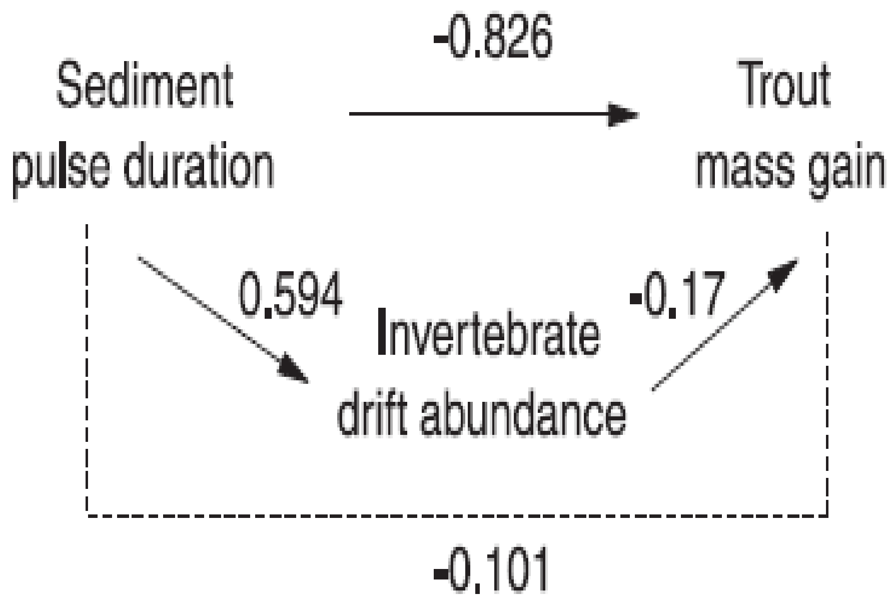
Shaw and Richardson 2001



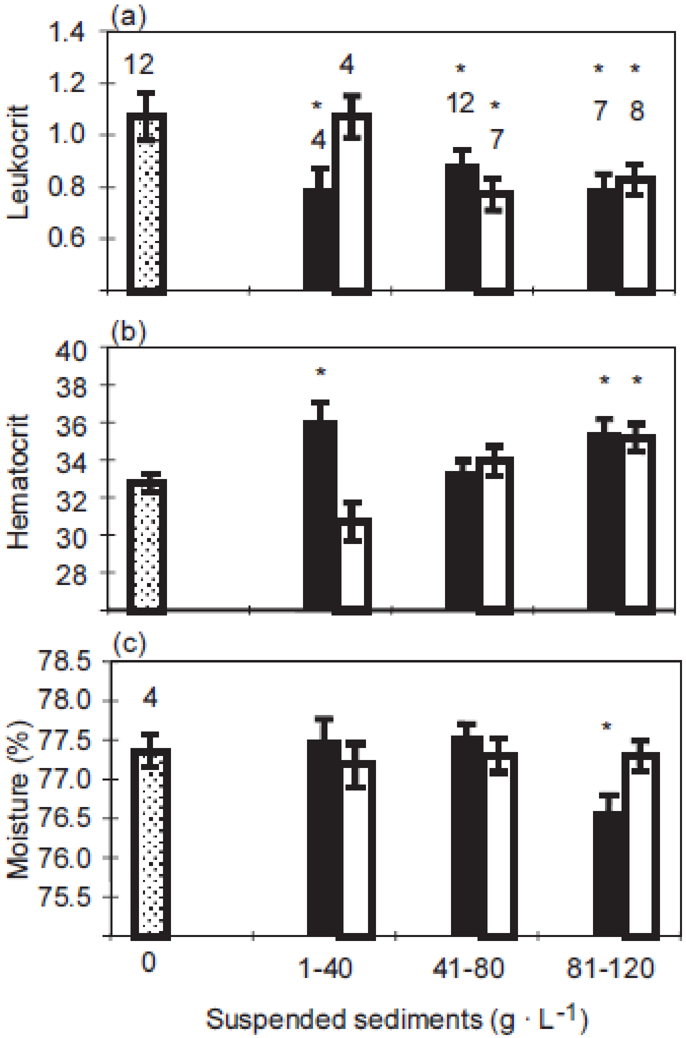


Lake and Hitch (1999)

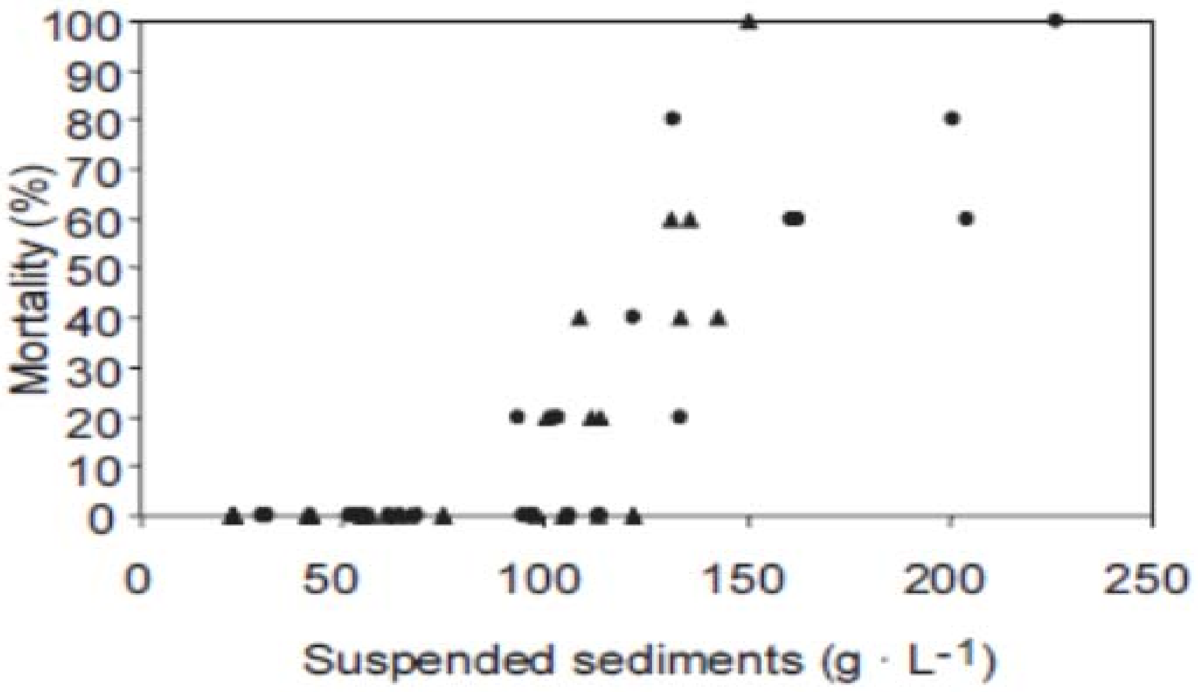
•Laboratory experiment testing the effects of sediment angularity and concentration as contributors to stress and mortality in salmonids.

•3 treatment categories at a 96 hour duration.

- Low (1–40 g/L)
- Medium (41–80 g/L)
- High (81–120 g/L)



Lake and Hitch (1999)



Regression equation, obtained by pooling round and angular data, was as follows: percent mortality = 0.046 ×concentration – 25.169 ($P < 0.001$, $r2 = 0.621$, $n = 59$)

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NMFS 2005

- Turbidity monitoring on an approved and lawful Timber Conversion Plan (Alder Springs Ranch).
  - NOAA OLE and OGC case #SW0204178

- NTMP, THP, TCP

- Results from independent assessments of physical water quality changes and fish population changes converged and both predicted approximately 20% juvenile steelhead mortality due to suspended sediment from roads and associated land disturbances in conformance with the California Forest Practice Rules.

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Effects of Suspended Load

- Take Home Messages:
    - The effects of suspended load on juvenile salmonids depends on a complex interaction between sediment concentration and duration of exposure.
    - The level of impact to juvenile salmonids resulting from the product of that relationship appears to have a linear correlation.
    - The existing policy for minimizing and avoiding the impacts of suspended sediment is flawed.

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Fine sediment and egg to fry survival (Jenson et al. 2009)

Table 3 Comparison of the change in the odds of survival resulting from a 1% increase in fines

Species	Metric	Change in sediment	Egg stage	Change in odds of survival (%)	95% Confidence interval
Chinook	<0.85 mm	+1%	Green and eyed	−16.9	(−20.4,−13.3)
Coho		+1%	Green and eyed	−18.3	(−23.9,−12.3)
Chum		+1%	Eyed	−13.6	(−18.3,−8.6)
All species		+1%	Green and eyed	−16.9	(−19.1,−14.6)
Chinook	<3.4–4.6 mm	+1%	Green	−6.7	(−9.0,−4.4)
		+1%	Eyed	−14.2	(−18.3,−9.8)
Steelhead		+1%	Green and eyed	−6.0	(−8.3,−3.6)
Coho		+1%	Green and unidentified	−9.2	(−12.4,−5.9)
Chum		+1%		−4.2	(−7.3, −1.0)
All species		+1%	Green and eyed	−7.1	(−8.5,−5.7)
Steelhead	<6.4 mm	10–11%	Green and eyed	−4.1	(−12.0, 4.6)
		30–31%		−10.1	(−21.6, 3.0)
		50–51%		−15.7	(−30.1, 1.5)

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Fine sediment and egg to fry survival

- Take Home Messages:
  - Even small increases in deposited fine sediment substantially increases the likelihood of take of listed salmonids.

Effects of deposited fine sediment on juvenile salmonid habitat.

- Suttle et al. (2004) experimentally manipulated fine bed sediment in a northern California river and examined responses of juvenile salmonids and the food webs supporting them.

- Harvey et al. (2009) setup a field experiment to measure the influence of deposited fine sediment on the survival and growth of juvenile rainbow trout in northwestern California

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Suttle et al. (2004)

- Field experiment in SF Eel River designed to isolate the effects of deposited fine sediment (i.e.<2mm).
- Steelhead reared in experimented riffles for 46 days.
- 6 experimental embeddedness treatments
  - 100% 80% 60% 40% 20% 0%

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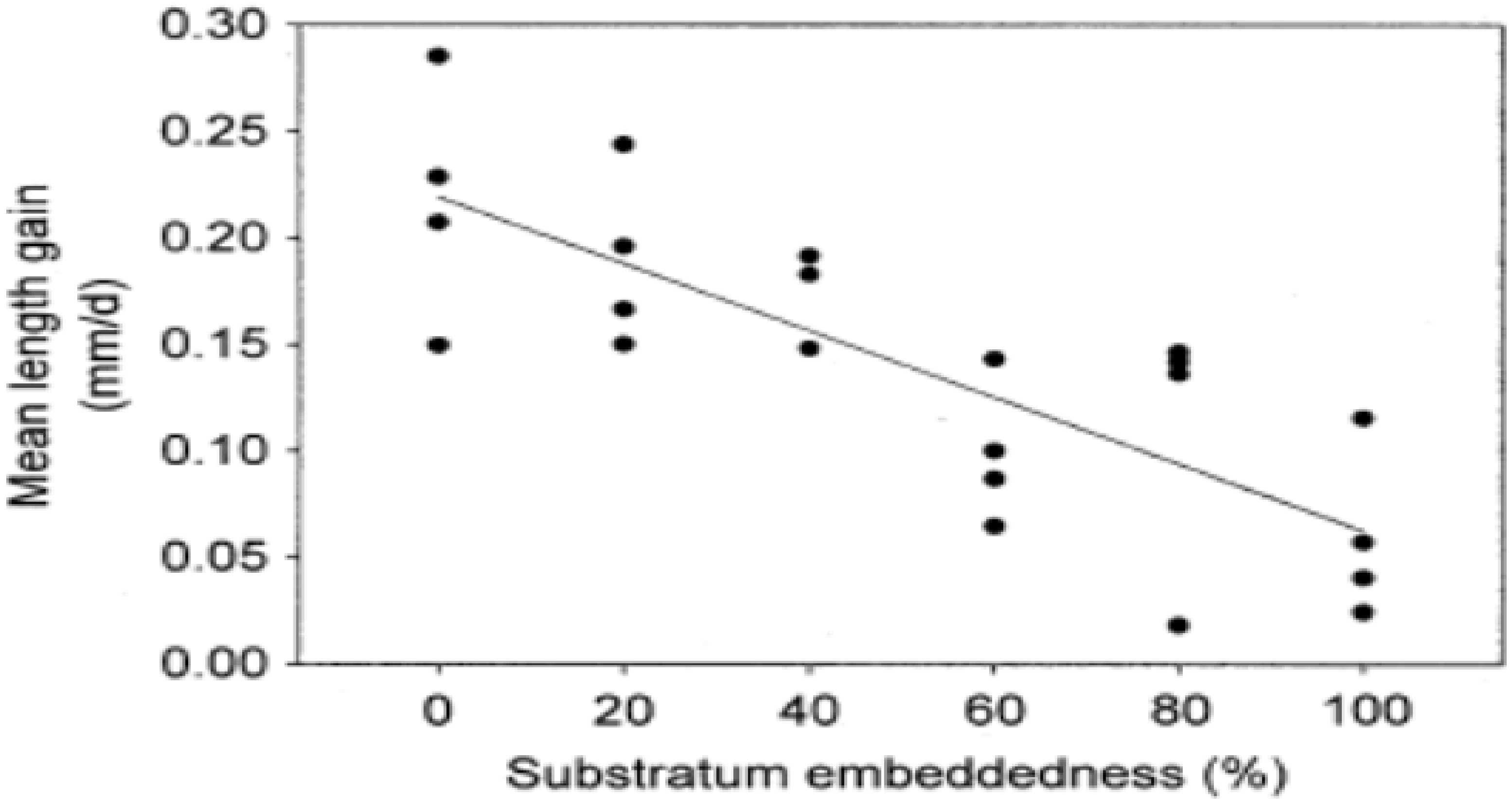
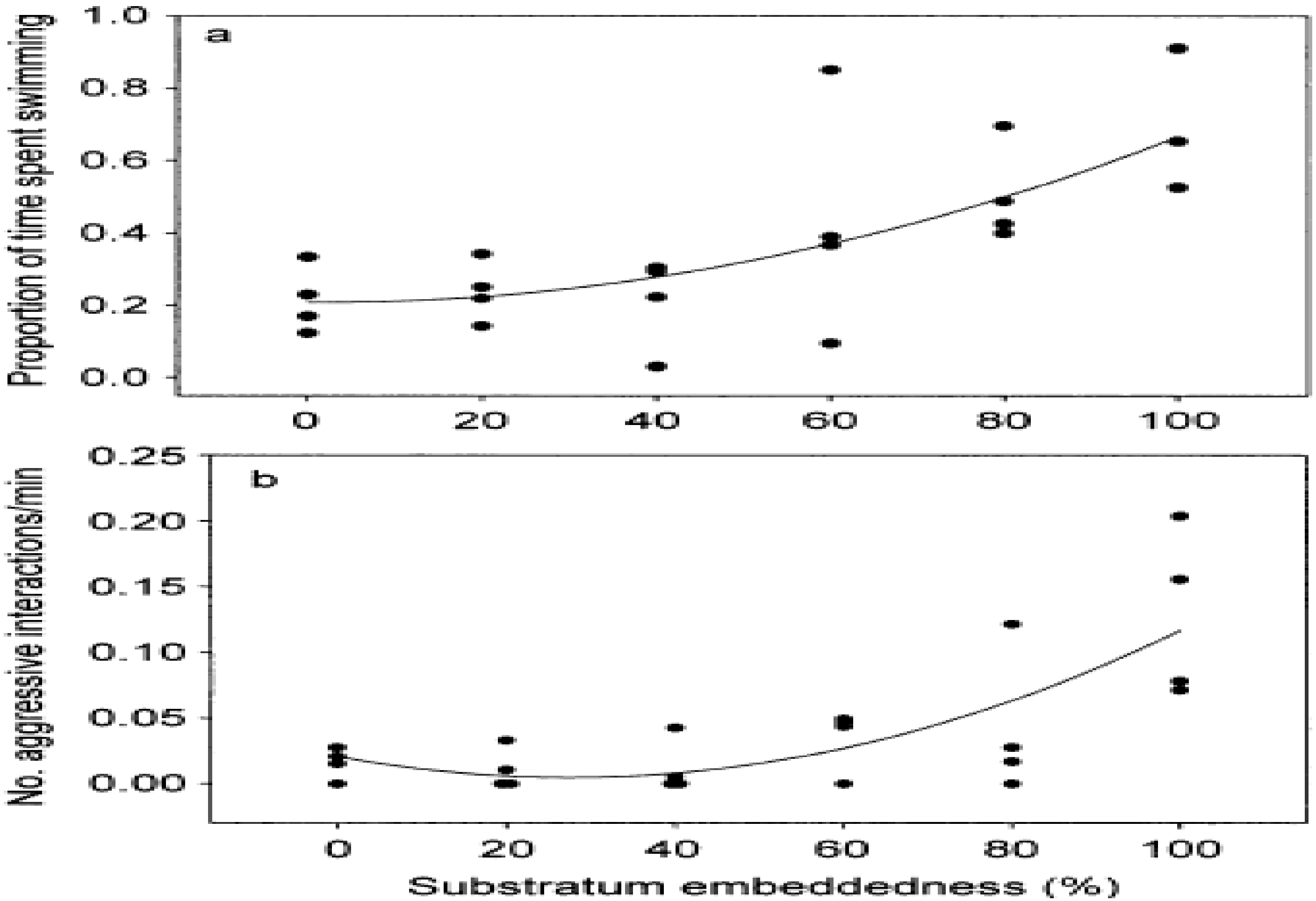
Suttle et al. (2004)

Suttle et al. (2004)



Suttle et al. (2004)



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Harvey et al. (2009)

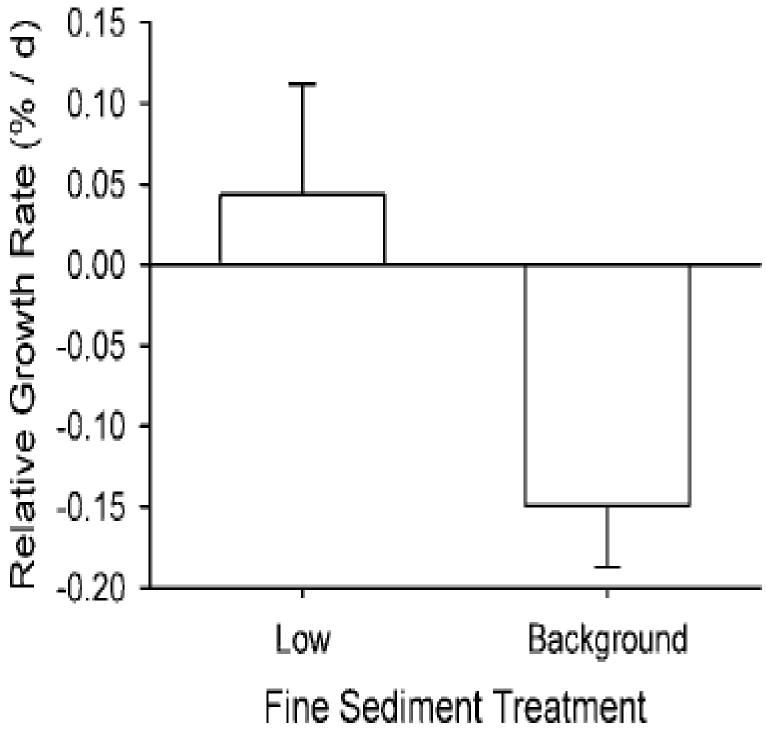
•Field experiment in Jacoby creek, designed to isolate the effects of deposited fine sediment (i.e. 13-mm-square mesh)

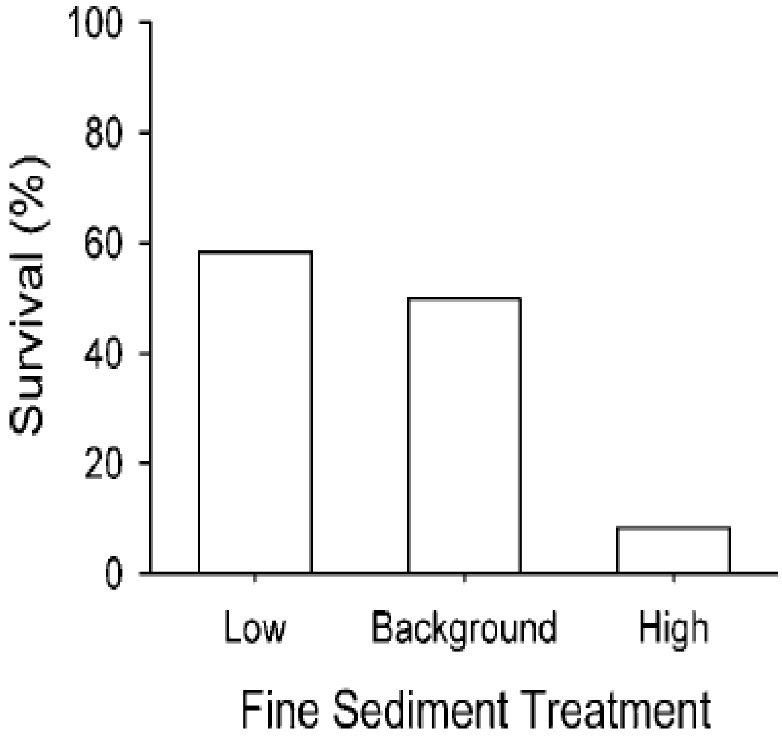
•RBT reared in experimented riffles for 40 days.

•3 experimental embeddedness treatments

- Low (about 0%), Background (25-50%), (High 50-100%)

Harvey et al. (2009)





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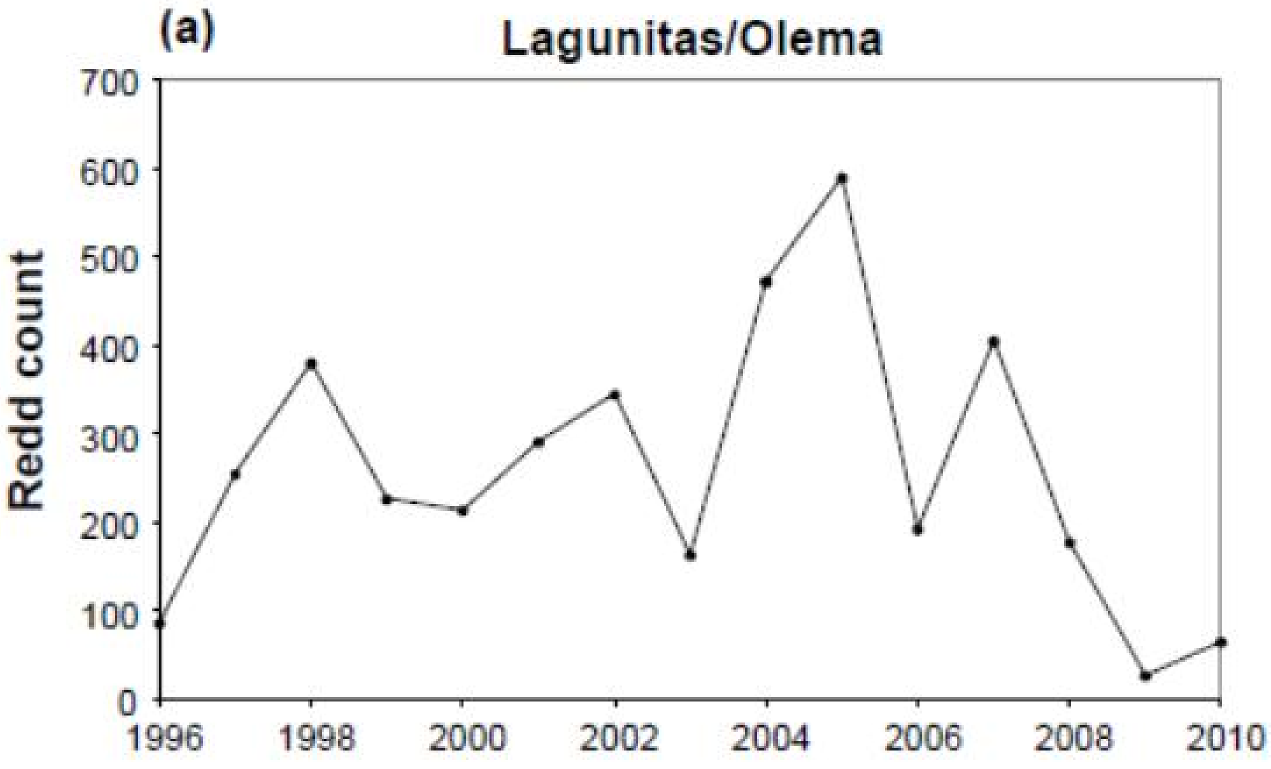
Effects of deposited fine sediment on juvenile salmonid habitat.

- Take Home Message

  - “The linear relationship between deposited fine sediment and juvenile steelhead growth suggests that there is no threshold below which exacerbation of fine-sediment delivery and storage in gravel bedded rivers will be harmless, but also that any reduction could produce immediate benefits for salmonid restoration.” Suttle et al. (2004)

Current Situation



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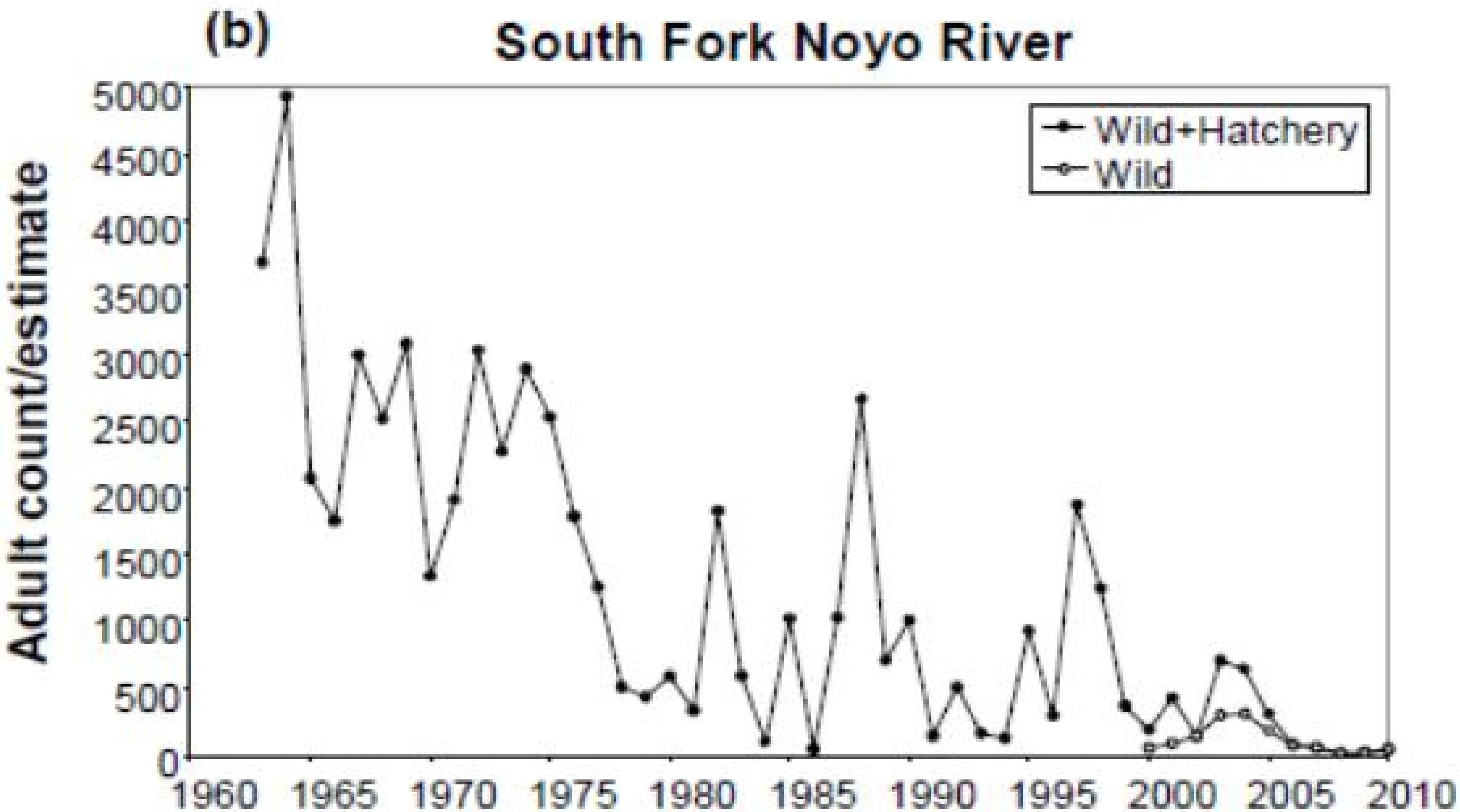

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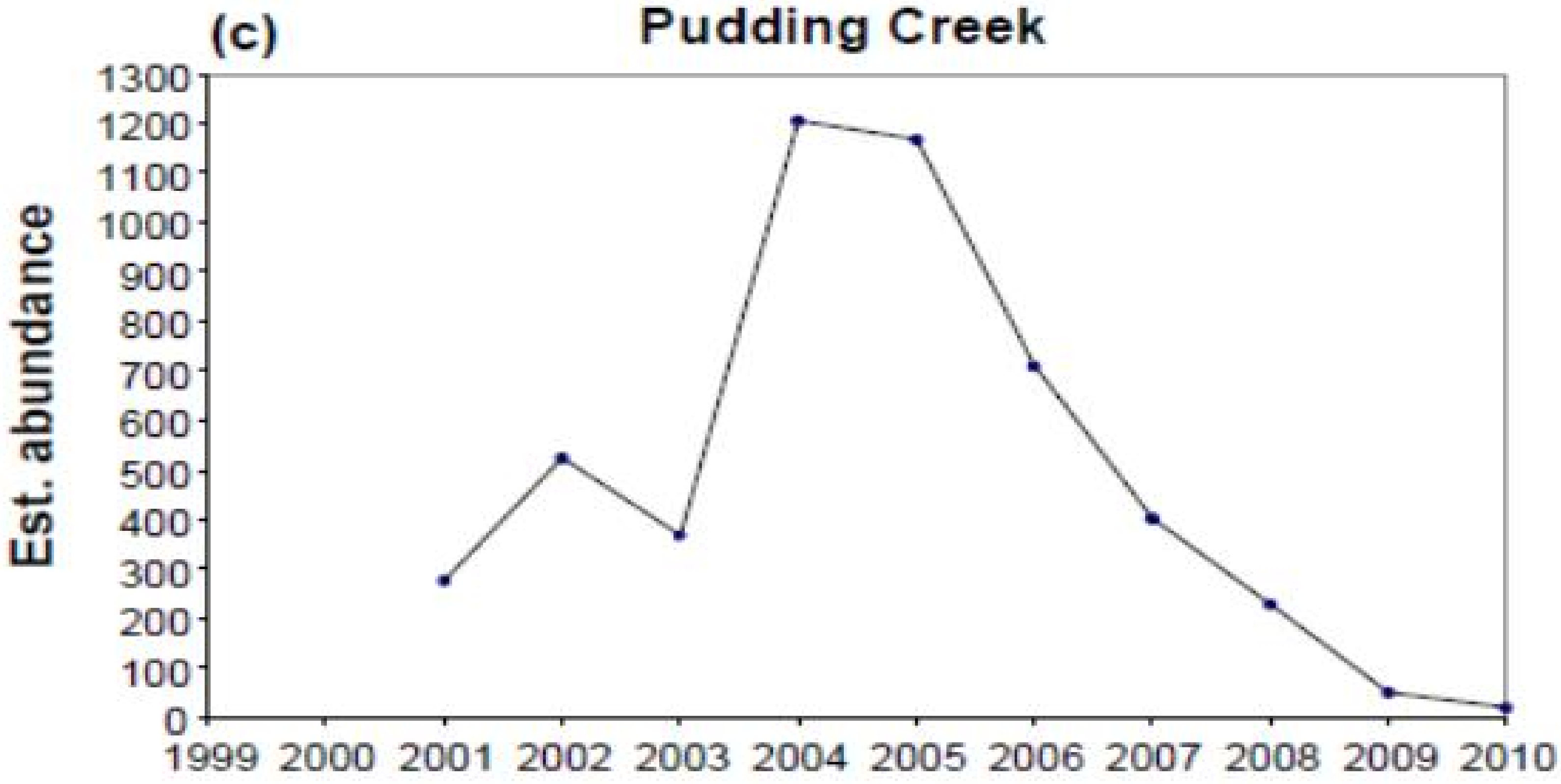

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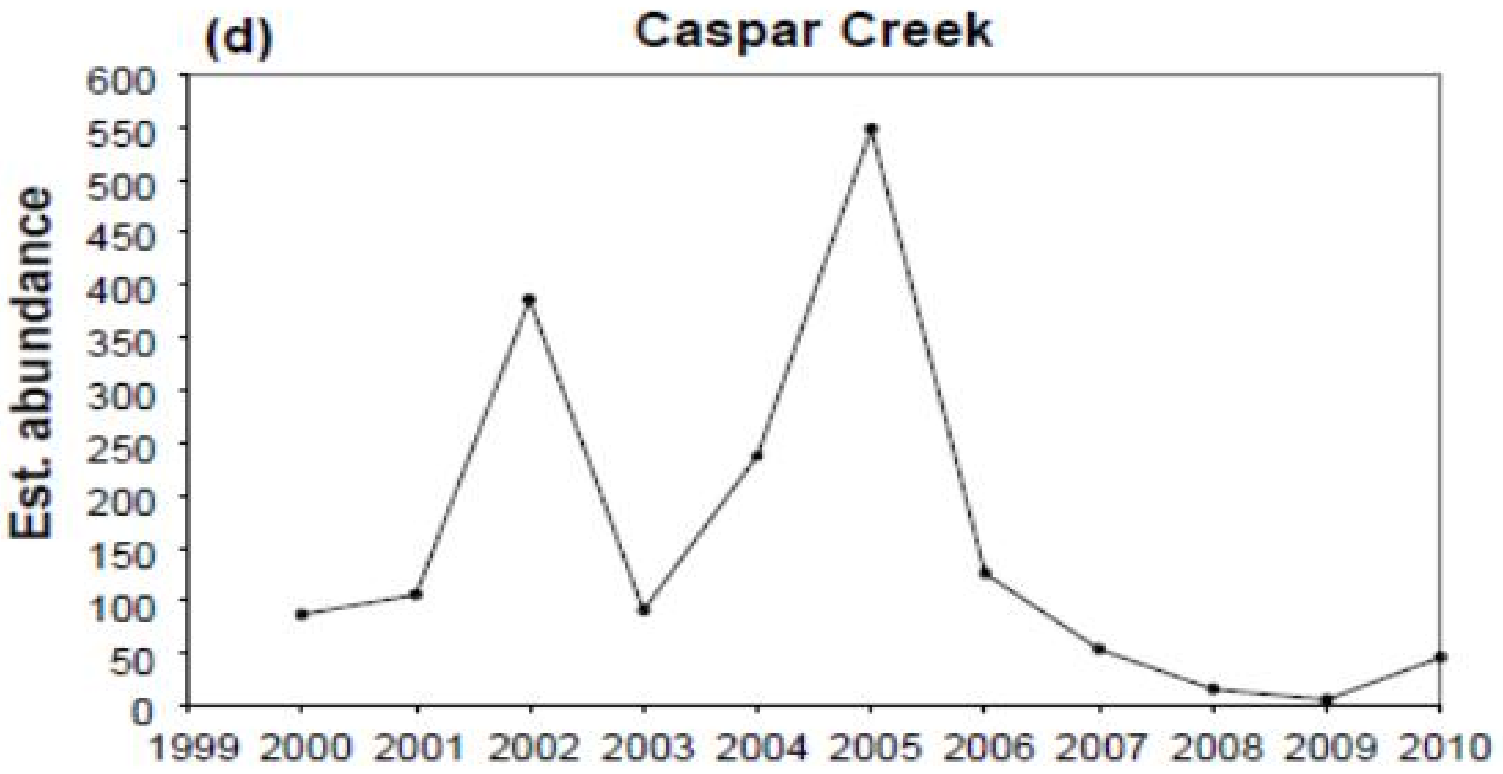
Current Situation



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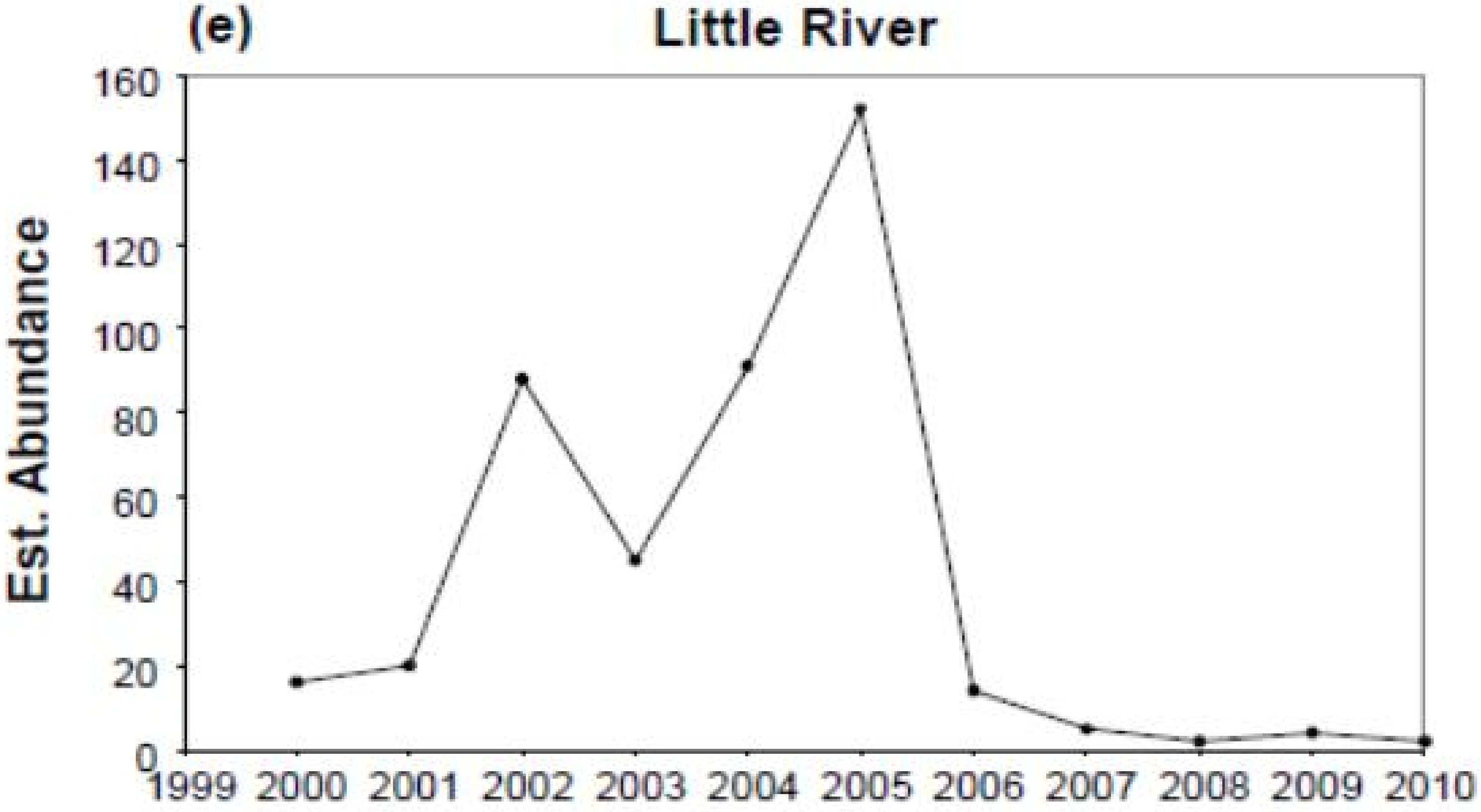
Current Situation



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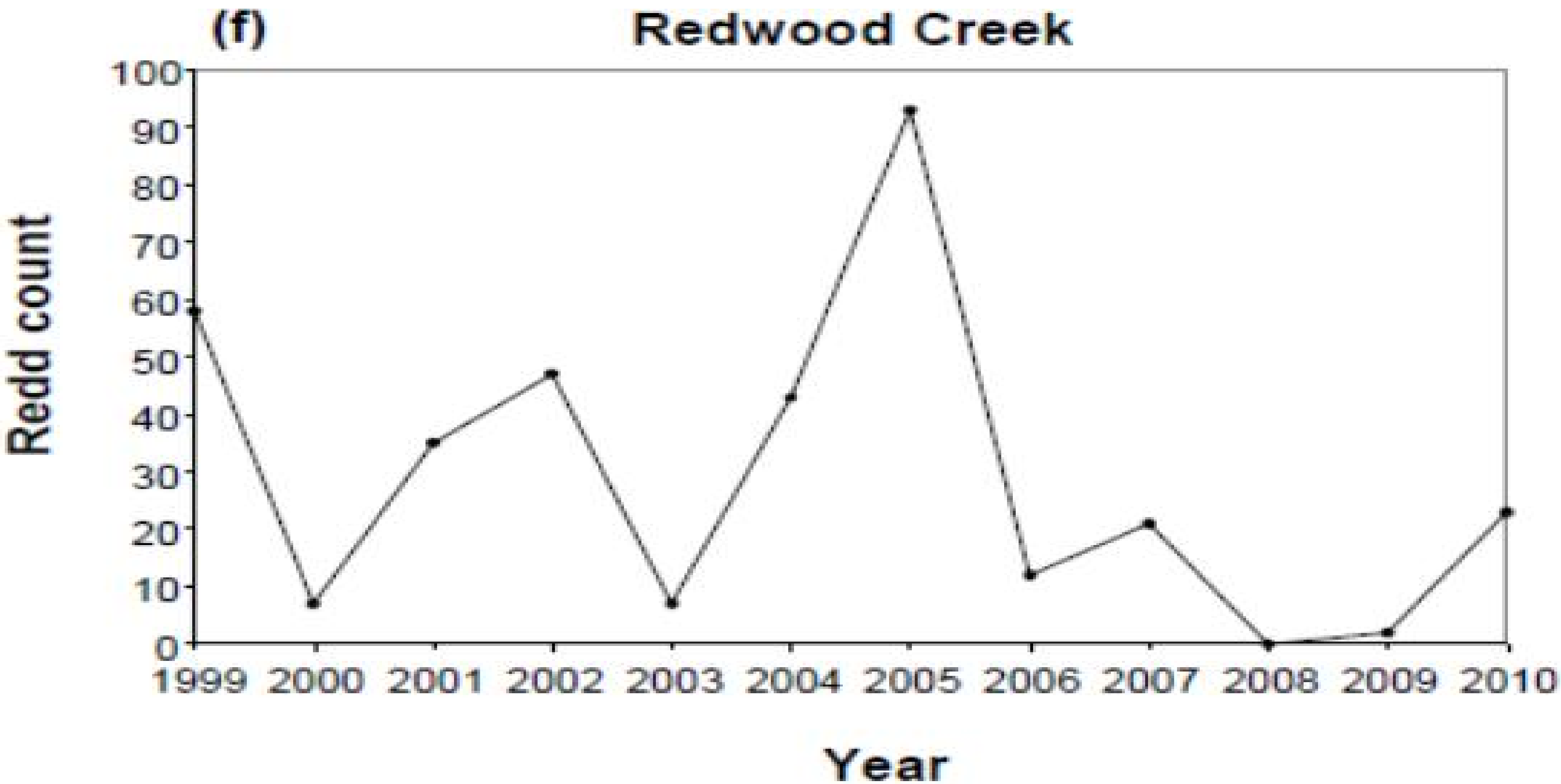
Current Situation



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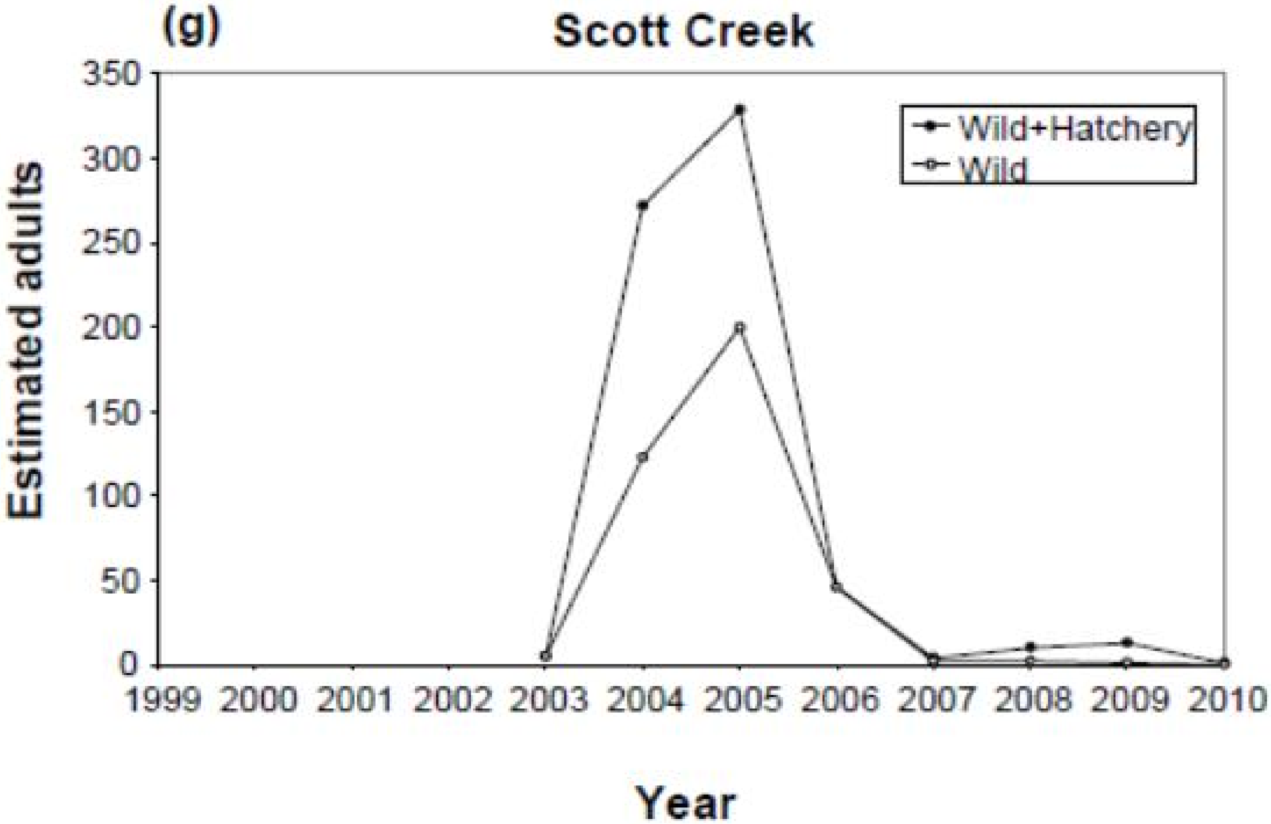
Current Situation



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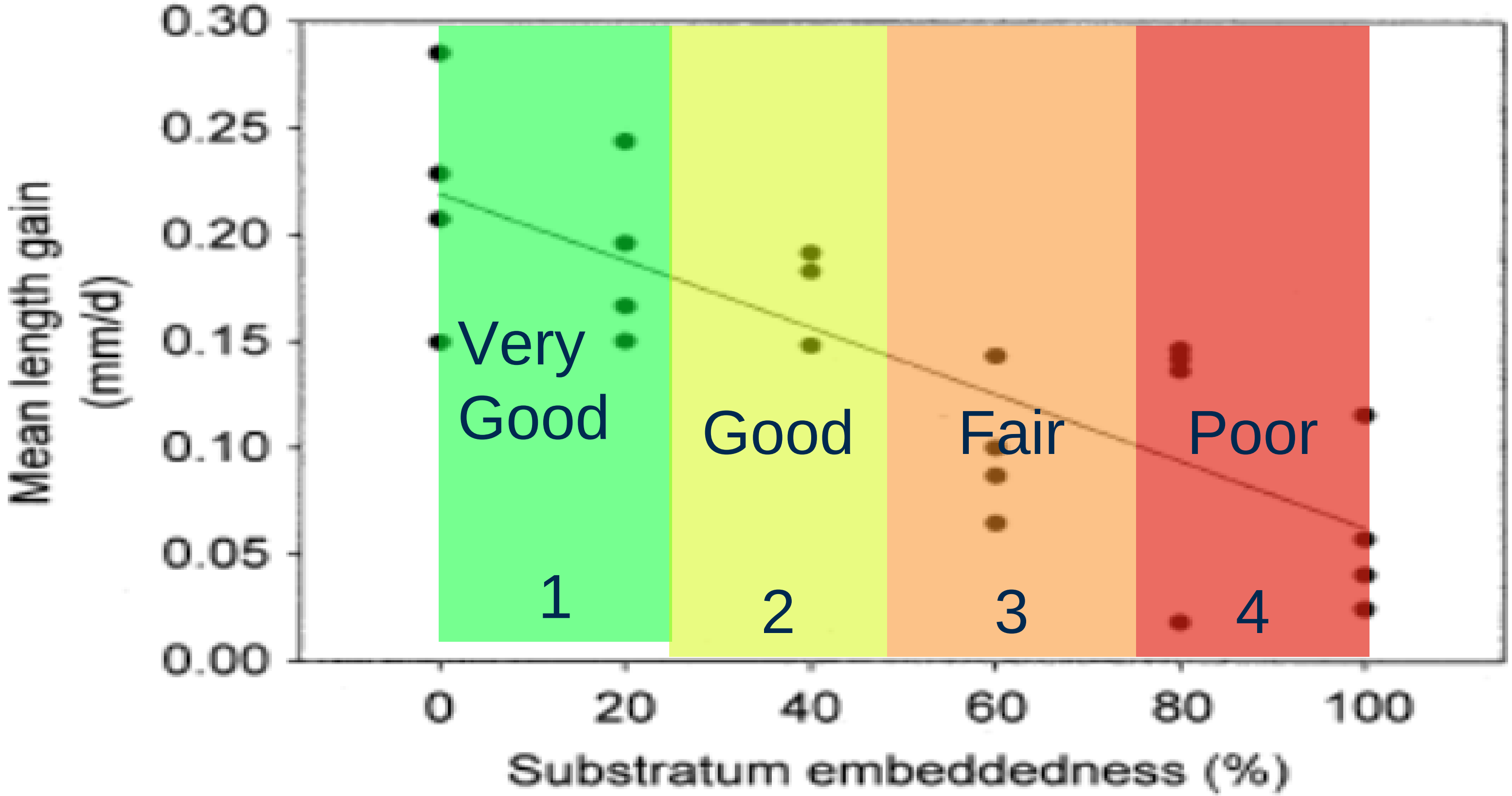
Existing Condition of Streams
CDFG habitat data



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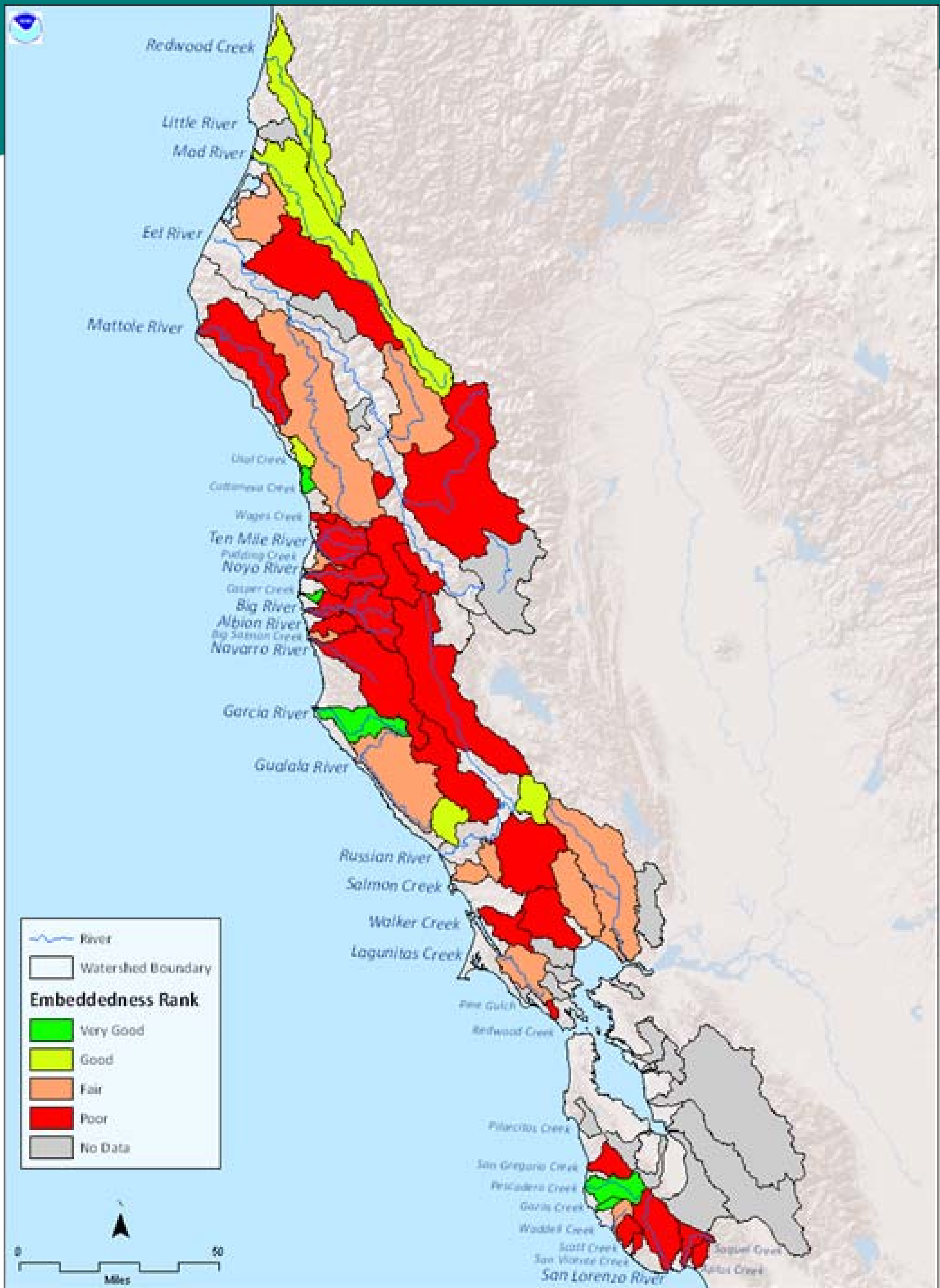
- 34% of the watersheds ranked good or very good.

- 48% of the watersheds ranked poor.



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Facts

- Fine sediment causes harm to salmonids when it is suspended and when it is deposited. Small increases in fine sediment significantly increases the likelihood of take in multiple life stages by:
  - Reduces growth and therefore decreases probability of ocean survival.
  - Suffocates eggs in redds
  - Increases stress
  - Increases mortality.
- The level of effect that suspended fine sediment has on salmonids depends on a complicated interaction between concentration and exposure.
- Anadromous salmonid populations are dangerously low.
- ASP watersheds are in poor condition
- The existing policy to prevent significant discharge is insufficient and ineffective in minimizing harm.

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Conclusions

- To reduce risk and uncertainty to listed salmonids any rule needs to meet these objectives:

  - High standard of erosion control

  - Clear and enforceable

  - Specifically regulate activities that cause discharge rather than regulate the quantity of discharge.

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The incidence and role of gullies after logging in a coastal redwood forest

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ABSTRACT

The distribution and morphological characteristics of channels were mapped in a redwood forest at Caspar Creek, California, USA, to evaluate the extent to which recent logging has influenced channel conditions in the area. In the North Fork Caspar Creek watershed, second-cycle logging of the early 1990s appears to have triggered increased coalescence of discontinuous gullies within clearcut tributary watersheds, and upstream channel limits in logged watersheds are now located significantly farther upslope than in control watersheds. The magnitudes of observed increases in peakflow after logging are consistent with the change in drainage density. Relations between channel morphological variables and indices of stream power are less well-defined in logged watersheds than in controls, suggesting that logging may have led to disruption of previously established channel forms. Correlations between suspended sediment yields and indices of gully erosion suggest that in-channel erosion associated with hydrologic change is an important source of post-logging sediment at Caspar Creek. Common sediment-control measures, such as use of riparian buffer strips and reduction of road surface erosion, would not be effective for reducing sediment input from this source.
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1. Introduction

Gullying has been triggered by increased runoff and reduced vegetation cover in many areas and is of great concern to land managers (Valentin et al., 2005). Gullies contribute to loss of fertile soils, disrupt transportation networks, and depress water tables in floodplains; and they create persistent sources of sediments that reduce water quality, fill reservoirs, aggrade downstream channels, and impair aquatic habitats. Gullying can increase sediment yields both through incision of existing channels and by expansion of drainage networks into previously unchanneled swales. In the latter case, channel incision may increase transport capacity through the swale, providing hillslope-derived sediment more direct access to downstream channels (Reid, 1989).

Most studies of gullies have been carried out in agricultural lands, grasslands, or arid areas, where the features are most visible and where they most directly challenge land-management activities; gullies in forests are less commonly encountered. Gullying occurred in some areas after conversion of forest to pasture or agriculture (e.g., Gábris et al., 2003; Parkner et al., 2006). In several areas, however, gullies have been identified as major sediment sources within unconverted forest lands. Often forest gullies are associated with road drainage or with areas compacted by logging equipment (e.g., Weaver et al., 1995; Croke and Mockler, 2001), but in other cases gullying appears to be a more generalized response to forest management (Heede, 1991) or to temporary vegetation changes upslope of the forested area (Vanwalleghem et al., 2003) or is an inherent feature of the forested setting (Parkner et al., 2007).

In part because of the long duration of forest management cycles and the inability to detect forest gullies on historical aerial photographs, the incidence of forest gullies and their relation to forest management remain poorly understood, particularly in settings where overland flow is uncommon. Observed changes in gully activity with forest conversion and subsequent reforestation (e.g., Gábris et al., 2003; Parkner et al., 2006) indicate that catchment vegetation can influence gullying and that those influences vary by vegetation type. Forest vegetation commonly has larger roots and produces larger quantities of coarse litter than other vegetation types, and forests provide woody debris that retains sediment on hillsides and in small channels (Maser et al., 1988). Vegetation type is also expected to influence runoff volumes (Bosch and Hewlett, 1982), peakflow magnitudes (Guillemette et al., 2005), and snowmelt timing (Winkler et al., 2005), each of which could influence gully erosion rates during short-term perturbations in vegetation cover. Vegetation can also regulate gully activity at a smaller scale. For example, Molina et al. (2009) have shown that herbaceous and shrubby vegetation on gully floors is effective in trapping sediment.

During recent forestry planning efforts in NW California, unanswered questions were raised concerning the role of management-related channel erosion in sediment production. This paper presents the results of a study implemented in the Caspar Creek Experimental Watersheds of north coastal California, USA, to (i) evaluate the distribution of incised channels and associated headcuts in the area; (ii) assess the relative importance of gully erosion as a sediment

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source there; and (iii) identify potential influences of forest land management on the extent and character of the gullied reaches.

2. Study area

The Caspar Creek Experimental Watersheds (N39°21′ W123°44′) include the 424-ha South Fork and 473-ha North Fork tributaries of Caspar Creek, which drains to the Pacific Ocean 10 km south of Fort Bragg, CA (Fig. 1). The Caspar Creek catchment is deeply incised into a flight of uplifted marine terraces formed over more than 300,000 years on Franciscan sandstone and shale (Merritts et al., 1991). Elevation in the experimental watersheds ranges from 37 to 320 m, with hillslopes steepest near the stream channel and becoming gentler near the broad, rounded ridgetops. About 35% of the slopes are lower than 17° and 7% are steeper than 35°. Longitudinal channel profiles are generally concave, but many include low-gradient reaches (1° to 5°) immediately followed by relatively short steeper reaches (5° to 20°). Mainstem channels and many tributary segments are bordered by narrow valley flats. Radiocarbon dating of charcoal in 3- to 4-m-deep valley fills in upper reaches of the North Fork suggests that deposition began about 7000 ^{14}C YBP and proceeded episodically through the middle to late Holocene (Steven Reneau, Los Alamos National Laboratory, personal communication, 1989).

The climate is typical of coastal watersheds in the region: winters are mild and wet, while summers are mild and dry. About 95% of the average annual precipitation of about 1200 mm occurs in October through April, and most rain falls in storms of long duration but low intensity. Snow is uncommon.

Soils are predominantly gravelly to sandy loams with a typical depth of 1.5 m. The upper meter of the most common soils includes 25 to 50% gravel and cobble and 30 to 50% clay and silt, and has a bulk density of 1.4 to 1.6 Mg m^{-3} (Wosika, 1981). Subsurface stormflow is rapid, and saturated areas are uncommon and drain quickly after storms. Soil pipes are present in most swales and transport a substantial discharge of storm flow to low-order channels (Albright, 1991). Channel flow often becomes spatially intermittent within a few weeks of a winter storm in catchments smaller than 20 ha (L. Keppeler, USFS, personal communication, 2008).

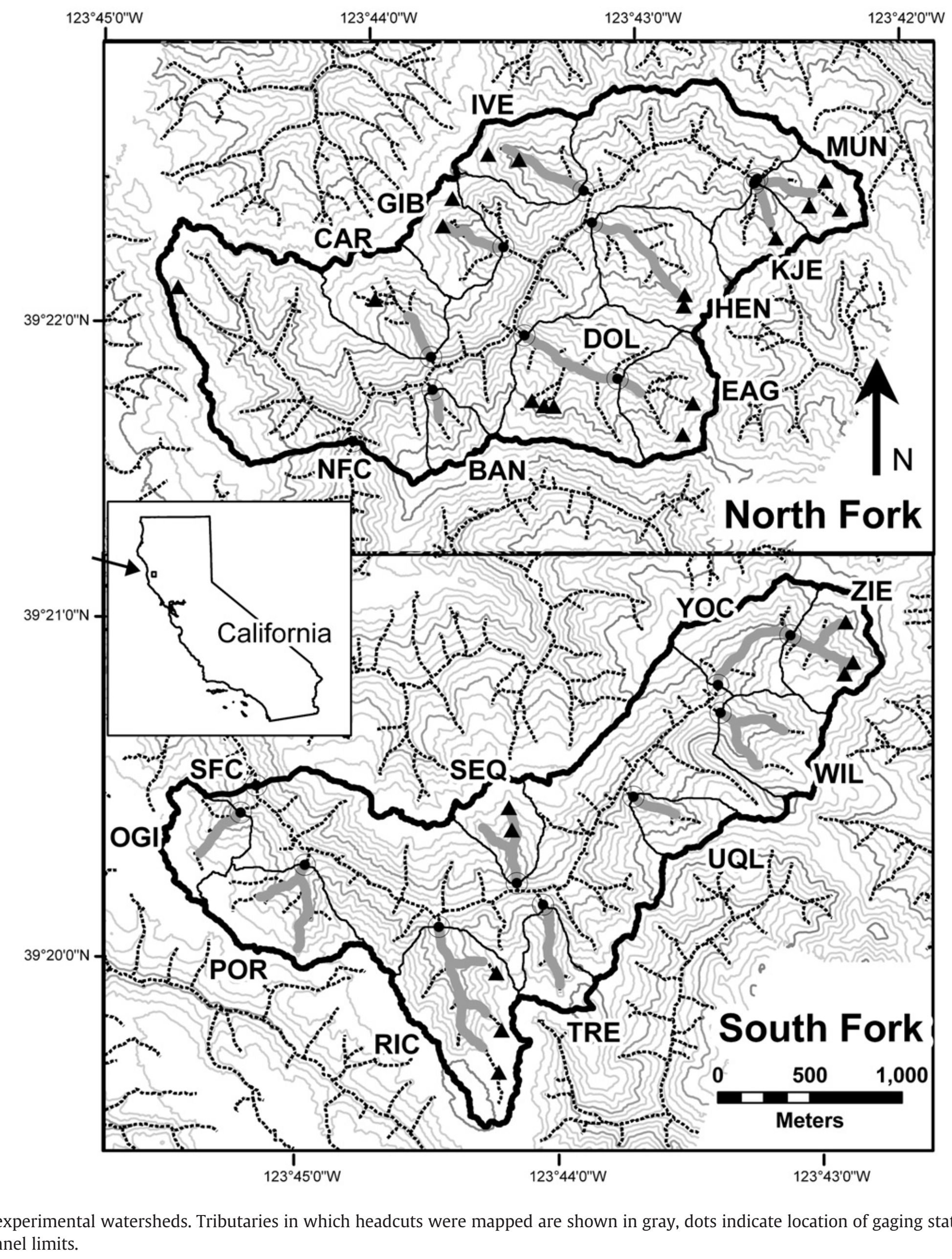

Fig. 1. The Caspar Creek experimental watersheds. Tributaries in which headcuts were mapped are shown in gray, dots indicate location of gaging stations, and triangles indicate locations of mapped channel limits.

Both experimental watersheds are densely forested with second- or third-growth stands dominated by coastal redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*). The entire study area was first logged between 1860 and 1904 using cleared-out channels as routes for log transport (Napolitano, 1998). The low incidence of old-growth wood in today's channels reflects this use, presenting a contrast to the high volumes of very large woody debris present in channels draining old-growth redwood forests of the region (e.g., Keller et al., 1995).

The Caspar Creek Experimental Watersheds were established in 1961 as a long-term research site run jointly by the California Division of Forestry and Fire Protection and the U.S. Forest Service (USFS) Pacific Southwest Research Station. Weirs and gaging stations were constructed by November 1962 on the North and South Forks. After an initial period of calibration between the two watersheds, roads were constructed in the South Fork in 1967, and nearly two-thirds of the stand volume was selectively logged from 1971 to 1973. Tractors commonly dragged logs down tributary valleys, once again severely disturbing some channels.

Twelve years after South Fork logging, thirteen additional gaging stations were constructed in tributaries and along the main stem of the North Fork. The new stations underwent calibration for 4 years. Thirty-seven percent of the North Fork watershed was then roaded, clearcut, and cable-yarded from ridgetops between 1989 and 1992. Selectively logged buffer strips were left around channels with drainage areas of greater than about 10 to 12 ha, and three subcatchments (HEN, IVE, and MUN, Fig. 1) were left as mature second-growth controls within the North Fork watershed. All watersheds but BAN and HEN included some roads constructed long before calibration began, and both new and old roads were located near ridges.

Both episodes of experimental logging resulted in increased runoff and sediment yield, followed by substantial recovery within a decade of logging (Ziemer, 1998). In the North Fork watersheds, increases in sediment production were correlated with increases in storm runoff (Lewis et al., 2001), which were attributable largely to reduced rainfall interception and transpiration after logging (Reid and Lewis, 2007). Ten new gaging stations were constructed on South Fork tributaries in 2000 in preparation for a third experiment.

Stands representing three broad categories are now present in the North and South Fork watersheds. In North Fork control watersheds, the overstory is dominated by 100- to 140-year-old redwoods and Douglas-firs with diameters at breast height (DBH) of about 70 to 150 cm. A subcanopy composed of hardwoods (principally tanoak, *Lithocarpus densiflora*) and small conifers is present at most sites, and shrubs, ferns and herbs provide a discontinuous ground cover. Total stand density for stems larger than 12.7 cm DBH is typically on the order of 500 stems/ha, with a basal area of about 100 $m^2\ ha^{-1}$ as of 1998. South Fork stands reflect the selection logging of the early 1970s, having a higher density of small trees than North Fork stands but including a similar range of stem sizes. South Fork stands typically have 600 to 1200 stems/ha and basal areas of 65 to 85 $m^2\ ha^{-1}$ as of 2009. The North Fork watersheds logged in 1989–1992 were clearcut, leaving only a disturbed and discontinuous cover of shrubs and herbs. Regrowth has been rapid, and herbicides were applied to watersheds EAG and GIB in 1994 and again in 1996 to control growth of shrubs and hardwoods, while logged watersheds BAN, CAR, and KJE were not treated. Logged watersheds were then thinned in 1998 (KJE) or 2001, leaving about 800 to 2000 stems/ha (basal area: 3 to 5 $m^2\ ha^{-1}$) and reducing crown area to about 20% of that present before thinning.

Most tributary segments with catchments larger than 1.9 ha show evidence of active incision characteristic of gullying, such as steep, raw banks, eroding headcuts, quasirectangular cross sections, and low width–depth ratios (Fig. 2). Old-growth roots often span channels or form headcut lips, and many old-growth stumps are now being undermined by gully-bank erosion and are toppling into channels. Much of the valley-axis area thus had supported trees for centuries, and gullies have expanded significantly since the trees were cut during first-cycle logging. Either previous gullies were limited in extent or earlier episodes of extensive gullying had stabilized by about 400 years ago. Many gullies in both confined reaches and valley flats excavate saprolite, also suggesting that the current extent of gullying is unprecedented.

Fig. 2. Headcut in MUN tributary, North Fork Caspar Creek. Note knapsack for scale; headcut is 1 m high.

At gully headcuts, the stream typically flows over a lip reinforced by a large root or piece of woody debris and scours a plungepool below. Several modes of headcut erosion are observed in the area: (i) gradual backwasting of the exposed face through ravel, spalling, and tractive erosion, (ii) sapping or tunnel erosion, with rapid retreat occurring when the upstream tunnel enlarges enough that a portion of the roof collapses, (iii) block failures induced by undercutting from backwasting, plungepool erosion, or sapping, and (iv) rapid tractive erosion when a reinforced lip loses its influence, causing a sudden drop in base level to the upstream reach. Most headcuts and associated plungepools show complex forms modified by woody debris, roots, bedrock, boulders, and in-place or toppled bank vegetation; few exhibit the regular morphology typical of grassland gullies. Little vegetation other than moss is present in growth position on gully floors.

3. Methods

The study relies on four kinds of data: (i) measurements of gully distribution and characteristics recorded during field surveys carried out during 2000–2002 and 2006–2008; (ii) measurements of process rates monitored within a subset of the gullies; (iii) sediment gaging records from gullied tributaries; and (iv) information from earlier studies of surface erosion and landslide distribution in the area.

3.1. Gully distribution and dimensions

Channel characteristics were mapped along the main axes of 16 tributaries from near the mouth of the channel to a point above which the drainageway is no longer primarily in the form of an active channel; this aspect of the study is described in more detail by Dewey (2007). Above most of the upstream mapping points in North Fork control channels and South Fork channels are 1 to 4 ha of swale in which either unchanneled reaches predominate or most of the channel is inactive and filled with duff. Additional pipe collapses and discontinuous gullies occur upstream of mapped reaches, and mapped reaches include occasional unincised zones. During this phase of the study, the upstream end of the channel could not be readily found in some clearcut watersheds where channels were obscured by dense regrowth and logging debris. Hillslope gullies and subsidiary tributaries generally were not mapped, but both branches of the main tributary channel were mapped if the dominant fork could not be identified.

In North Fork tributaries, headcut locations and channel width and depth measurements were added to a preexisting channel map. In the South Fork, channels had not previously been mapped, so tributary thalwegs were mapped relative to tapelines established from surveyed benchmarks. Tapeline slope was calculated from surveyed endpoint locations or measured with a hand level; and a stadia rod was used to measure bank height, channel width, and thalweg elevation relative to the tapeline.

Bank-to-bank width and bank-to-thalweg depth were measured at 1074 locations along 3290 m of valley axis in eight North Fork tributaries and at 2124 locations along 5670 m in eight South Fork tributaries. Depth is reported as an average of measurements taken relative to the left and right banks, and depth and width are interpolated linearly between measurement points to allow estimation of average values for 25- and 50-m channel segments. In North Fork tributaries, the product of depth, width, and channel increment length is summed by 2-m channel pixels to estimate channel-segment volumes. South Fork volumes are estimated by summing values for measurement-bounded channel increments. Measurements from 18 surveyed cross sections indicate that the product of thalweg depth and bankfull width overestimates cross-sectional area in these tributaries by a factor of 1.40 (95% confidence interval: ± 0.09), with no significant dependence on width-to-depth ratio, and this value is used as a correction factor for estimating volumes.

Each mapped channel contains many vertical or near-vertical steps in its long profile. These features were classified as headcuts for this study if the step face included some component of material other than woody debris or bedrock. The location and height (relative to the plungepool nadir) of almost all headcuts higher than 0.3 m were recorded, but only headcuts higher than 0.44 m are considered in this analysis in order to ensure a consistent resolution between tributaries. The distribution of recorded heights suggests that 0.26- to 0.44-m headcuts would account for 15 to 20% of the headcuts higher than 0.26 m, or about 7 to 10% of the total headcut height, so exclusion of these features from the data set is not expected to substantially influence conclusions. A 2-m digital elevation model (DEM) was used to estimate drainage area and elevation at each channel pixel in the North Fork, while results from the South Fork are based on information from a 10-m DEM.

To evaluate the extent of the channel network, upstream channel limits were subsequently mapped in 10 forested tributaries and 7 clearcut tributaries in North Fork watersheds and in 8 South Fork catchments (Fig. 1). Because channels are discontinuous in their upstream reaches, the limit was defined for this portion of the study as the upper boundary of the upstream-most channel segment longer than 5 m that has well-defined banks and appears to have carried flow within the previous decade.

We evaluate the distribution of incised channels and associated headcuts by testing for patterns of association between headcut characteristics or channel form descriptors and such factors as drainage area, stream power, and treatment category. Because channel characteristics vary by position in the watershed and because different sections of each watershed were logged or mapped (Table 1), long-profile variations in channel and headcut characteristics must be accounted for if differences in characteristics between treatment groups (i.e., control, recently clearcut North Fork, and selectively logged South Fork watersheds) are to be validly identified. To do so, each channel or headcut characteristic identified for 25- or 50-m reaches within a treatment group was first regressed individually against catchment area, channel gradient, and a stream power index (the product of catchment area and segment gradient, as defined by Montgomery and Dietrich, 1989). If a significant relationship to one of these variables was found, multiple regression was used to determine if treatment is an additional significant variable, with significance assessed at the 0.05 level throughout the sequence of analyses. Comparisons are restricted to data from

Table 1
Mapped tributaries at Caspar Creek; channel attributes are reported here only for the mapped portion of the channel network.

	Gaged area (ha)	Percent logged	Percent roads[a] 1986	1995	Gage record	Area above survey (ha)	Total channel length (m)[b]	Buffer strip length (m)	Surveyed channel length (m)	Surveyed channel volume (m^3)	Surveyed bank area (m^2)	Headcuts >0.44 m
North Fork clearcut 1989–91												
BAN	10	95	0.0	3.5	1985–1995	5.1	875	88	157	63	106	19
CAR	26	96	4.7	6.4	1985–now	14.2	2343	242	284	611	646	15
DOL[c]	77	36	–	–	1985–now	26.6	–	n.a.	586	1243	1174	37
EAG	27	100	0.8	4.6	1985–now	8.9	2406	105	149	339	383	6
GIB	20	100	5.2	7.8	1985–1995	1.2	2115	107	360	312	427	41
KJE	15	97	0.3	6.3	1985–1995	3.7	1179	235	253	1511	880	14
North Fork control												
HEN	39	0	0.1	0.1	1985–now	3.7	3341	n.a.	675	640	821	78
IVE	21	0	7.6	7.6	1985–now	3.4	948	n.a.	494	512	625	53
MUN	16	0	5.2	5.2	1985–1995	6.9	1293	n.a.	330	633	467	21
South Fork selectively cut 1971–73												
OGI[d]	18	50	–	–	2000–now	5.2	–	n.a.	350	337	495	18
POR	32	54	–	–	2000–now	1.7	–	n.a.	780	1207	1135	41
RIC	49	67	–	–	2000–now	1.3	–	n.a.	1144	1356	1477	75
SEQ	17	67	–	–	2000–now	1.2	–	n.a.	668	845	810	45
TRE	14	67	–	–	2000–now	1.8	–	n.a.	489	464	665	33
UQL	13	67	–	–	2000–now	3.4	–	n.a.	269	258	316	42
WIL	26	67	–	–	2000–now	3.5	–	n.a.	699	687	867	94
YOC[c]	53	67	–	–	2000–now	25	–	n.a.	575	645	750	41
ZIE	25	67	–	–	2000–now	1.4	–	n.a.	694	458	706	57

[a] Percent of area in roads and landings; values are approximate due to potential instability of drainage patterns on ridge-top roads.
[b] Includes all sub-tributaries, as estimated from DEMs on the basis of expected headwater catchment area; value listed for DOL is exclusive of length in EAG.
[c] DOL is located downstream of EAG, and YOC is downstream of ZIE; surveyed values listed for DOL and YOC pertain only to the segment of the watershed below the upstream gage.
[d] OGI includes 27% private land that has been selectively logged several times since 1971.

watershed areas of 3 to 30 ha to ensure that samples are available from similar locations in both logged and control watersheds.

3.2. Gully erosion rates

Tributaries YOC, ZIE, and MUN (Fig. 1) were selected for monitoring of headcut changes over a multi-year period using one of three methods. Five headcuts (group I) were surveyed in detail with an electronic total station in the summers of 2000 and 2002. The resulting maps are capable of revealing local changes of more than about 5 cm. At six other headcuts (group II), transects were surveyed first by survey laser and later by rod and tape to measure headcut retreat and bank erosion upstream and downstream of the headcut; survey error is estimated to be about 2 cm. An additional 41 headcuts (group III) were visually checked for obvious retreat, with changes >30 cm expected to be visible. Headcuts were revisited during the summer of 2003 and in February 2004, and additional measurements were made at three rapidly eroding headcuts. Nine headcuts from groups I and II at which benchmarks could be relocated were again surveyed in 2006 and 2008 using an electronic total station. Cross-section measurements from groups I and II also allowed monitoring of bank erosion on 15 near-vertical banks between the summers of 2000 and 2002; 12 of the 15 measurements are from banks within 4 m downstream of a headcut. Flow conditions over the monitoring period ranged widely: the highest flow recorded at IVE during 24 years of monitoring occurred in December 2005, while all other years produced maximum flows with recurrence intervals of less than 2 years.

During the autumn of 2000, paint was applied to roots at the point at which each emerged from a headcut face to allow long-term monitoring of bank recession at several headcuts. Distances between the wall surface and the painted portions of roots were measured at 43 points on four headcut overfalls during July 2008.

In calculations of erosion rates, a bulk density of 1.5 Mg m^{-3} is assumed when converting rates from units of m^3 y^{-1} to Mg y^{-1}.

3.3. Suspended sediment transport at gaging stations

We use records from nine tributary gaging stations in the North Fork watershed (Fig. 1; Table 1) to evaluate variations in annual suspended sediment yield as they may relate to gullying; results of the suspended sediment study evaluated by storm period are reported elsewhere (Lewis, 1998; Lewis et al., 2001). Automatic sediment samplers at each gage are programmed to sample at predetermined turbidity levels. These sediment measurements allow ongoing recalibration of the turbidity record at each gage in order to produce a continuous estimate of suspended sediment concentration during storms (Lewis, 1996), and transport rates are then computed from discharge and concentration. Measured storms account for 90 to 99% of sediment output in most years (Jack Lewis, USFS, personal communication, 2007), with the highest percentages for years with large flows, so the approach is expected to account for at least 95% of the suspended load during the period analyzed.

For the current study, analysis requires annual loads rather than storm-period loads, so storm loads had to be estimated at several stations during periods with missing record. Loads at individual stations were estimated for missing portions of the pre-logging record using correlations between peakflow and storm sediment load or between loads at nearby gages if peakflow data were missing. Regressions are not expected to be stable during the post-logging period, so missing values after logging were estimated using ratios between the target storm load and temporally adjacent storm loads at stations with records for the missing storm. Imputed values account for about 4% of the total suspended sediment load.

3.4. Other erosion studies

The incidence of landslides and treethrows mobilizing more than 7.5 m^3 has been recorded in the North Fork watershed since 1986 through “large-event surveys” carried out during post-storm channel inspections. Each event is located on a channel map with reference to existing benchmarks, and the scar is diagramed and dimensions measured. Associations with roads or treethrows are noted if present, and the volume of displaced sediment still present on the scar is estimated.

Surficial erosion has also been evaluated in the area. Rice (1996) reported results of void measurements on erosion plots at 175 hillslope sites and 129 road sites distributed randomly through the North Fork watershed after completion of second-cycle logging. Sample units on logged and unlogged hillslopes were circular 0.08-ha plots, while those on roads were 1.5-m-wide segments of the road prism oriented perpendicular to the road centerline. Processes assessed on plots include rilling, sheet erosion, and soil displacement from yarding. Rice weighted erosion plot data according to the proportion of the sub-watershed logged and the road length present in each and reported results as average total erosion per unit area of each sub-watershed.

Rice's results require reevaluation for the current study because several sub-watersheds had experienced different durations since logging at the time of the survey (1995). Given the large reported differences in erosion between logged and unlogged sites (Rice, 1996), most soil displacement appears to be associated with treatment, so Rice's estimates for rates of road and hillslope surface erosion are here recalculated as erosion per year since the onset of construction of new logging roads in the treated watersheds. Rates for the control watersheds are also calculated assuming a 5-year period of visibility rather than the 10-year period implicitly assumed by Rice (1996).

For the calibration period, hillslope surface erosion is estimated simply as the mean of the rates calculated for the post-calibration period in the three control watersheds. This approach cannot be adopted to estimate road-related erosion during the calibration period because road erosion for the post-calibration period in both logged and control watersheds reflects the period of high-intensity use during logging. Results from Reid and Dunne (1984) suggest rates of surface erosion on gravel-surfaced logging roads are about 75 times higher during high-intensity use periods than during periods of light use. Log hauling typically took place over a 4-month period within each logged watershed, and mainline roads in all watersheds would have undergone longer periods of heavy use as neighboring units were logged. Road erosion rates are assumed to be 75 times higher than “background” road erosion rates for a 4-month period during the post-calibration period, and the corresponding rate for light-use periods is then calculated algebraically from the road erosion plot data (i.e., $plot\ total = (0.33\ year \times 75x) + [(post\text{-}calibration\ duration - 0.33\ year) \times x]$, where x = rate for the light-use period). The mean of estimated light-use rates calculated per unit area of road is then applied to the area of roads present during the calibration period in each watershed.

4. Results

4.1. Gully distribution and characteristics

Gully headcuts are common along all mapped valley axes (Fig. 3, Table 1), with some present in catchments of <1 ha. There are 284 mapped headcuts higher than 0.44 m in North Fork tributaries and 446 in South Fork tributaries, indicating average frequencies of 8.6 and 7.9 headcuts per 100 m of channel, respectively. Differences in gully characteristics between logged and control watersheds are not evident to casual observation (e.g., Fig. 3), indicating that any major

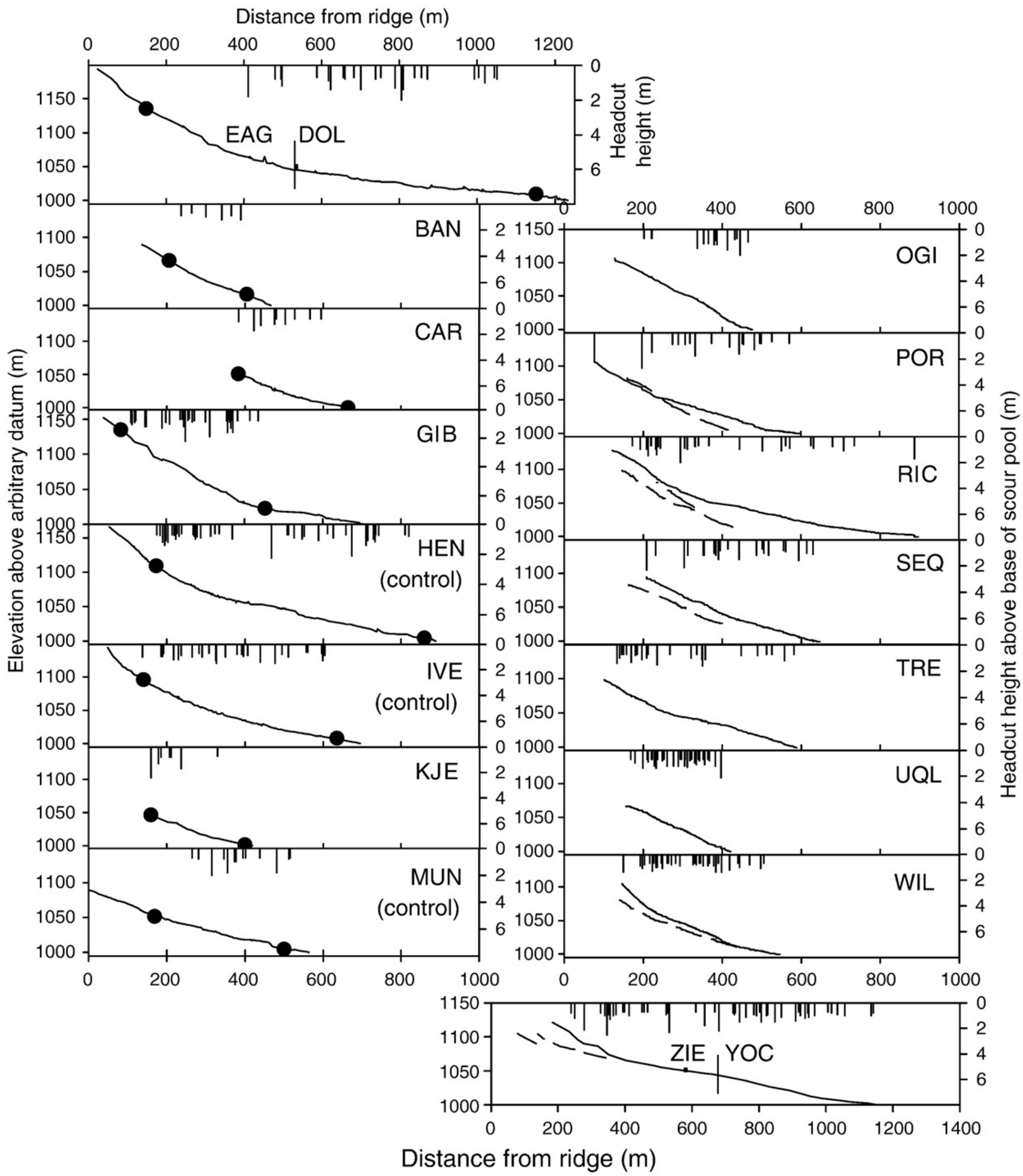

Fig. 3. Surveyed valley-axis profiles in gaged tributaries. Tics indicate the locations and heights of headcuts along the main tributary channel, and dashed lines show profiles of additional mapped branch tributaries. Dots on North Fork profiles indicate the limits of headcut mapping. For clarity, mapped headcuts shorter than 0.64 m are omitted. See Table 1 for watershed descriptions.

incision episode predated second-cycle logging. In particular, the logged KJE catchment was noted to have exhibited particularly well-developed gullies even before the recent logging.

Headcut characteristics are assessed for 50-m channel reaches; analysis of shorter reaches would provide too few headcuts in each sample unit. Headcut height (h) in North Fork tributaries increases weakly but significantly with channel-segment gradient (s) ($\log h = -0.34 + 0.58s$, $p<0.001$, $r^2 = 0.064$), and the relation shows no significant difference between logged and control tributaries. In contrast, mean headcut spacing decreases significantly with increasing gradient (Fig. 4), and headcuts in control watersheds are spaced significantly more closely than in logged watersheds. Although variance is high, control watersheds show a tendency for the proportion of channel elevation drop accounted for by headcuts to increase with increasing catchment area (Fig. 4C), while logged watersheds show no significant increase (Fig. 4D). For channel segments with catchment areas larger than 10 ha, control watersheds show the mean proportion of elevation drop from headcuts to be 0.53 ± 0.09, compared to 0.26 ± 0.09 in logged watersheds.

Sample frequencies allow assessment of channel characteristics in reaches shorter than those required for headcut assessments. Comparisons of morphological attributes in 25-m channel reaches in control and logged watersheds show patterns similar to that of Fig. 4D: the systematic variations in width, depth, and cross-sectional

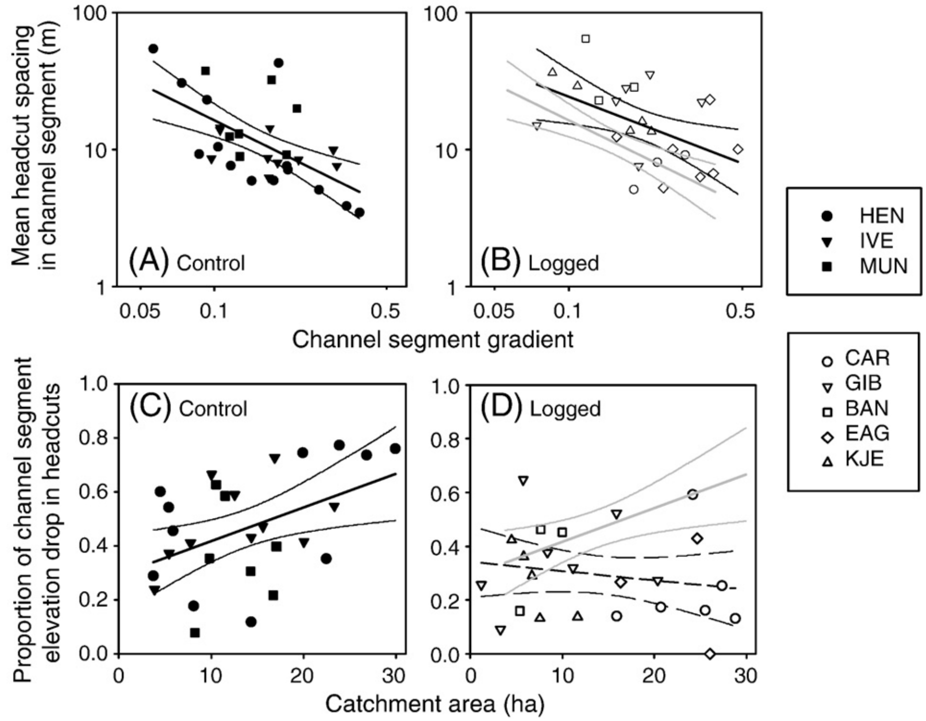

Fig. 4. Headcut spacing as a function of channel gradient in 50-m reaches of North Fork (A) control ($r^2 = 0.36$) and (B) logged ($r^2 = 0.23$) tributaries; and relative importance of elevation drop from headcuts in (C) control ($r^2 = 0.20$) and (D) logged ($r^2 = 0.03$) tributaries of North Fork Caspar Creek. The gray lines in Fig. 4B and D depict the regression and 95% confidence band for controls. Dashed lines indicate relations not significant at the 0.05 level.

area with stream power index expressed in control tributaries are not evident in recently logged tributaries (Fig. 5A). Relations between these attributes and watershed area, however, are strongly expressed in the pervasively gullied KJE tributary (Fig. 5B), in which gullies appear to have coalesced before monitoring began and which exhibits no significant relations to stream power index.

Results for South Fork tributaries, which were selectively logged in the early 1970s, are generally intermediate to those in North Fork logged and control watersheds. Headcut spacing in 50-m reaches and the proportional elevation drop accounted for by headcuts are most similar to those in logged North Fork watersheds (Fig. 6). Morphological data from South Fork tributaries generally show higher variance than found for either logged or control North Fork channels, which may in part result from the use of a l0-m DEM for the analysis rather than the 2-m DEM used for North Fork tributaries.

Mapping of upstream channel limits in North Fork control watersheds shows a mean stream power index of 0.69 ± 0.12 ha at the channel head (Fig. 7, Table 2). Results for the South Fork watershed (0.61 ± 0.09 ha) are not significantly different from those for North Fork control watersheds, while values for clearcut North Fork watersheds (0.34 ± 0.10 ha) are significantly lower than those in either control watersheds or South Fork tributaries. The catchment area at the head of forested channels averages 1.9 ± 0.3 ha, compared to 1.2 ± 0.5 ha for logged tributaries, representing a 28% increase in drainage density within 30-ha watersheds.

4.2. Rates of erosion at headcuts and streambanks

Measured rates of headcut retreat are highly skewed in part because of differences between activity levels at headcuts held in place by roots or woody debris and those not impeded by such features. Of the 52 headcuts observed over a four-year period in MUN, YOC, and ZIE tributaries, three migrated rapidly (>30 cm y^{-1}) during at least one year, for a total of 4.8 m during four headcut-years. The other 204 headcut-years produced no rapid retreat. The resulting mean rate of rapid retreat for all 52 monitored headcuts is 4.8 m for 208 headcut-years, or 2.3 cm y^{-1} per headcut.

For the subset of 11 headcuts that were surveyed in more detail between 2000 and 2002 or 2008, displacement totaled 158 cm over 64 headcut-years that exhibited no rapid retreat, producing a mean retreat of 2.5 cm y^{-1}. Individual retreat rates ranged from 0 to 30 cm y^{-1}, with two headcuts showing no retreat over the 8-year period. Recession monitored around painted roots over an 8-year period indicated an average retreat of 0.7 cm y^{-1} for 32 headcut-years. The combined estimate for gradual retreat is thus 1.9 cm y^{-1}, and the estimated total retreat rate is 4.2 cm y^{-1} with an expected accuracy of about ±50%. Data are too few and variance too high to isolate the effect of the 25-year recurrence interval storm of December 2005. Field observations suggest that many of the changes in headcut location noted between 2002 and 2006 may have resulted from the storm, but that the storm did not cause widespread or severe channel disruption.

As is the case for headcuts, banks show a skewed distribution of rates. Two monitored banks eroded rapidly (6 and 8 cm y^{-1}), while eight showed rates of <1 cm/y; retreat averaged 1.6 ± 1.2 cm y^{-1}. Erosion generally occurred either by block failure–often associated with undercutting–or by more widespread but gradual backwasting and spalling.

4.3. Suspended sediment yields

Lewis (1998) demonstrated that, with the exception of KJE, storm-based sediment yields increased significantly above expected levels after logging; these patterns are evaluated in more detail by Lewis et al. (2001). For the present study, we evaluate annual yields to assess the relative importance of individual watersheds as sediment sources. Mean annual suspended sediment yields at North Fork tributary gaging stations range between 2 Mg km^{-2} y^{-1} at watershed BAN before logging and 68 Mg km^{-2} y^{-1} at KJE after logging (Table 3).

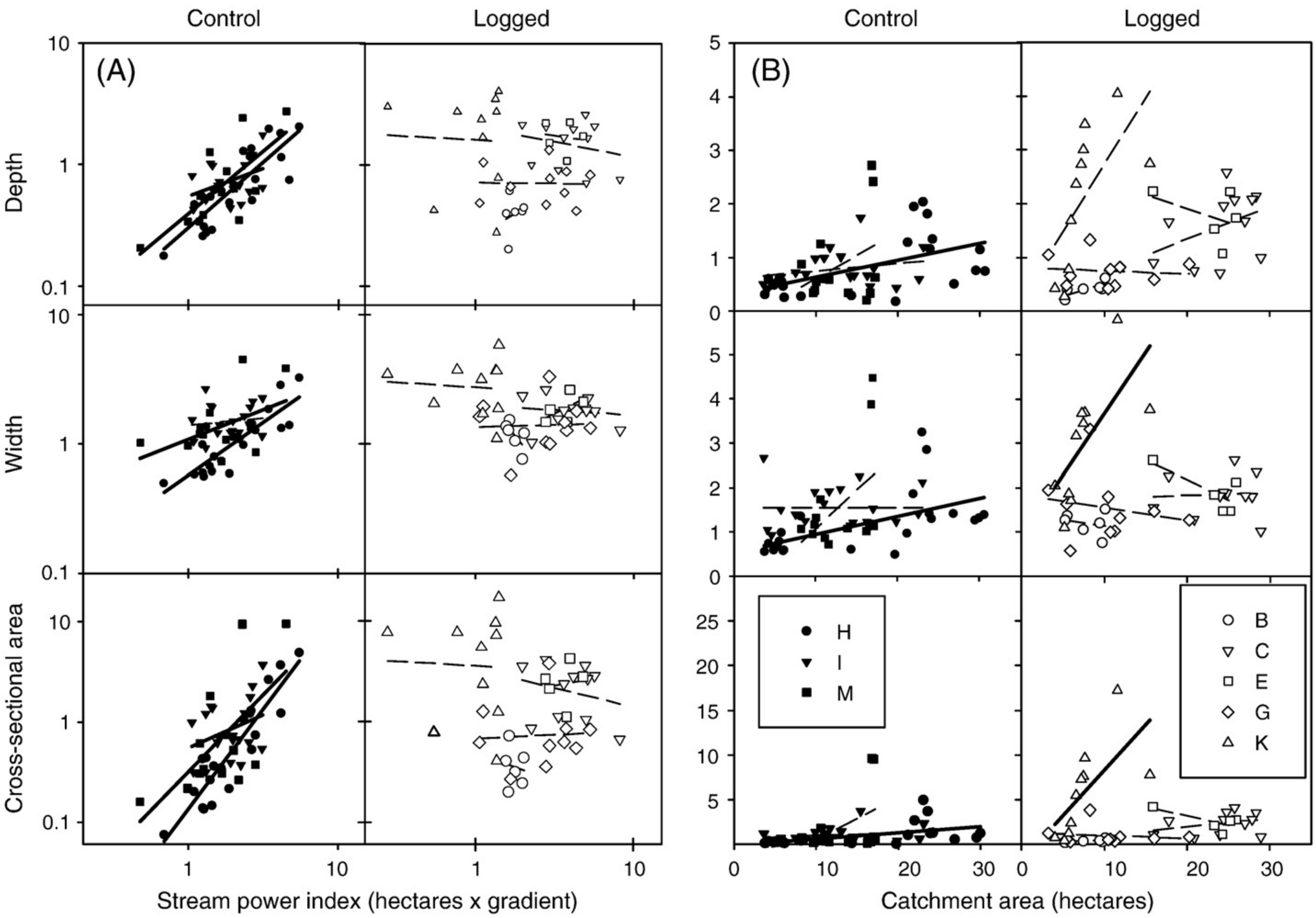

Fig. 5. Variation in mean segment depth (m), width (m), and cross-sectional area (m^2) with (A) stream power index and (B) catchment area in control and logged tributaries of North Fork Caspar Creek. Relations significant at the 0.05 level are depicted by solid lines.

4.4. Rates of erosion from other sources

Plot erosion data reported by Rice (1996) and here recalculated to reflect the post-calibration period reveal strong differences between logged and control watersheds. Logged watersheds show road and hillslope soil displacement rates averaging 2600 ± 1170 Mg km^{-2} y^{-1}, with 6 to 33% of the individual watershed totals associated with road erosion. In control watersheds, displacement averages about 8% of that calculated for the logged watersheds over the analogous period (Table 3), and 0% to 76% is associated with road erosion. Erosion on hillslope plots in control watersheds averages 67 ± 61 Mg km^{-2} y^{-1}. These calculations assume that all erosion voids are visible for 5 years in control watersheds. If the period of visibility is actually longer, as assumed by Rice (1996), the difference between displacements in logged and control watersheds would be accordingly greater. Hillslope and road erosion rates for the calibration period are highly uncertain. Differences in estimated displacement from roads during the calibration and post-calibration periods reflect the increase in road area in the treated watersheds and the expected influence of heavy road use in both treated and control watersheds. Overall, expected road sediment after the onset of road construction increased on average by factors of 16 in treated watersheds and 6 in controls.

Large-event surveys in the gaged watersheds disclosed 8 events capable of contributing sediment to streams during the calibration period and 25 in the post-calibration period. Mapped events were associated with major storms in 1986 (5 events), 1990 (4 events in two storms), 1993 (7 events), and 1995 (17 events in two storms). Field-based estimates for each of these events indicate that an average of about 25% of the sediment displaced was transported off-site. Annualized average volumes of off-site displacement from large events during the post-calibration period (Table 3) are not significantly different at the 0.05 level between treated watersheds (19 ± 14 Mg km^{-2} y^{-1}) and controls (26 ± 2 Mg km^{-2} y^{-1}).

5. Interpretation and discussion

Results of the field study are here interpreted to address three questions: (i) What is the relative importance of gully erosion between logged and forested tributary watersheds at Caspar Creek? (ii) How important is gully erosion relative to other sediment sources in the area? (iii) What mechanisms might contribute to differences in the extent and importance of gully erosion between logged and unlogged watersheds?

5.1. Sediment production from gully erosion in logged and forested tributaries

Once the distribution of channel characteristics and their associations with land-management categories are identified, it is possible to extrapolate information from the sampled portions of a watershed to the watershed as a whole. Measured headcut and bank erosion rates can then be applied to the channel network expected in each setting to allow comparison of average sediment production rates from gully sources typical of forested North Fork watersheds, recently clearcut

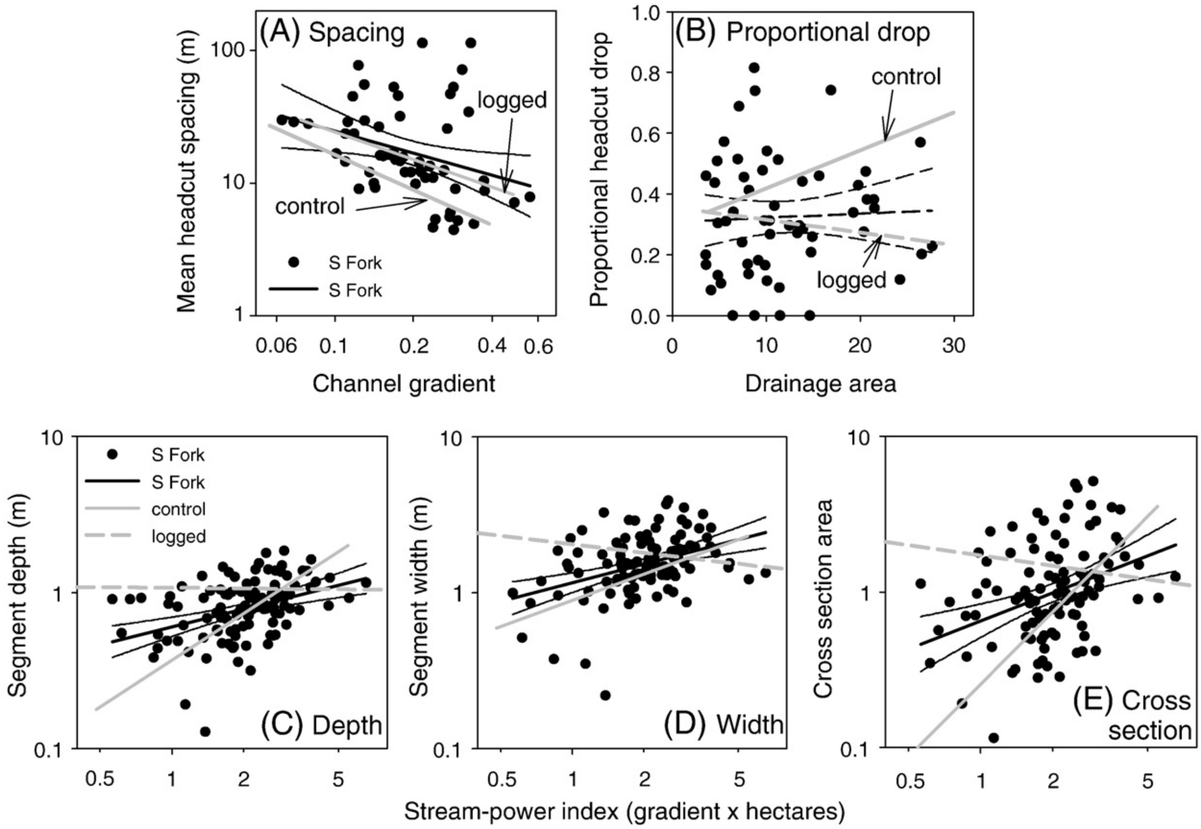

Fig. 6. Patterns of headcut distribution in South Fork tributary channels, and morphological characteristics of South Fork channels. Relations shown in gray depict analogous relations for North Fork tributaries (pooled regressions from Figs. 4 and 5), and relations significant at the 0.05 level are depicted by solid lines.

North Fork watersheds, and selectively logged South Fork watersheds. Calculations of sediment production from bank and headcut erosion are based on sparse data from a limited period, so estimates are not reliable. However, estimated sediment input rates are useful for indicating the potential magnitude of sediment contribution from the gullies and for assessing relative contributions among the three watershed treatment categories.

Sediment production by retreat of headcuts higher than 0.44 m in control watersheds can be estimated if we assume (i) the area of each eroding headcut face is equivalent to the product of the headcut height (relative to the base of the plungepool) and the channel width measured immediately downstream; (ii) the calculated average retreat rate of 4.2 cm y^{-1} for monitored headcuts applies to all headcuts higher than 0.44 m; and (iii) headcut frequency and dimensions in unmapped channels are similar to those in mapped reaches of similar size. Estimated sediment input from this source is then 28 Mg km^{-2} y^{-1} in North Fork control tributaries smaller than 30 ha (Table 2), with 19% of that sediment originating in channels draining areas of 1.9 to 3 ha, which were not characterized in control watersheds during the gully survey. Because headcut face areas are calculated as the product of channel width and the face height above the base of the plungepool, plungepool erosion is implicitly included as a component of headcut erosion.

A portion of the sediment eroded from headcuts and plungepools is deposited immediately downstream on the depositional downstream lip of the plungepool. Lip aggradation is estimated to account

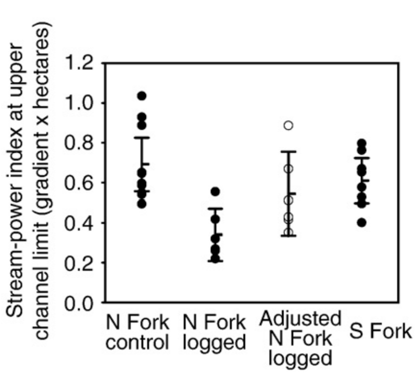

Fig. 7. Distribution of stream power indices at the upper limits of tributary channels. Error bars indicate the 95% confidence interval around the means (horizontal lines). "Adjusted" values account for hydrologic change after North Fork logging (see text Section 5.3.1).

Table 2
Comparison of channel characteristics for three watershed categories; dimensions represent means for 25-m channel segments; 95% confidence intervals are indicated for means.

Attribute	North Fork		South Fork
	Control	Clearcut[a]	
Catchment area at channel head (ha)	1.9 ± 0.3	1.2 ± 0.5	2.3 ± 1.0
Stream power index at channel head (ha)	0.69 ± 0.12	0.34 ± 0.10	0.61 ± 0.09
Channel width: mean at 3 to 7 ha (m)	0.97 ± 0.34	1.31 ± 0.38	1.37 ± 0.26
Channel width: mean at 15 to 25 ha (m)	1.83 ± 0.48	1.76 ± 0.25	2.16 ± 0.38
Channel depth: mean at 3 to 7 ha (m)	0.49 ± 0.07	0.53 ± 0.23	0.72 ± 0.11
Channel depth: mean at 15 to 25 ha (m)	1.16 ± 0.35	1.35 ± 0.40	1.02 ± 0.18
Estimated headcut erosion (Mg km^{-2} y^{-1})	28	50	37
Estimated plungepool deposition (Mg km^{-2} y^{-1})	7	12	9
Estimated bank erosion (Mg km^{-2} y^{-1})	54	73	66
1992–95 average suspended load (Mg km^{-2} y^{-1})	25 ± 17	39 ± 20	-

[a] Exclusive of watershed KJE.

Table 3
Estimated rates of sediment displacement and suspended sediment yields in North Fork sub-watersheds.

Site	Area (ha)	Years in category[a]	Suspended sediment yield	Road plot erosion[b]	Hillslope plot erosion[b]	Landslides and treethrows[c]	Gully erosion[d]
			($Mg\ km^{-2}\ y^{-1}$)				
Treatment watersheds, calibration period[a]							
BAN	10	6.1	2	0	67	0	52
CAR	26	6.1	6	44	67	0	90
EAG	27	4.7	8	6	67	3	68
GIB	20	4.9	9	58	67	27	82
KJE	15	3.8	67	5	67	0	136
Treatment watersheds, post-calibration period[a]							
BAN	10	3.9	9	330	1490	0	78
CAR	26	3.9	29	440	690	37	111
EAG	27	5.3	54	190	4290	31	90
GIB	20	5.1	29	510	2280	6	100
KJE	15	6.2	68	460	1680	20	142
Control watersheds, calibration period[a]							
HEN	39	5	14	0	67	11	82
IVE	21	5	9	44	67	1	74
MUN	16	5	21	6	67	0	77
Control watersheds, post-calibration period[a]							
HEN	39	5	30	0	33	24	82
IVE	21	5	7	1290	130	26	74
MUN	16	5	23	35	39	27	77

[a] "Calibration period" ends with the onset of new road construction at each logged watershed and after 5 years at each control watershed.
[b] Based on hillslope and road erosion plot measurements (Rice, 1996); values represent total amounts displaced.
[c] Based on landslides and treethrows mapped during channel surveys; values represent amounts transported off-site.
[d] Based on application of average bank and headcut erosion rate to the areas of bank and headcut face expected to be present in the watershed (see text); values represent amounts transported beyond the plungepool lip.

for approximately 25% of the combined headcut and plungepool erosion, as plungepool depths below the residual pool surface (i.e., below the depositional lip) average about 25% of the total headcut height on 23 headcuts mapped on the YOC-ZIE tributary in 2006.

We expect retreat rates for 0.26- to 0.44-m headcuts to be lower than 4.2 cm y^{-1} and rates to be negligible for faces shorter than 0.26 m. But even at a retreat rate of 4.2 cm y^{-1}, the expected population of 0.26- to 0.44-m headcuts in control tributaries would produce no more than 4 Mg km^{-2} y^{-1} of sediment if the assumptions listed above apply also to the small headcuts.

The monitoring results indicate an average bank erosion rate of 1.6 cm y^{-1}, a value similar to rates measured on vertical faces in a grassland gully in alluvial clay soils near Berkeley, CA (2 to 4 cm y^{-1}; Reid, 1989) and on near-vertical road-cut faces in a western Washington forest (1.6 cm y^{-1}; Reid, 1981), suggesting that the measured values are reasonable for the observed conditions. To estimate total sediment input from bank erosion in control watersheds, we assume (i) the proportion of banks "susceptible to erosion" is represented as the proportion mapped during a 1995 channel survey as either undercut or vertical and composed of either alluvium or colluvium; (ii) the estimated average gully-bank erosion rate of 1.6 cm y^{-1} applies to the entire area of bank susceptible to erosion; and (iii) the measured relations between channel depth, stream power index, and drainage area defined for 3- to 30-ha watersheds apply to the full estimated drainage density of channels draining catchments of 1.9 to 30 ha.

The 1995 maps of North Fork tributaries indicate that at that time approximately half of the channel length was susceptible to erosion, with no significant difference in values between control and logged tributaries. The relation for channel depth as a function of drainage area obtained by pooling the data shown in Fig. 5 for control watersheds was then used to estimate mean depth for each channel increment throughout a hypothetical 30-ha control watershed. The above assumptions produce an estimated annual input from bank erosion of about 54 Mg km^{-2} y^{-1} in the hypothetical watershed (Table 2).

About 9 Mg km^{-2} y^{-1} of the calculated amount, or 17%, is expected to be from unmapped channels with drainage areas of 1.9 to 3 ha and so is based on extrapolation of the defined relations to smaller watersheds. A second estimate for the unmapped low-order reaches may be made by assuming that channel form near the head of a clearcut tributary is similar to that near the head of a control tributary, despite the difference in drainage areas at the channel head. If the mean bank face height of 0.5 m calculated for a mapped 40-m reach draining 1.2 to 1.3 ha of clearcut is assumed also to represent that near forested channel heads, the input by bank erosion from 1.9- to 3-ha drainage areas in control watersheds is estimated to be 12 Mg km^{-2} y^{-1}, a value similar to that estimated using morphological extrapolations.

Similar calculations employing the same estimate of the proportion of erodible bank for channels characteristic of South Fork watersheds provide an estimated input of 37 Mg km^{-2} y^{-1} from headcut erosion and 66 Mg km^{-2} y^{-1} from bank erosion in a 30-ha watershed, values 32% and 22% higher than corresponding estimates for North Fork control conditions (Table 2).

If headcut and bank retreat rates measured in North Fork control channels and South Fork channels are assumed to apply also to North Fork logged channels (exclusive of KJE), such channels would supply expected inputs of 50 Mg km^{-2} y^{-1} from headcuts and 73 Mg km^{-2} y^{-1} from banks (Table 2), values 79% and 35% higher than corresponding estimates for control conditions. The high values for headcut inputs relative to those in controls largely reflect the increased headcut face area expected in the extended low-order channel network and are offset slightly by the increase in headcut spacing downstream. Higher input rates from bank erosion reflect increases in both channel length and depth. Possible increases in headcut and bank retreat rates after logging and in percentage of the bank susceptible to erosion were not considered, so these calculations may underestimate erosion from these sources in logged watersheds.

5.2. Contribution of gully erosion to the sediment yield

To assess the importance of gully erosion relative to other sediment sources, we first compare estimates of sediment production from gullies to those from other erosion processes. Such comparisons, though instructive, do not account for differences in sediment delivery to streams and through stream channels, so we then evaluate correlations between tributary sediment yields and potential controlling influences.

5.2.1. Comparison of gully erosion rates to those of other erosion processes

The importance of gullying to sediment production can be evaluated relative to rates and patterns of erosion from other sources if total gully erosion is estimated for the length of channel present in each gaged watershed of the North Fork catchment. This calculation differs from the previous one in that channel characteristics within individual watersheds are taken into account instead of using regressed relations to construct typical conditions in a hypothetical watershed.

Sediment input from channel-bank erosion is calculated as the product of eroding bank area and average bank erosion rate (1.6 ± 1.2 cm y^{-1}) and is converted to units of mass by applying an average bulk density for the characteristic soil (1.5 Mg m^{-3}). The area of eroding banks in unmapped portions of each tributary channel network is estimated by assuming (i) channel heads are located at a drainage area of 1.9 ha for pretreatment conditions and 1.2 for logged conditions; (ii) regressions between channel depth and drainage area (Fig. 5) apply also to unmapped channels within each category of watershed; and (iii) 50% of the bank length is susceptible to erosion. Unmapped reaches (primarily first-order channels) account for 48% to 94% of the estimated channel length in individual North Fork watersheds (Table 1) and for an average of about half the estimated eroding bank area. Inputs from headcut erosion are calculated by applying an average retreat rate (4.2 ± 2.1 cm y^{-1}) to the estimated area of headcut face present in each tributary, using assumptions analogous to those employed for estimating bank erosion. Deposition on plungepool lips is then estimated as 25% of the volume eroded by headcut retreat and is subtracted from the total for each watershed.

Comparison of the estimated sediment inputs from gully erosion after logging to displacement rates estimated from road erosion plot data shows displacements from road erosion to average about 5 times greater than those accounted for by gully erosion (Table 3), and those calculated from hillslope erosion plot data to be an order of magnitude greater. Displacement from gullying, in turn, exceeds displacement from treethrows and landslides by a factor of 6.

Comparison of displacement rates to measured suspended sediment yields requires an estimate of the proportion of the total sediment load that is transported in suspension at the tributary gages. Bedload was not measured, but sediment volumes and particle sizes accumulated in the North Fork weir pond suggest that coarse sand and gravel transported as bedload contribute about 20 to 30% of the total clastic load at the North Fork weir. Comparison of the proportion of the sediment load accounted for by gravel at the weir (13%) to the proportion of gravel present in the upper 1 m of the watershed's dominant soils (25 to 50%) indicates that a high proportion of the gravel has broken down to smaller sizes before reaching the weir, suggesting that the proportion of sediment transported as bedload is likely to be substantially higher in tributaries. We here assume that 40% of the tributary sediment load is transported as bedload, so we expect total sediment loads to be about 67% higher than the suspended sediment yields noted in Table 3.

Comparison of rates of plot erosion with estimated total sediment yields demonstrates that sediment delivery rates for surface erosion sources are quite low, necessarily averaging <3% for logged watersheds. Field observations indicate that much of the sediment displaced on hillslopes is quickly redeposited on hillslopes, in low-order swales, or on valley flats and so does not contribute directly to sediment yield over the short term. Estimated sediment displacement from gully erosion is also more than sufficient to account for the total sediment loads, suggesting that deposition within channels and on valley flats may significantly influence the amount and timing of sediment exports from the tributary watersheds.

About half of the sediment from treethrows and landslides in the post-calibration period was generated during storms in HY 1995 and so would influence only a single year of the suspended sediment record for the period; most of the rest originated in 1993. Mean suspended sediment load for HY1995 in the logged watersheds was 83 Mg km^{-2}, so estimated total load averaged about 140 Mg km^{-2}. In comparison, sediment displaced by large events amounted to 63 Mg km^{-2} that year, suggesting that large events would not have provided the principal influence on sediment yield even during a year with a high incidence of landsliding.

In addition, examination of annual yields at tributary gaging stations indicates that ranks in suspended sediment yield were relatively consistent from year to year between 1991 and 1995 (Fig. 8), suggesting that yields during this period were largely controlled by chronic sediment sources rather than by discrete, sediment-producing events such as landslides. The highest rankings for tributaries were not associated with years in which significant landslides were observed in those tributaries.

The relative importance of landslides is likely to be greater over the long term because of the rare occurrence of very large slides. Between 1986 and 2006, for example, total sediment displacement from landslides and near-stream treethrows throughout the North Fork watershed averaged over 200 Mg km^{-2} y^{-1}, and the two largest slides were responsible for about 50% of the displacement. Had the period of extensive suspended sediment monitoring (1986–1995) included one of these major landslides, landsliding would have undoubtedly influenced short-term sediment yields; but results here suggest that influences from the landslides and treethrows mapped during the study period are not expressed as acute short-term increases in yield. Both of the largest slides occurred on logged slopes and triggered debris flows, resulting in considerable volumes of deposition within and adjacent to downstream channels. Such deposits are likely to be remobilized later by channel incision, thus eventually contributing additional suspended sediment through gully erosion.

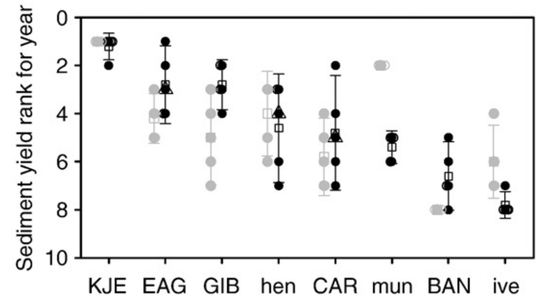

Fig. 8. Relative rankings in suspended sediment yield at tributary gages for HY1986–1990 (gray symbols) and HY1991–1995 (black). Open squares indicate mean rank for the period, and data points for years with significant landslides within individual watersheds are enclosed in triangles. Upper-case gage names indicate logged tributaries.

5.2.2. Correlation analysis

The relative importance of various sediment sources can be further evaluated by analyzing correlations between sediment displacement rates and suspended sediment loads in the gaged watersheds to identify the combination of source inputs that best explains the observed distribution of yields. Irrespective of sediment delivery ratios and grain size distributions, the relative distribution of suspended sediment yields among watersheds is expected to reflect the relative distributions of sediment displacements from the processes that most strongly influence sediment yields. For this analysis, accuracy of the estimated sediment inputs is not as important as the relative values between watersheds within a source type.

We carried out a step-wise multiple regression of mean suspended sediment yields (*SSY*, Mg km^{-2} y^{-1}) from calibration and treatment periods in each tributary watershed against estimated erosion from gullies (*G*, Mg km^{-2} y^{-1}), large events, road plot erosion, hillslope plot erosion (P_h, Mg km^{-2} y^{-1}), and watershed area to evaluate the relative influence of these factors on the distribution of suspended sediment yields. The resulting model (Fig. 9) employs only two variables:

$$SSY = -41.5 + 0.702\,G + 0.0050\,P_h \quad \text{adjusted}\,R^2 = 0.77 \qquad (1)$$

The relation is highly significant at the 0.05 level, with gully erosion by itself explaining 73% of the variance; the hillslope term is only marginally significant ($p = 0.05$). Results suggest that gullying and associated processes in logged watersheds contribute about three to seven times as much sediment to the sediment yield as processes assessed by hillslope erosion plots. In forested watersheds the difference is greater, with gullying responsible for about 50 to 200 times as much as hillslope erosion. Delivery of hillslope sediment to streams depends in part on the extent of the channel network, so the

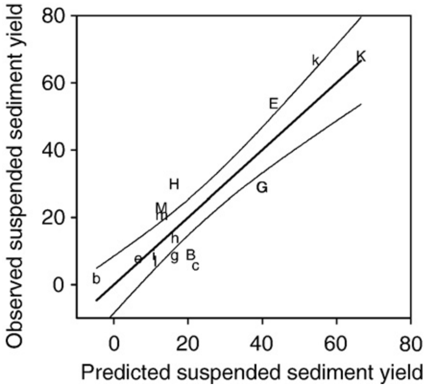

Fig. 9. Observed suspended sediment yields (Mg km^{-2} y^{-1}) versus those predicted using Eq. (1). Lower case initials represent data for the calibration period in each watershed. Points C and G are superimposed.

apparently overwhelming importance of gully erosion may actually reflect both in-channel erosion and the increased efficiency of hillslope sediment delivery due to network expansion after logging.

If bedload is assumed to account for 40% of the total sediment yields at the tributary gages, about half the estimated sediment eroded from banks and headcuts is not delivered to the downstream gage, suggesting that either the rate of gully erosion is overestimated or that significant deposition is occurring within channels and on floodplains. An average channel aggradation rate of 2 mm y^{-1} would account for the undelivered sediment, and if aggradation is indeed occurring at this rate it should become evident from cross-section measurements in the near future. However, given the uncertainties associated with the rate calculations, we expect that imprecision in estimated rates is likely to account for most of the discrepancy.

5.3. Mechanisms of influence from logging

Results of the study suggest that clearcut logging at Caspar Creek caused a significant increase in gully-related sediment inputs (Table 2). Such increases might reflect either increased runoff associated with logging or direct disturbance of low-order channels during cable-yarding operations. Several kinds of information suggest that hydrologic change is likely to be the dominant influence in the present case.

5.3.1. Evidence from network extent

Field measurements suggest that channel heads migrated upslope after logging (Fig. 7, Table 2) and that the stream power index associated with the channel head locations is significantly lower in logged watersheds than in controls. However, the utility of the stream power index (calculated as drainage area × gradient) as a measure of actual stream power relies on the assumption that drainage area is a valid index for relevant discharges. Once a site is logged, the relation between drainage area and characteristic discharges changes.

In North Fork watersheds, the mean peakflow for flows with recurrence intervals longer than 0.15 y increased by about 60% in the 2 years following logging (Reid and Lewis, 2007), with increases approaching an asymptote of 34% for flows occurring fewer than 3 times a year. Effective maximum stream power per unit drainage area during storms thus averaged 60% greater after logging, so a post-logging stream power index of 0.34 ha is equivalent to a pre-logging index of 0.54 ha. If this hydrologic shift is accounted for, the recalculated index at the channel head is not significantly different at the 0.05 level between logged and control watersheds (Fig. 7).

This finding may indicate either that the hydrologic change is sufficient to explain the shift in channel-head location or that the hydrologic change prevented healing of mechanically disrupted sites over the period following yarding. Low-order tributaries are most susceptible to influences from direct disturbance, and the prevalence of subsurface soil pipes in headwater channels would make these sites particularly sensitive to mechanical disruption. However, changes in channel characteristics in logged watersheds along the 65% of the mapped channel length protected by buffer strips could not be explained by direct disruption because disturbance within the buffers is minimal.

5.3.2. Evidence from nested gages

If hydrologic change is an important influence on in-channel erosion after logging, channels downstream of logged watersheds should show increases in sediment yield that cannot be explained simply by changes in sediment input from upstream.

A pair of nested stream gages provides the data needed to assess downstream variations in sediment yield in a gullied tributary. Gage DOL (with a 77-ha catchment) lies downstream of gage EAG (27 ha) in the North Fork watershed. The catchment above the EAG gage was logged in 1991, and hillslopes abut most of the channel length in the EAG catchment. In contrast, the catchment between the EAG and DOL gages has not been logged since 1904, and valley-fill terraces buffer much of the channel from hillslope inputs. Several small tributaries enter this reach, but alluvial fans at most tributary mouths appear to trap much of their coarse sediment load and some fine sediment. Visible sources of sediment within the reach include incised stream banks, headcuts, and near-stream treethrows.

Suspended sediment loads measured during storms at the EAG gage were subtracted from corresponding loads at the DOL gage to estimate the load derived from the unlogged portion of the watershed. Loads for the pre-logging period in both portions of the watershed were then regressed against the mean of loads at control watersheds HEN and IVE to allow prediction of expected loads. The ratios of observed to expected loads from the unlogged portion of the watershed show a response similar in initial timing and magnitude to that from the logged portion of the watershed (Fig. 10), though the downstream portion reattained pre-logging values more quickly than the logged portion. Sediment input increased again in the DOL reach after regrowth in the EAG watershed was thinned in 2001. The sediment record thus indicates that upstream logging influenced sediment production in unlogged portions of the downstream watershed where there was no direct disturbance, suggesting that hydrologic change may indeed have been influential.

However, it is also possible that a portion of the sediment appearing to have originated within the DOL reach may represent

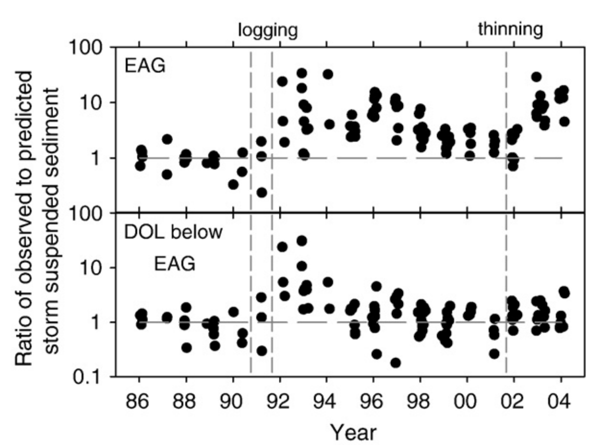

Fig. 10. Deviations from expected suspended sediment load in the clearcut EAG watershed and in the unlogged portion of the DOL watershed downstream of the EAG gage. Values are corrected for bias from back-transformation from logarithms (Baskerville, 1972).

breakdown of bedload into suspendible sizes: bedload originating in EAG would not be included as part of the EAG suspended load, but if it then becomes suspendible downstream it would be recorded as suspended load at the DOL gage. The potential magnitude of this effect can be estimated if bedload is assumed to represent 40% of the total sediment load at EAG. DOL produced 197 Mg of excess suspended sediment between 1991 and 1995, compared to 38 Mg from EAG, indicating that breakdown could explain no more than 13% of the post-logging sediment increase from DOL even if all EAG bedload were transformed to suspended load within the DOL channel. In actuality, much of the EAG bedload is trapped on a minimally channeled fan immediately downstream of the gaging station, so breakdown is likely to explain substantially less than 13% of the increase.

At a larger scale, Lewis et al. (2001) evaluated post-logging changes in suspended sediment yield across the full suite of nested North Fork stream gages and found that observed changes in suspended sediment yield after logging were more closely correlated with changes in flow than with other indices of management activity levels.

5.3.3. Morphological changes associated with headcut migration

Results of morphological comparisons for logged and control tributaries also are consistent with a situation in which channel morphology has been influenced by increased flows after logging. Contrasts in morphology between logged and control watersheds are revealed not so much by changes in average channel dimensions as by changes in the relations between channel dimensions and controlling variables such as stream power and drainage area (Fig. 5), and by increased spacing between headcuts after logging (Fig. 4).

The contrasting patterns may reflect an increase in the frequency with which headcuts are destabilized after logging. Newly mobilized headcuts can retreat rapidly until they encounter an upstream plungepool or a hardened lip supporting another headcut. Two discontinuous gullies will then have coalesced, thereby decreasing the number of headcuts present and increasing mean headcut spacing while not strongly affecting headcut height. Mean channel cross-sectional area would increase in areas where gullies coalesce. Reaches experiencing large changes in any particular year are likely to be distributed randomly, as the ability for a headcut to retreat rapidly is strongly influenced by the condition of elements armoring its lip. Overall patterns of distribution for morphological characteristics are thus likely to become less regular as the system continues to adjust to the sequence of changing hydrologic conditions.

Increased coalescence of headcuts in logged tributaries is also consistent with observed differences in the relative importance of stream power and watershed area as predictors of morphological characteristics in control watersheds and in the pervasively gullied watershed KJE (Fig. 5). In general, where gullies are present as a series of headcuts, headcut location–and the resulting distribution of incised reaches–might strongly reflect local conditions. For a channel segment with uniform discharge, incision is expected to persist along the steeper portions (high stream power), while low-gradient portions (low stream power) may aggrade. In contrast, where gullies have coalesced, incised reaches also span low-gradient reaches between initial headcut locations. Because the fine-scale patterns of incision and deposition in discontinuous gullies are superimposed on the broader pattern of a general increase in channel size downstream, which persists when gullies coalesce, stream power would then become a less efficient predictor of attributes such as width, depth, and cross-sectional area than would drainage area alone.

Morphological characteristics associated with reaches having low headcut frequencies are expected to diverge between logged and control watersheds as logged channels adjust to altered conditions. If low frequencies in logged reaches indicate coalescence of headcuts rather than absence of incision, mean channel depths in reaches with few headcuts should be greater in logged watersheds than in control watersheds, and this is indeed the case. In channel segments where headcut frequency is less than about 10/100 m of channel, mean channel depth is significantly greater in logged watersheds (1.6 ± 0.4 m) than in control watersheds (0.7 ± 0.2 m, Fig. 11). In the South Fork watersheds, the distribution of channel depths as a function of headcut spacing is most similar to that in North Fork control watersheds.

5.4. Contrasts between North Fork and South Fork channels

The intermediate position of South Fork channel morphological relations relative to those in North Fork control and clearcut watersheds (Fig. 6) may in part reflect ongoing recovery from second-cycle logging effects in the South Fork. The South Fork was logged 28 to 30 years before the gully survey, while North Fork watersheds had been logged only 10 to 12 years earlier. Differences in silvicultural strategy and yarding techniques would also contribute to differing results. The selective logging employed in the South Fork resulted in less hydrologic change from altered vegetation than in the clearcut North Fork watersheds, but the use of tractors for yarding produced widespread compaction that continues to generate overland flow at some sites. In addition, South Fork logs were skidded along some low-order channels, directly disrupting the channels and likely producing a very different distribution of channel forms immediately after logging than was present after North Fork logging. Also in contrast to the North Fork, South Fork roads and landings were located mid-slope and in riparian zones, strongly

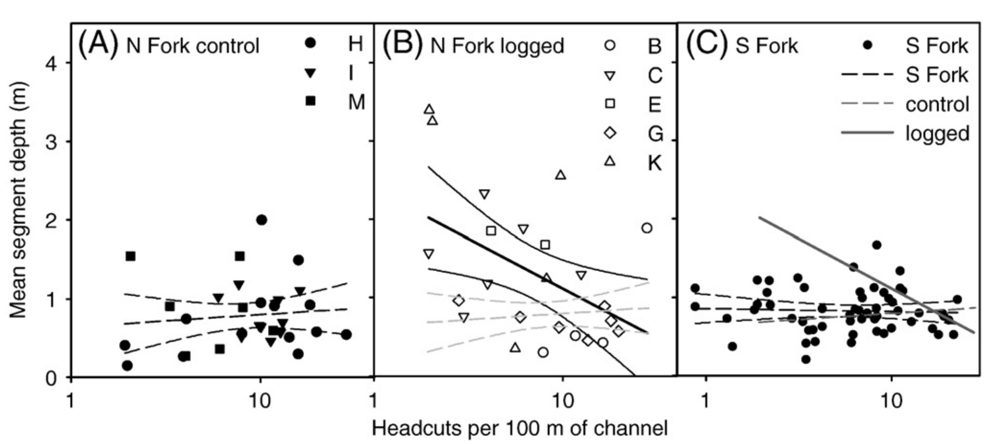

Fig. 11. Mean depth and headcut frequency in 50-m channel segments of (A) North Fork control, (B) North Fork logged, and (C) South Fork tributaries. Relations significant at the 0.05 level are depicted by solid lines.

increasing the potential for channel disruption by road drainage and mechanical disturbance.

5.5. Implications for forest management

Efforts to prevent and mitigate logging-related erosion in rain-dominated watersheds of the Pacific Northwest have generally relied on the use of buffer strips and on reduction of road-related erosion. Road erosion was found not to be of major consequence at North Fork Caspar Creek (Lewis et al., 2001), possibly reflecting drainage control efforts, road closures following logging, and pervasive water-barring of abandoned roads. Most North Fork roads and log landings are located near ridge tops, further reducing their potential for contributing sediment directly to channels. Robust buffer strips were incorporated into the logging plan, providing extensive filter strips below upland sediment sources and preventing direct disturbance to a significant portion of the stream network. Despite these measures, suspended sediment yields increased significantly after logging, and much of the increase appears to originate from gully-related processes that are not amenable to mitigation either through road improvements or buffer strips. If increased runoff after logging generates sediment from within downstream channels, control of excess sediment from this source would be possible only through management of the level of hydrologic change induced by logging, and this would require either management of the rate of logging within a watershed or modification of the silvicultural strategy used.

Logging prescriptions generally consider only the distribution of channel types present at the time that plans are developed and so do not reflect the possibility that the channel network may expand after logging. The apparent 28% increase in drainage density after logging at Caspar Creek would strongly increase the connectivity between hillslope sediment sources and the downstream channel network. Plans to maintain a prescribed distance between ground-disturbing activities and stream channels are defeated if channel networks expand into the disturbed sites after logging.

Logging plans on lands administered by US Federal agencies or regulated by California State agencies are required to include an evaluation of potential cumulative and indirect impacts of the planned logging, but considerable uncertainty and controversy have at times surrounded the definition of what constitutes an adequate impact analysis. Observation of the Caspar Creek gullies suggests that future analyses might usefully consider the possible influence of logging-related hydrologic changes on downstream channel morphology and sediment inputs.

Interest is growing in the use of indirect methods for inferring long-term erosion rates to allow comparison to management-related sediment inputs. Several studies have evaluated concentrations of cosmogenic ^{10}Be in soils and sediment to estimate long-term input rates (e.g., Kirchner et al., 2001). In the case of Caspar Creek, Ferrier et al. (2005) used results of such a study to conclude that recent erosion rates evaluated from monitoring data at Caspar Creek are lower than rates characteristic of the pre-logging period. Such conclusions rest heavily on the assumption that the distribution of sediment sources that produced the sampled sediment is typical of the distribution present over the period for which long-term rates are to be inferred. However, examination of the Caspar Creek tributaries indicates that gullying is now pervasive, that it probably initiated with or was greatly accelerated by first-cycle logging, and that many of the gullies excavate cosmogenically “pristine” sediment sources such as buried saprolite and bedrock. Under these conditions, samples obtained from in-channel sediments will contain lower ^{10}Be concentrations than would be expected from sediment exported before gully initiation, and estimated “long-term” erosion rates may instead disproportionately reflect accelerated erosion resulting from first-cycle logging.

6. Conclusions

Results of the Caspar Creek study suggest that erosion along incised channels is an important source of sediment in tributary watersheds. Gullying in the area appears to have expanded with first-cycle logging of the late 1800s. Channels had not yet recovered from the earlier impacts at the onset of second-cycle logging nearly 100 years later, and increased runoff resulting from second-cycle logging accelerated erosion within the still-incised channels. Recently logged watersheds show a higher drainage density than controls, with the extent of the increase similar to that expected on the basis of the observed change in runoff.

Because an appreciable portion of the increased sediment input at Caspar Creek is associated with hydrologic changes caused by logging and because a significant portion of the excess sediment is generated along channels in and downstream of the logged areas, the strategies most often used in the region to reduce sediment inputs from logging–control of road-related erosion and establishment of riparian buffer strips–are not effective for reducing an important component of the logging-related sediment input at Caspar Creek. In addition, efforts to reduce impacts from surface erosion by ensuring that soil-disturbing activities are not carried out near streams would need to take into account the potential for upslope expansion of the channel network after logging.

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