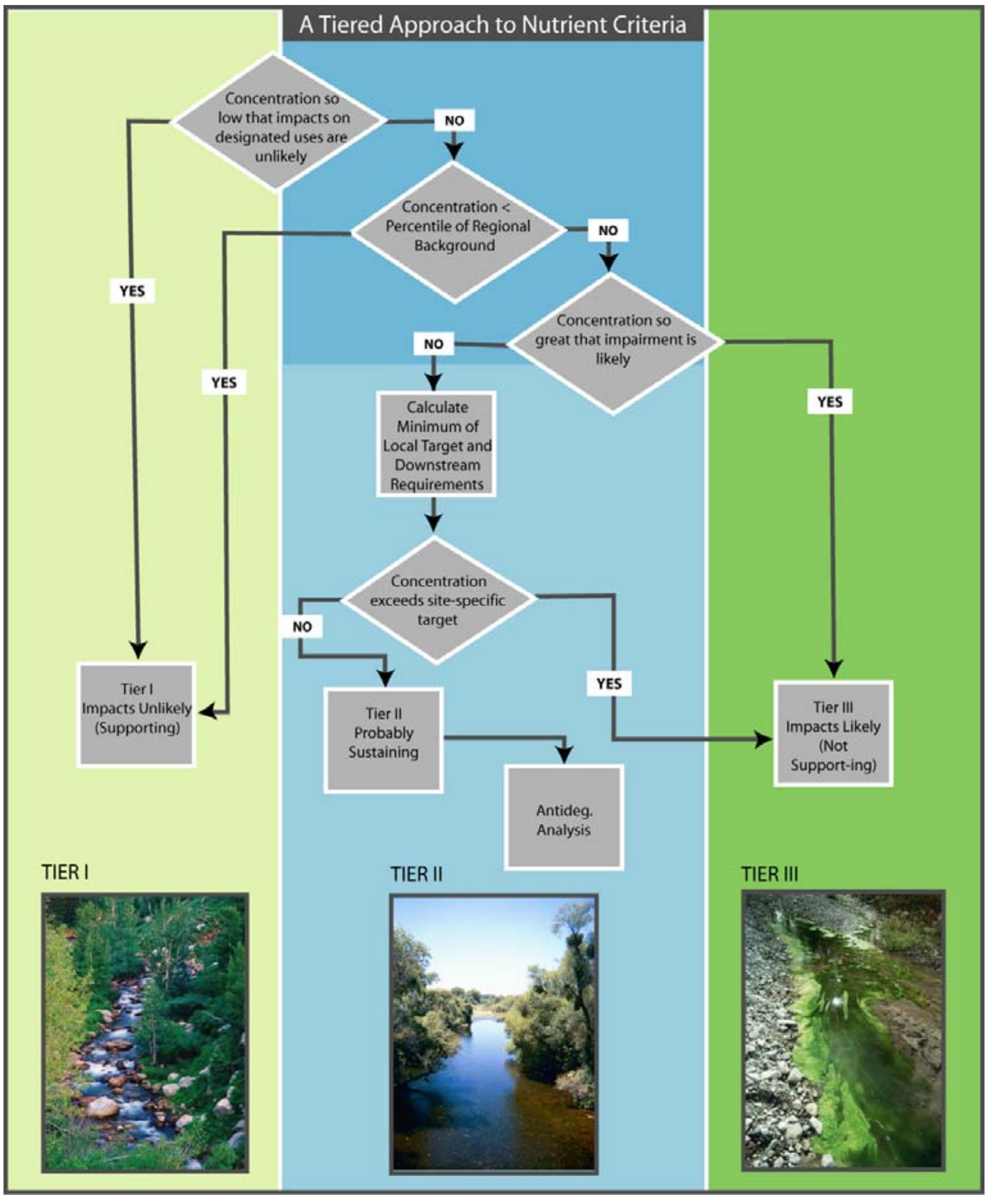
Progress Report
Development of Nutrient Criteria in California: 2003-2004



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1.0 INTRODUCTION

It is generally understood that nutrient loads have complex, and often indirect, effects on aquatic ecosystems that may lead to impairment of beneficial uses of water bodies. In many instances, these effects are also influenced by non-nutrient factors that may act differently in individual water bodies to mitigate or worsen problems caused by excess nutrients. Because nutrients have the potential to alter entire ecosystems, it is difficult to reproduce these effects in controlled laboratory studies. In this regard, nutrients are a class of chemicals distinct from toxicants where controlled studies can be used to identify endpoints of adverse impacts on specific organisms of interest, and where these endpoints may be translated into criteria. Of necessity therefore, efforts to obtain nutrient criteria must follow approaches that are different from those that have been widely applied for developing criteria for toxicants.

The process for developing nutrient criteria for the region started in 1998 with the publication of the *National Strategy for the Development of Regional Nutrient Criteria* (USEPA 1998). EPA then proceeded to develop national criteria recommendations on the basis of aggregated Level III ecoregions. Existing nutrient data from these ecoregions were used to assess nutrient conditions from 1990 to 1998. Reference conditions were then developed based on 25th percentiles of all nutrient data including a comparison of reference condition for the aggregate ecoregion versus the subecoregions. These 25th percentile values were characterized as criteria recommendations that could be used to protect waters against nutrient over-enrichment (USEPA, 2000). However, EPA also noted that States and Tribes may "need to identify with greater precision the nutrient levels that protect aquatic life and recreational uses. This can be achieved through development of criteria modified to reflect conditions at a smaller geographic scale than an ecoregion such as a subecoregion, the State or Tribe level, or specific class of waterbodies." EPA also encouraged that States and Tribes "critically evaluate this information in light of the specific designated uses that need to be protected."

1.1 EPA Region IX National Approach to Nutrient Criteria

EPA Region IX made an early commitment to the regional team concept for developing nutrient criteria by calling together the Regional Technical Advisory Group (RTAG) in 1999 prior to the completion of the EPA guidance documents for developing nutrient criteria. The RTAG conducted a pilot project in 1999 and 2000 to evaluate regional reference conditions for streams and rivers in aggregated Ecoregion II (Western Forested Mountains). The results of this project suggested that the proposed reference condition distributions used by EPA would require some refinement and supporting studies to ensure that the adopted criteria were appropriate.

In 2001 the California State Water Resources Control Board (SWRCB) created the State Regional Board Technical Advisory Group (STRTAG) to work in parallel with the RTAG and assume responsibility for nutrient criteria development for California and to better coordinate the activities of the individual Regional Boards. The RTAG and STRTAG continue to work in close association today. The RTAG and STRTAG reviewed the findings of the pilot study using the original Level III ecoregions to evaluate the draft default 304(a) criteria included in the criteria document that had been completed for rivers and streams. The comparison tables for total phosphorus and total nitrogen suggest that if the EPA reference-based values (draft 304(a)) are adopted that a large number of potentially un-impacted waterbodies would be misclassified as impaired. Therefore the RTAG and STRTAG responded to this potential for misspecification by adopting a resolution to pursue the EPA approved alternative to development alternate nutrient criteria.

1.2 Proposed California Approach

The EPA national approach has relied on a statistical analysis of monitoring data to select targets for nutrients. While this is a starting point, it bears little relationship to the nutrient concentrations or loads that present a risk to attaining specific designated uses. The proposed California approach relies on using selected biological responses in addition to nutrient concentrations. Although biological responses are not always measured and are more difficult to predict than concentrations, these measures appear to be more generalizable than nutrient concentrations. That is, it may be possible to agree that a given density of periphyton biomass is injurious to support of any coldwater fishery, or a given frequency of blue-green algal blooms impairs a municipal supply use, even if the nutrient concentration that will cause that result varies widely from stream to stream. Despite the additional data requirement, the advantage of the proposed approach is a more robust link to actual impairment of use, rather than an approach that relies on concentration data alone.

This document describes the proposed California approach, as well as practical suggestions for its implementation. Specifically, the development of nutrient criteria and nutrient TMDLs is evaluated in terms of the *risk* of impairment of designated uses. Several concepts from EPA’s (1998) *Guidelines for Ecological Risk Assessment* – in particular, the use of conceptual models and surrogate measures (or indicators) – are particularly useful for the development of nutrient criteria and the estimation of nutrient TMDLs. The risk-based approach is described in Section 2.

Section 3 discusses potential indicators and targets for nutrient assessment, while Section 4 shows how these targets can be incorporated into a tiered approach to nutrient criteria. Section 5 provides an overview of potential tools available for evaluating targets. Finally, Section 6 demonstrates the connection between a risk-based approach to criteria and the development of nutrient TMDLs.

2.0 POTENTIAL PATHWAYS OF BENEFICIAL USE IMPAIRMENT BY NUTRIENTS

The approach taken for the California pilot study is to propose nutrient criteria based on an evaluation of risk relative to designated beneficial uses. Essentially, the objective is to control excess nutrient loads/concentrations to levels such that the risk or probability of impairing the beneficial uses is limited to a low level. If the nutrients present – regardless of actual magnitude – have a low probability of impairing uses, then water quality standards can be considered to be met. (Of course, in some cases further reductions in nutrients may be desirable to meet non-regulatory management goals – but this is not an issue to be addressed through criteria and standards.)

The basic problem is thus to link specific beneficial uses to levels of nutrients that are likely to impair those uses. Establishing this connection is an exercise in risk assessment, for which the techniques developed for ecological risk assessment (ERA) in particular are highly relevant. This section first discusses the beneficial uses of California fresh water bodies. This is followed by a general description of the risk-based approach. Finally, Section 2.3 describes the conceptual linkage between nutrient loads and risk of use impairment.

2.1 BENEFICIAL USES

State policy for water quality control in California is directed toward achieving the highest water quality consistent with maximum benefit to the people of the state. Aquatic ecosystems and underground aquifers provide many different benefits to the people of the state. Beneficial uses define the resources, services, and qualities of the state's aquatic systems that guide protection of water quality; they also serve as a basis for establishing water quality objectives. Several studies have linked nutrient enrichment to beneficial use impairment. The list of beneficial uses provides a starting point in understanding the relationships between nutrients and use impairment.

The following beneficial uses are used throughout California for freshwater systems. It should be noted that in general, waterbodies are assigned multiple beneficial uses.

Agricultural Supply (AGR): Uses of water for farming, horticulture, or ranching, including, but not limited to, irrigation, stock watering, or support of vegetation for grazing. Adverse impacts of elevated nutrients are unlikely for this use.

Areas of Special Biological Significance (ASBS): Designated by the SWRCB. These include marine life refuges, ecological reserves, and designated areas where the preservation and enhancement of natural resources requires special protection. Elevated nutrients, while most likely not posing a toxicological threat, could significantly alter the natural ecology of the systems that are protected by this use designation.

Cold Freshwater Habitat (COLD): Uses of water that support cold water ecosystems, including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish, or wildlife, including invertebrates. These habitats typically have clear, low nutrient waters and are susceptible to significant degradation by elevated nutrient loads.

Freshwater Replenishment (FRSH): Uses of water for natural or artificial maintenance of surface water quantity or quality. Elevated nutrients in replenishment waters may have adverse impacts when released downstream to waters with other beneficial uses.

Groundwater Recharge (GWR): Uses of water for natural or artificial recharge of groundwater for purposes of future extraction, maintenance of water quality, or halting saltwater intrusion into freshwater aquifers. Elevated nutrients are unlikely to have major impacts on this use unless nitrate levels are so high as to exceed criteria for protection of human health. Excessive algal growth may, however, indirectly degrade uses of water for groundwater recharge by increasing levels of total organic carbon and total dissolved solids.

Industrial Service Supply (IND): Uses of water for industrial activities that do not depend primarily on water quality, including, but not limited to, mining, cooling water supply, hydraulic conveyance, gravel washing, fire protection, and oil well repressurization. Elevated nutrients are unlikely to result in major impairments of this use, except that excessive algal growth might result in clogged intake pipes.

Fish Migration (MIGR): Uses of water that support habitats necessary for migration, acclimatization between fresh water and salt water, and protection of aquatic organisms that are temporary inhabitants of waters within the region. Elevated nutrients, while most likely not posing a toxicological threat, could stimulate primary productivity and result in increased food supplies or shelter for aquatic life. Excessive primary productivity, however, could result in excessive periphyton growth, which could shed and create blockages or dams that inhibit migration. Additionally, excessive primary productivity can cause depletion of oxygen supplies and impact aquatic life.

Hydropower Generation (POW): Uses of water for hydroelectric power generation. Elevated nutrients are unlikely to result in significant impairment of this use.

Municipal and Domestic Supply (MUN): Uses of water for community, military, or individual water supply systems, including, but not limited to, drinking water supply. Elevated nutrients could stimulate primary productivity and result in clogged intake pipes. Blooms of certain blue-green algae can release toxic substances that may impair domestic supply, and a variety of algal species can result in taste and odor problems in finished water. Additionally, elevated concentrations of nitrate (>10 mg/l) exceed levels deemed protective of human health.

Navigation (NAV): Uses of water for shipping, travel, or other transportation by private, military, or commercial vessels. Nutrients are unlikely to impair navigation uses. However, excessive primary productivity could result in nuisance macrophyte and filamentous algal growth, which could inhibit navigation.

Industrial Process Supply (PRO): Uses of water for industrial activities that depend primarily on water quality. Elevated nutrients could stimulate primary productivity and result in clogged intake pipes.

Preservation of Rare and Endangered Species (RARE): Uses of waters that support habitats necessary for the survival and successful maintenance of plant or animal species established under state and/or federal law as rare, threatened, or endangered. Elevated nutrients, while most likely not posing a toxicological threat, could significantly alter the natural ecology of the systems that are protected by this use designation.

Water Contact Recreation (REC-1): Uses of water for recreational activities involving body contact with water where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, water-skiing, skin and SCUBA diving, surfing, whitewater activities, fishing, and uses of natural hot springs. Elevated nutrients can exacerbate algal blooms that cause unaesthetic conditions for contact recreation, while blooms of some species can cause skin irritation and potential toxic effects. This use may also be indirectly impaired by degradation of the aquatic life uses that support fishing.

Noncontact Water Recreation (REC-2): Uses of water for recreational activities involving proximity to water, but not normally involving contact with water where water ingestion is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tide pool and marine life study, hunting, sightseeing, or aesthetic enjoyment in conjunction with the above activities. Elevated nutrients can exacerbate unsightly algal blooms that cause unaesthetic (visual and olfactory) conditions for noncontact recreation. This use may also be indirectly impaired by degradation of aquatic life uses that support wildlife.

Shellfish Harvesting (SHELL): Uses of water that support habitats suitable for the collection of crustaceans and filter feeding shellfish (clams, oysters, and mussels) for human consumption, commercial, or sport purposes. Elevated nutrients, while most likely not posing a toxicological threat, could stimulate primary productivity and result in increased food supplies or shelter for aquatic life. Excessive primary productivity, however, could result in depletion of oxygen supplies and impact aquatic life. Blooms of toxic algal species may also severely impair this use.

Fish Spawning (SPWN): Uses of water that support high quality aquatic habitats suitable for reproduction and early development of fish. Elevated nutrients, while most likely not posing a toxicological threat, could stimulate primary productivity and result in increased food supplies or shelter for some types of aquatic life – but may alter habitat suitability for others. Excessive primary productivity could also result in depletion of oxygen supplies in spawning gravels.

Warm Freshwater Habitat (WARM): Uses of water that support warm water ecosystems including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish, or wildlife, including wildlife. Elevated nutrients, while most likely not posing a toxicological threat, could stimulate primary productivity and result in increased food supplies or shelter for aquatic life. Excessive primary productivity, however, could result in depletion of oxygen supplies and impact aquatic life.

Limited Warm Water Habitat (LWARM): Uses of water that support warmwater ecosystems which are severely limited in diversity and abundance as the result of concrete-lined watercourses and low, shallow dry weather flows which result in temperature, pH, and/or dissolved oxygen conditions. Naturally reproducing finfish populations are not expected to occur in these waterbody types. Elevated nutrients may further degrade such naturally limited habitat, but are probably unlikely to have significant effects relative to the limitations on support of aquatic life caused by habitat condition.

Wildlife Habitat (WILD): Uses of water that support wildlife habitats, including, but not limited to, preservation or enhancement of vegetation and prey species used by wildlife, such as waterfowl. Elevated nutrients could stimulate primary productivity and result in increased food supplies or shelter for wildlife. However, alteration of the natural aquatic ecology may indirectly impair certain desirable wildlife support uses.

While all beneficial uses must be considered, some are unlikely to be impaired by nutrients before other, more sensitive assigned uses covering the basics of the national “fishable, swimmable” goals are also impaired (e.g., agricultural supply, freshwater replenishment, groundwater recharge, industrial service supply, hydropower generation, navigation, industrial process supply, wildlife habitat). Such uses are not likely to be the driving force for nutrient criteria at a site. Areas of Special Biological Significance and Preservation of Rare and Endangered Species would appear to require site-specific management plans. Shellfish Harvesting applies to salt waters, which are not considered here. Accordingly, the remainder of this discussion focuses on some of the other beneficial uses that are both commonly assigned and, to one degree or another, sensitive to impairment by nutrients. These are: Cold Freshwater Habitat (COLD), Fish Migration (MIGR), Municipal and Domestic Supply (MUN), Water Contact Recreation (REC-1), Noncontact Water Recreation (REC-2), Fish Spawning (SPAWN), and Warm Freshwater Habitat (WARM).

2.2 Risk-Based Approach

Ecological risk assessment (ERA) is a process for evaluating the likelihood that adverse ecological impacts may occur in response to one or more stressors. ERA consists of three phases: planning and problem formulation, risk analysis, and risk characterization and is described in detail in EPA's *Guidelines for Ecological Risk Assessment* (U.S. EPA 1998). Keys to a successful ERA are identifying (1) the pathways by which stressors cause ecological effects and (2) informative and representative assessment endpoints. Assessment endpoints are the link between scientifically measurable endpoints and the objectives of stakeholders and resource managers (Suter 1993). Endpoints should be ecologically relevant, related to environmental management objectives, and susceptible to stressors (U.S. EPA 1998).

A pivotal tool of the ERA process is development and evaluation of a conceptual model, and selection of assessment endpoints. A conceptual model is a graphical and narrative description of the potential physical, chemical and biological stressors within a system, their sources, and the pathways by which they are likely to impact multiple ecological resources (Suter 1999). The conceptual model is important because it links exposure characteristics such as water quality parameters (related to water quality standards) with the ecological endpoints important for describing the management goals (related to aquatic life support as designated under the Clean Water Act).

Conceptual model development has been identified as the single most valuable component of EPA's watershed-level ecological risk assessment case studies (Butcher et al., 1998). In each of the five EPA-sponsored case studies, conceptual model development in accordance with the ERA framework was identified as particularly valuable in providing a solid foundation for stakeholder communication, strategic data collection, and priority ranking and targeting.

Conceptual models consist of two general components (US EPA, 2001): (1) A description of the hypothesized pathways between human activities (sources of stressors), stressors, and assessment endpoints; and (2) a diagram that illustrates the relationships between human activities, stressors, and direct and indirect ecological effects on assessment endpoints. The conceptual model consolidates available information on ecological resources, stressors, and effects, and describes, in narrative and graphical form, relationships among human activities, stressors, and the effects on valued ecological resources (Suter, 1999).

In large part, the pathways or connections between sources, stressors, and effects are a series of hypotheses. Those pathways or relationships that are of greatest interest or concern to stakeholders will form the risk hypotheses that are specifically examined in the risk assessment. Thus, the conceptual model will summarize or depict those risk hypotheses. Specific assumptions or hypotheses may be based on theory and logic, empirical data, information from other watersheds, or mathematical models. Thus, they are formulated using a combination of professional judgment and available information on the ecosystem at risk, potential sources of stressors, stressor characteristics, and observed or predicted ecological effects on selected or potential assessment endpoints.

A conceptual model provides a visual representation for the cases where multiple stressors contribute to water quality problems. With the conceptual model, some attribute or related surrogate (termed an "indicator" in both the watershed approach [U.S. EPA (1995)] and the TMDL program) provides a measurable quantity that can be used to evaluate the relationship between pollutant sources and their impact on water quality (U.S. EPA (1999a).

The specific exposure pathways contained within a conceptual model determine what needs to be analyzed to complete the TMDL. For instance, the Garcia River TMDL (USEPA Region 9, 1998) contained the following general problem statement:

The Garcia River watershed has experienced a reduction in the quality and quantity of instream habitat which is capable of supporting the cold water fishery, particularly that of coho salmon and steelhead. Controllable factors contributing to this habitat loss include the acceleration of sediment production and delivery due to land management activities and the loss of instream channel structure necessary to maintain the system's capacity to efficiently store, sort, and transport delivered sediment.

This general problem statement was followed by a series of specific instream and upland problem statements that are essentially individual risk hypotheses. For instance, the problem statement relating to fine sediment in spawning gravels reads as follows:

Spawning gravels of the Garcia River watershed are impacted and likely to suffer additional impacts by the delivery of fine sediment to the stream which fills the interstices of the framework particles: 1) cementing them in place and reducing their viability as spawning substrate; 2) reducing the oxygen available to fish embryos; 3) reducing intragravel water velocities and the delivery of nutrients to and waste material from the interior of the redd (salmon nest), 4) and impairing the ability of fry (young salmon) to emerge as free-swimming fish...

An important role of these statements is to lay out the rationale for selecting measures or indicators and the choice of modeling or linkage analysis tools. The goal (supporting the cold water fishery) is tied to a stressor (delivery of fine sediment to the stream) by an exposure process (filling of spawning gravels by fine sediment). This leads directly to the consideration of measures of spawning gravel condition, and the need for linkage tools that can assess the process of upland sediment generation, loading to the stream, and impact on the substrate.

2.3 Conceptual Models of Nutrient Impairment

There are many complex ways in which excess nutrient loads can impact one or more beneficial uses. General conceptual models for the impairment of key uses in lakes and streams by nutrients are presented in Figures 2-1 and 2-2. Additional linkages may be significant in individual waterbodies; however, most of the major linkage connections are captured in these figures.

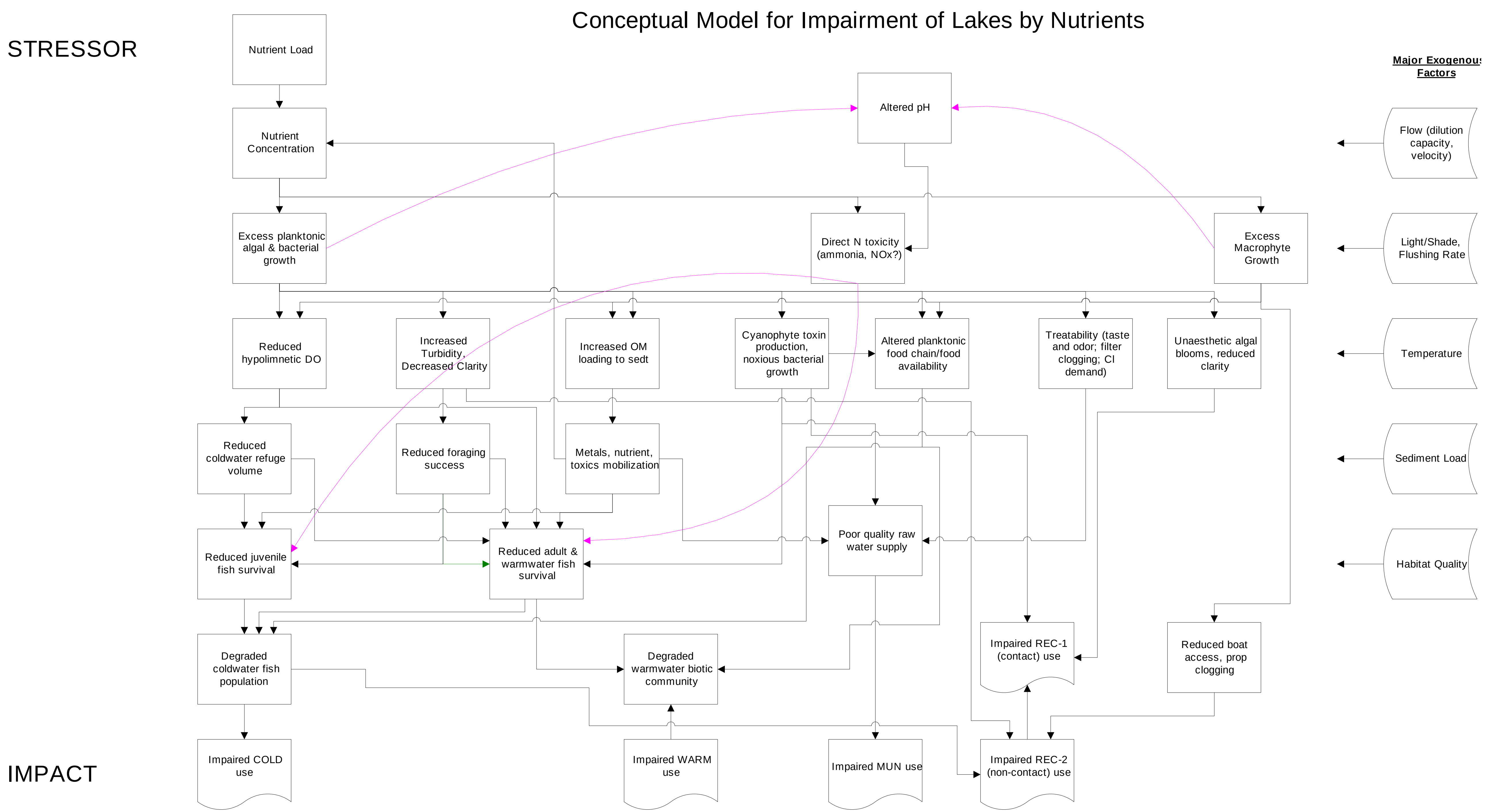
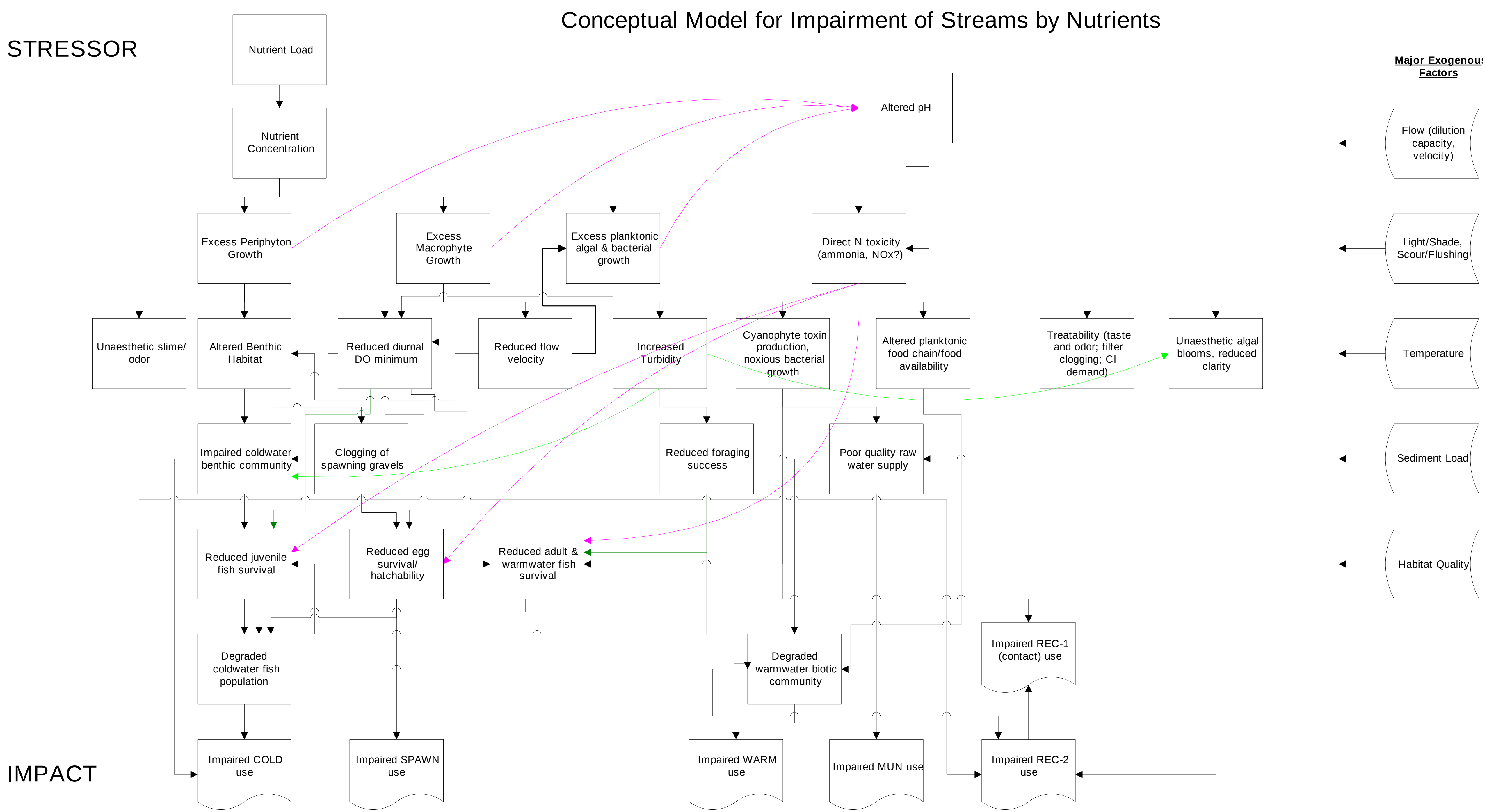


Figure 2-1 Conceptual Model for Lakes



Note: AGR use is assumed not to be impaired by nutrient loads
NAV is insensitive, but could be impacted by macrophytes

Figure 2-2 Conceptual Model for Streams

2.4 RISK HYPOTHESES

Each pathway (from the nutrient load stressor to one of the use impairments) through the conceptual models constitutes a risk hypothesis. Given the complexity of the conceptual models, there are many individual pathways or risk hypotheses to consider.

In a place-based watershed ERA, one would typically begin with a full conceptual model (modified as appropriate for the watershed under study), identify the most significant pathways, then proceed with the analysis using these selected pathways as the key risk hypotheses. For generalized nutrient criteria the concept is still relevant; however, there is not the luxury of sifting the many potential risk hypotheses for importance based on site-specific characteristics. Therefore, it is necessary to pare the list to identify, in generic form, those risk hypotheses that are most likely to be important and/or can stand in as surrogates for other, less common risk pathways.

The complex conceptual models may first be reduced to a table showing the relationship of key uses to major stressor response factors that can be key causes of impairment of use, as shown in Table 2-1. The stressor-response factors primarily relate to problems of excess algal or macrophyte growth, and may be further simplified to generic risk hypotheses.

These simplified, generic risk hypotheses are summarized as follows:

Lakes/Reservoirs

Excess nutrient load results in excess planktonic algae (and macrophyte) biomass that may increase turbidity, alter the food chain, create unaesthetic conditions, and alter the DO balance, leading to impairment of uses. The exact format of the risk hypotheses depends on the uses that are designated and characteristics of the waterbody.

Rivers/Streams

Excess nutrient loads result in (a) excess planktonic algae biomass (larger, slow moving rivers) and/or (b) excess periphyton or macrophyte biomass (smaller, higher-gradient systems) that may alter the food chain and benthic habitat, cause unaesthetic conditions, and alter the DO balance, leading to impairment of uses. The exact format of the risk hypotheses depends on the uses that are designated and the characteristics of the waterbody.

These generic risk hypotheses are useful for criteria development because they help focus in on the key points in common site-specific risk hypotheses that control the linkage between stressors and impacts. Specifically, these are planktonic algae biomass in lakes, reservoirs, and larger, slower moving rivers, and periphytic algae or macrophyte biomass in higher gradient streams and rivers.

Table 2-1
Stressor-Response Factors

Use	Reduced hypolimnetic DO	Increased Turbidity	Cyanophyte toxins	Altered food chain	Toxic metal, NH4 cycling	Taste & Odor	Unaesthetic Blooms	Excess Macro-phytes	Secondary Factors
	Key Stressor-Response Factors								
COLD	X	X	X	X	X				Summer chl a, cyanophyte blooms, tyrbidity, metals, ammonia
WARM	X		X		X				Cyanophyte blooms, turbidity, metals, ammonia
MUN		X	X			X			Taste & odor, filter cloggers, cyanophyte toxins
REC-1		X	X				X		Frequency of algal blooms, cyanophyte toxins, impaired REC-2
REC-2		X					X	X	Impairment of WARM, macrophyte density

2.4.1 MEASURES, INDICATORS AND TARGETS

In an ecological risk assessment, the true assessment endpoints are the valued ecosystem characteristics that are desired to be protected. In a regulatory context, the designated beneficial uses and their associated narrative criteria may be considered as assessment endpoints. These assessment endpoints (such as health of a salmonid fishery) are often difficult to predict or measure directly. Therefore, an ERA usually proceeds through the evaluation of simpler endpoints (referred to as indicators or *measures*) that are measurable and predictable, and serve as surrogate measures to link stressors and outcomes.

In current ERA guidance, these "measures" include 1) **measures of effect** (formerly known as "measurement endpoints"), defined as "measurable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed", 2) **measures of exposure**, defined as "measures of stressor existence and movement in the environment and their contact or co-occurrence with the assessment endpoint", and 3) **measures of ecosystem and receptor characteristics** (U.S. EPA, 1998). The TMDL and Watershed Approach literature tends to refer to these measures as "indicators".

A target is simply a value of an indicator that is consistent with attaining the assessment endpoint or management objective. In other words, a target is equivalent to a criterion value for protecting a specific use at a given site.

2.5 MEASURES OF EFFECT AND MEASURES OF EXPOSURE

In the context of nutrients, measures of effect are those measurable quantities that are associated with impairment of the use and caused by nutrients. These could include things such as a decline in the stock or recruitment success of a coldwater fishery (for the COLD use), the occurrence of unaesthetic algal mats (for the REC uses), or algal-derived taste and odor problems in finished drinking water (for the MUN use). Measures of effect are very useful in *retrospective* risk assessments – that is, they confirm that a problem has occurred. For water quality regulation, measures of effect are similarly a key component of use assessment. Measures of effect are also key for tracking improvements in response to management actions. They are generally of less use, however, in *prospective* risk assessments, in which the need is to determine whether an adverse impact on the assessment endpoint or beneficial use, which has not yet been documented, is likely to occur. In addition, a measure of effect can be difficult to attribute to a specific source. For instance a degraded fishery might be due to elevated nutrient loads, toxicity, or habitat alteration. For these reasons, measures of effect are of limited use in developing nutrient criteria.

For nutrients, measures of exposure would refer foremost to nutrient concentrations or loads and are direct measurements of the loaded stressor (nutrients) that is hypothesized to cause an adverse impact on the assessment endpoints.

Some measures of great relevance to the analysis of impairment by nutrients are midway between the somewhat arbitrary definitions of measures of effect and measures of exposure. Most notably, increased algal biomass is an effect resulting from nutrient load that in turn serves as a stressor relative to a variety of ecological processes that support beneficial uses. This class of *intermediate measures* plays a key role in the generic risk hypotheses set forth for nutrient criteria determination in Section 2.4 because they represent a key intersection along the complex path from nutrient loading to impairment of beneficial uses.

Some states have addressed nutrient criteria through direct measures of exposure – setting target concentrations of nutrients applicable to a class of waterbodies. Other states have focused on intermediate measures. For instance, Georgia and Alabama assign chlorophyll *a* criteria to lakes and, if impairment is assessed, allocate nutrient loads on a site-specific basis to meet these criteria.

Reliance of measures of exposure alone (e.g., nutrient concentration targets) presents problems because the amount of nutrients that a waterbody can assimilate without impairment of uses varies widely, depending on a large number of cofactors. The intermediate measures appear to be more generalizable. That is, it may be possible to agree that a given density of periphyton biomass is injurious to support of any coldwater fishery, or a given frequency of blue-green algal blooms impairs a municipal supply use, even if the nutrient concentration that will cause that result varies widely from stream to stream. The drawback to the use of intermediate measures is that they are more difficult to predict, and do not provide a direct indication of what nutrient loads may be appropriate without a site-specific analysis.

The proposed approach for California nutrient criteria relies on both measures of exposure and intermediate measures or indicators, and seeks to capture the strengths of each. Specifically, the setting of targets relies primarily upon intermediate measures assigned to ensure support of a beneficial use; however, the target is then interpreted into a corresponding measure of exposure through a procedure that takes into account the stratifying or differentiating factors that distinguish the response of one waterbody for another. For instance, suppose that a given use in a reservoir will be supported if growing season mean chlorophyll *a* concentrations are held to 25 µg/L or less (an intermediate measure). This may then be interpreted into a corresponding target level of nutrient load (a measure of exposure) by a procedure that takes into account key factors (such as hydraulic retention time, depth, volume, latitude, and so on) that determine the nutrient response within the lake.

The California approach is intentionally positioned as a compromise between the one-size-fits-all approach of applying statistical nutrient criteria (which may have little relevance to the support of a given use in a specific waterbody) and the development of a true site-specific criterion (which would require intensive study and allocation of scarce resources that may not be available). As such, the California approach will yield criteria that are more closely related to actual use support than generic ecoregional targets, but are applicable without a detailed, site-specific study. As will be seen in Section 4, one outcome of the assessment process associated with criteria of this type is to identify those marginal sites for which a more detailed analysis is warranted before allowing additional nutrient loads.

2.6 INDICATORS AND TARGETS FOR LAKES AND RESERVOIRS

The intermediate measures most relevant to support of specific uses in lakes and reservoirs are primarily algal or macrophyte biomass. Appropriate levels vary with the use. Careful consideration should also be directed toward the spatial and temporal specification of the intermediate measure target. For instance, support of an oligotrophic, cold water fishery in a lake is most appropriately defined in terms of a growing season mean chlorophyll *a* concentration (as a surrogate for algal biomass) throughout the lake. In contrast, a warm water fishery is unlikely to suffer direct deleterious effects (and may even benefit) from increased algal production, and is only likely to suffer impairment from indirect effects that occur when biomass production is high enough to create conditions of depleted dissolved oxygen, excessive algal turbidity, altered pH, or elevated metal and ammonia concentrations. For a municipal supply use, algal concentrations at the water supply intake are highly relevant, but concentrations elsewhere in the lake/reservoir are of less importance. In addition, impairment of this type of use may be more dependent on the frequency of blooms of noxious algal species that cause treatment problems than on the average algal biomass.

These intermediate measures are in terms of algal growth and may be linked back to measures of exposure in terms of nutrient concentrations or loads. The linkage for criteria development should use a simplified modeling approach that takes into account the major stratification factors that cause site specific differences in response.

Targets, which form the actual criterion numbers, can then be expressed in terms of nutrients. For most nutrient impacts in lakes and reservoirs, the nutrient loading rate is more important than the nutrient concentration, because lakes store and recycle nutrients. It is important to note that both nitrogen and phosphorus are potentially controlling on algal response, and thus the acceptable level of one major nutrient may depend on the level of the other. For example, if a lake is strongly phosphorus limited then sensitivity to additional nitrogen loads will be low. One approach is to develop the targets as a response surface that shows the acceptable combinations of nitrogen and phosphorus levels. Another possible approach is to develop the criterion in terms of a composite limiting nutrient that reflects the average stoichiometric needs of algal cells. Walker (1987) advocates this method and developed an estimate of the composite limiting nutrient concentration as

$$X_{PN} = \left[P^{-2} + \left\{ (N - 150)/12 \right\}^{-2} \right]^{-1/2}$$

where X_{PN} is the composite limiting nutrient concentration (µg/L), and P and N are the total phosphorus and total nitrogen concentrations (µg/L).

2.7 INDICATORS AND TARGETS FOR RIVERS AND STREAMS

Analysis of nutrient risk hypotheses is generally more difficult for rivers and streams than for lakes. In many cases, periphytic algal biomass (usually measured as chlorophyll *a* per unit area) provides an appropriate intermediate measure for a variety of potential risk factors. However, the linkage between measures of exposure and the intermediate measures is generally less predictive and more uncertain than in lakes. This occurs due to the importance of a variety of confounding factors, including scour/sloughing, grazing, restrictions on growth by canopy shading, and the direct supply of nutrients from the sediment.

Linkages can, however, be built successfully between nutrient concentrations and potential maximum periphytic algal growth in the absence of other limiting factors, as summarized in Section 4. As with lakes and reservoirs, the linkage analysis should include a representation of the most significant co-factors to set site specific requirements. However, because of the uncertainty inherent in predicting nutrient response in streams it may be preferable to use multiple lines of evidence to set stream nutrient criteria.

In addition to algal measures, streams must also meet established numeric criteria for other factors that may be related to nutrient response, including DO, pH, and ammonia toxicity for aquatic life support and nitrate concentrations for municipal supply use. Thus a stream criterion may reflect the minimum of criteria obtained from analysis of a variety of risk hypotheses

2.8 Downstream Requirements

In many cases, concentrations necessary to meet criteria in receiving lakes and estuaries will be more stringent than those needed to prevent excess algal growth in the flowing streams that feed the terminal waterbody. The receiving lake or estuary, however, is impacted by the net loading from all feeder streams, and not directly by nutrient high levels in any individual stream segment. Clearly, downstream requirements should be interpreted back to tributary streams, but the way in which this is best done is open to discussion. The simplest approach would be to assign the loading target from the receiving waterbody to all upstream segments. This, however, is likely to result in overly stringent criteria, as significant losses of nutrient load typically occur in transit through the stream system, thus potentially allowing higher nutrient concentrations in upstream reaches. The inheritance of criteria from downstream segments can thus be improved through the use of analyses that account for such transit losses (e.g., Smith et al., 1997). Whether or not downstream requirements should be applied equally to upstream segments, or apportioned among upstream segments on another basis is essentially a policy decision.

2.9 Recent Literature on Nutrient Impacts in Water Bodies

The role of nutrients in aquatic ecosystems is complex, and the addition of excess nutrients to a water body results in a host of effects, from the microbial level to the top predator level. Although researchers have a general idea of these relationships, based on a body of scientific literature that, with the exception of lakes, stretches back at least four decades, it is less common to find quantitative relationships between nutrient levels and specific nutrient impacts. In large part this is due to the differences between natural systems, where similar nutrient concentrations may not cause similar responses because of non-nutrient factors, such as flow, shading, sediment loads, etc. However, from the perspective of numeric nutrient criteria development, quantitative relationships are important because they can help relate a desired level of biological response (such as dissolved oxygen or chlorophyll *a* levels) to a specific nutrient level, and can be used over a geographic region or for a group of similar water bodies.

For this review we have focused on reporting information that is most pertinent to nutrient criteria development in California and that does not repeat the excellent and thorough reviews of the state of understanding that have been presented in the US EPA guidance documents for lakes and reservoirs (US EPA, 2000a), streams and rivers (US EPA 2000b), and estuaries (US EPA, 2001). This review is based on literature from the last 15 years where biotic effects of nutrients on streams, lakes, estuaries and coastal waters have been studied.

2.9.1 STREAMS AND RIVERS

In examining literature on nutrients in streams, we focused on studies where authors had reported relationships between nutrient levels and any biological impacts. In almost all instances the response that was defined quantitatively was that between nutrients and mean or maximum chlorophyll levels in periphyton. In Table 2-2 we present regressions between chlorophyll and nutrient concentrations from the literature. When several alternative expressions were presented by authors, we focused on those with the best fits (highest r^2 values). In several instances, authors presented data on nutrients and chlorophyll levels, but did not perform a regression. In these cases, we independently estimated best fits using simple and multiple linear regression on the published data. These are also presented in Table 2-2.

Most studies reported in Table 2-2 show a fairly strong correlation between observed mean and maximum chlorophyll concentrations and some nutrient species (most commonly one or more of the following: TP, TN, SRP, and TKN). In most cases, phosphorus or nitrogen species alone could explain the observed chlorophyll levels, and in some cases, both nitrogen and phosphorus were required to explain the observations. This compilation of studies shows that it is incorrect to make a simple generalizations that phosphorus is the primary limiting nutrient in freshwaters (as opposed to nitrogen being the primary limiting nutrient in marine waters). Further, it was noted by several authors that chlorophyll concentrations are significantly impacted by the flow rate (Snelder *et al.* (2004), Biggs (2000), Biggs and Close (1989), Welch *et al.* (1988), Heiskary and Markus (2001)). Biggs (2000) explicitly considered flow in the regressions, where the effect of scour by flood flows is incorporated as a factor called "days of accrual." Chlorophyll concentrations were positively correlated to days of accrual, and the inclusion of this factor in the regressions improved the quality of the fit. In one case, conductivity was better at explaining chlorophyll a levels in periphyton than any nutrient species, but this may be the consequence of a correlation between nutrients and conductivity (Chetelat *et al.*, 1999).

Dodds *et al*. (1997) used data on benthic chorophyll (mean and maximum), planktonic chlorophyll, and nutrients to classify streams as oligotrophic, mesotrophic, or eutrophic. These boundaries are shown in Table 2-3. Values presented in this table can be a starting point for development of criteria in California.

Other studies have focused on effects that do not fit the formats of Tables 2-2 and 2-3, but are nonetheless important from the perspective of nutrient criteria. Sabater *et al.* (2000) explored the connection between chlorophyll *a* concentrations and the surrounding riparian vegetation. They found that in logged reaches of the stream there are much higher concentrations of planktonic chlorophyll (246.7 mg/m^2 in the logged reach versus 46.2 mg/m^2 in the shaded reach) and that the density of algal mats is increased. These findings serve to reiterate the impact of riparian communities on instream conditions. Sosiak (2002) found that, following a decline of nutrient loads over a period of 16 years, there were accompanying declines in periphyton and macrophyte biomass. A study in San Joaquin River, California, a river draining an arid region, found that algae communities were strongly affected by nutrients as well as salinity levels, both of which originate in agricultural drainage.

There also exists a significant body of literature evaluating changes in algal communities in response to nutrients in streams as well as other water bodies (e.g., Hill *et al*., 2000; Chetelat, *et al*., 1999; Winter and Duthie, 2000). However, in most cases it is difficult to relate changes in particular algal species to impairment of use. There are some exceptions, as when a particular alga starts to dominate the community, or when it imparts an odor to the water, but in general we will not focus at this level of detail for nutrient criteria development.

Table 2-2
Correlations between chlorophyll, nutrients, and other factors.

Citation	Parameters	Regression analysis	Comments
Correlations obtained from literature sources:			
Basu and Pick, 1996	Chl *a* and TP	Log chl *a* = -0.26 + 0.73 log TP	r^2 = 0.76, p<0.001, n=31
Van Nieuwenhuyse and Jones, 1996	Chl *a* and TP	Log chl = -1.65 + 1.99 log TP – 0.28 (log TP)^2	S=0.32, R^2=0.67, n=292
Chetelat *et al*., 1999	Chl a, TP	Log Chl a = 0.905 log TP + 0.49	r^2 = 0.56; Conductivity a better explainer than TP (r^2 = 0.71)
Biggs, 2000 (from Snelder *et al*., 2004)	Maximum Chl *a* and SIN	Log_{10} (maximum chl *a*) = 4.285 (log_{10} D_a) – 0.929 (log_{10} D_a)^2 + (0.504 log_{10} SIN) – 2.946	D_a = Days of accrual as determined from D_a = (1/*FRE*3) × 365.25 where FRE3 is the mean number of flood evens per year that exceed 3 times the median flow.
	Maximum Chl *a* and SRP	Log_{10} (maximum chl *a*) = 4.716 (log_{10} D_a) – 1.076 (log_{10} D_a)2 + (0.494 log_{10} SIN) – 2.741	As above
Dodds *et al*., 2002	Mean Chl *a*, TN, and TP	Log_{10} (mean Chl *a*) = 0.155 + 0.236 log_{10} TN + 0.443 log_{10} TP	r^2 = 0.40 (Mean Chl *a* regressions were also reported for a USGS data set but had much lower r^2 values.)
	Maximum Chl *a*, TN, and TP	Log_{10} (max Chl *a*) = 0.714 + 0.372 log_{10} TN + 0.223 log_{10} TP	r^2 = 0.31
Winter and Duthie, 2000	Mean Chl *a*, TN, TP		Both the relationships between mean chl *a* and TN (r^2=0.33, *p*=0.04); and mean chl *a* and TP (r^2=0.17, *p*=0.16) are significant.
Correlations developed by us from data reported in studies:			
Biggs, 2000	Chl *a*, SIN, SRP, Days accrual	Chl *a*= -4.309+1.495(SRP) +0.604 (D_a)	r^2= 0.22, showed a marginal increased relationship with the addition of SIN
Heiskary and Markus, 2001	Max ChlT (Chl *a* + Pheo) and TP	Max Chl T = -19.815 + 0.632 (TP)	r^2=0.78, showed a marginal increased relationship with the addition of NO_3
	Max ChlT and TKN	Max Chl T = -86.109+144.539 (TKN)	r^2=0.85, same r^2 value whether or not TP was added
Welch, 2001	Max Chl *a* and NO_3+NO_2-N	Max Chl *a* = 48.928 + 0.238 NO_3+NO_2-N	r^2=0.26, showed no increased relationship with the addition of TP
Biggs and Close, 2001	Mean Chl *a*, TKN, and TP	Mean Chl *a* = 11.501+0.813 (TP)	r^2=0.19, showed a marginal increase with the addition of TKN

Table 2-3
Classification of streams into oligotrophic, mesotrophic, or eutrophic categories (Dodds *et al*, 1998).

Variable	Oligotrophic-Mesotrophic Boundary	Mesotrophic-Eutrophic Boundary	N
Mean Benthic Chl (mg/m^2)	20	70	286
Max Benthic Chl (mg/m^2)	60	200	176
Planktonic Chl (ug/l)	10	30	292
TN (ug/l)	700	1500	1070
TP (ug/l)	25	75	1366

2.9.2 LAKES AND RESERVOIRS

Lakes and reservoirs are somewhat more amenable to development of correlations with nutrient chemistry because the complications arising from variable flow do not occur. For this reason, there have been comprehensive studies of nutrient-chlorophyll relationships for a much longer time, and nutrient chemistry data have been used to classify lakes into categories such as oligotrophic, mesotrophic, eutrophic (Vollenweider, 1968, and reproduced in Wetzel, 2001). Despite the age of the Vollenweider study, it is still accepted widely in the limnology literature. The guidance document for lakes and reservoirs (USEPA, 2000a) provides a comprehensive review of this literature, and will not be repeated here.

Although is generally considered that phosphorus is the main limiting nutrient in freshwaters, recent re-evaluation of large, global lake data sets shows that the relationship is not linear over large ranges, and that at moderately elevated phosphorus concentrations, lakes become nitrogen limited. The chlorophyll-phosphorus relationship is linear up to a point and then becomes flat due to nitrogen limitation (Prairie *et al*., 1989; McCauley *et al*., 1989). This is important information to consider in developing a predictive approach for criteria, although the model employed in our work method (BATHTUB) explicitly includes the possibility of both nitrogen and phosphorus limitation.

Other studies looking at changes in algal and zooplankton communities in response to nutrient loads, as discussed in the stream section above, are considered too detailed and limited in spatial coverage for broad application to nutrient criteria (e.g., Avalos-Perez *et al*., 1994; Balseira, *et al*., 1997; Cottingham, 1998; Koehler and Hoeg, 2000). However, some studies that use controlled experiments in lakes to evaluate the changes due to nutrient addition, particularly on upper trophic levels (e.g., Blanc and Margraf, 2002), may be useful to develop a scientific rationale for the linkage between lower trophic levels and beneficial uses.

2.9.3 ESTUARINE AND COASTAL WATERS

Because estuaries and some coastal zones have complex flows, with tidal effects, and varying degrees of mixing of freshwater and saltwater flows, it is very difficult to make quantitative generalizations about nutrient conditions across estuaries. For this reason, it is thought that the criteria development for estuarine waters will have to be conducted on a case-by-case basis. Although the mechanisms of interaction are different in these waters, the data needs will be broadly similar to that for stream and lake criteria development. The interactions of nutrients in estuaries and coastal waters as described in the most current research are well-documented in the US EPA Guidance Document (US EPA, 2002). As this is a recent document, it covers most recent research reports, and the effort is not duplicated here. What follows is a general discussion highlighting aspects of interest to Pacific coast.

Generally, however, the following aspects of nutrient-related responses are applicable everywhere. Excess nutrients, almost always nitrogen, allow the formation of algal blooms on the water surface during the warmest months of the year. As the algae in these blooms die and settle to the bottom, their decomposition consumes oxygen from the deeper layers. The depleted or lowered oxygen in these zones (anoxic or hypoxic zones) have adverse effects on all other biota. The likelihood of depleted oxygen in deeper waters is a function of the nutrient loading, the degree of mixing in the waters, and the degree of vertical stratification. Well-mixed, poorly stratified estuaries are less likely to have nutrient problems and, of the estuaries studied nationally for nutrient problems, it was found that most of the estuaries likely to be nutrient-impaired were along the coasts of the Atlantic Ocean and the Gulf of Mexico (29 estuaries on the east coast compared to 6 on the west coast) (Bricker *et al*., 1999). The 6 west coast estuaries with potential problems include: San Francisco Bay, Newport Bay, Tijuana Estuary, Elkhorn Slough, Tomales Bay, South Puget Sound, and Hood Canal (Bricker *et al*., 1999). The difference between the east and west coasts can be attributed to various reasons related to 1) the lower population density and runoff (and proportionally lower nutrient loads), 2) lower temperatures, and 3) lower atmospheric deposition of nitrogen.

Nutrient enrichment has also been associated with other infrequent problems, although, to date, most of these reported problems have been on the eastern U.S. One consequence of nutrient enrichment that is much less understood than the formation of anoxic zones is an increased frequency of algal blooms with toxins (termed harmful algal blooms, or HABs). It is thought that HABs are more likely to occur in the presence of nutrient enrichment, but because these are somewhat unpredictable events, it is not known what other factors play a role and whether control of nutrient loads alone can reduce the problem. Yet another consequence of elevated nutrients is thought to be the presence of the toxic dinoflagellate, *Pfiesteria piscicida*. *Pfiesteria*-like cells were positively correlated with phytoplankton biomass, which was shown to be positively correlated to increased nutrient concentrations (Pinckney *et al.*, 2000).

3.0 TECHNICAL APPROACHES FOR DEVELOPING NUTRIENT CRITERIA

There are three general types of approaches for nutrient criteria: Reference Site, Empirical, and Modeling.

The ***Reference Site*** approach relies on comparing conditions in a waterbody to conditions in a similar waterbody that is known to be unimpaired. This could involve comparison to individual reference sites, or comparison to a statistical summary of a large number of unimpacted sites. The statistical methods proposed by EPA in the nutrient criteria technical guidance manuals are thus a type of reference site approach, but one that is not informed by a careful consideration of site characteristics.

A general problem with using a Reference Site approach is that it does not directly establish a quantitative criterion. Rather, it provides information on the levels of nutrients that may be consistent with full support of uses. For this reason, the Reference Site approach is probably most useful in setting the Tier I/II criterion boundary, which is explained more fully in Sub-section 3.4.

The ***Empirical*** approach generally involves a statistical analysis of the relationship between measures of exposure (such as nutrient concentration) and a measure of effect (or intermediate measure that is assumed to track well with measures of effect). Such an approach can be powerful, and has been used with particular success in setting targets for lake phosphorus loading to control mean annual algal biomass. Empirical analyses, however, can provide misleading results if care is not taken to include or control for all the significant confounding factors. For instance, comparison of macroinvertebrate data and nutrient concentrations in Ohio streams appears to show a good correlation, with decreasing diversity associated with higher nutrient concentrations. However, a more sophisticated analysis shows that the observed impairment of benthic biota is more strongly associated with habitat degradation, and streams with high nutrient loads also tend to be streams in urban and agricultural areas that are subject to direct impacts on physical habitat (Ohio EPA, 1999). Thus, nutrient criteria based on the correlation between benthic macroinvertebrate condition and nutrient concentrations would not yield meaningful results. When properly conducted, an Empirical approach may be particularly appropriate for setting the Tier II/III criterion boundary, which is more fully explained in Sub-section 3.4.

A ***Modeling*** approach relies on developing a process-based relationship between stressors and targets, and can be implemented at varying degrees of sophistication. A Modeling approach has the advantage of making explicit the roles of different factors in controlling results; however, it too may yield incorrect results if significant confounding factors are ignored. Simulation models alone do not provide a firm and defensible foundation for criteria development. However, models are valid and useful tools for the nutrient criteria analysis for a number of reasons:

- Models can provide a process-based interpretation of observed data, thus helping to sort out multiple causes.
- Models provide a tool for generalizing from conditions at specific sites to conditions representative of typical conditions in a classification stratum.
- In some areas, very few "unimpacted" reference sites exist due to extensive human modification of the stream network and the addition of point and nonpoint loads.
- Many observed cases of impairment may be due to factors other than nutrients. For instance, poor biological integrity may be due to habitat alteration, while elevated periphyton concentrations in a low-order stream may be due more to removal of riparian shading than to nutrient levels. To attribute these impacts to nutrients could result in unnecessarily stringent criteria. Conversely, some lakes receive nutrient loads that would be sufficient to cause impairment due to eutrophication if it were not for the suppression of algal growth by high turbidity. Models provide a method for controlling these confounding factors.

In general, no single method provides all the answers for nutrient criteria development. It is therefore advisable to rely on a weight of evidence approach.

3.1 Methods for Lakes and Reservoirs

3.1.1 Statistical and Reference Site Approaches

Finding unimpacted reference sites for lakes and reservoirs in the arid west can be difficult because there are few natural lakes and most impoundments are heavily managed for water supply, causing changes in hydrologic regime and lake response. In Southern California, many impoundments store water derived from outside the basin, partially decoupling the waterbody from its ecoregion. This means that both the direct reference site and regional-statistical approaches are of limited use in establishing nutrient criteria.

For older lakes, one interesting possibility is to use internal reference conditions derived from sediment core analysis. Assuming the lake has not been dredged, sediment cores will often yield a dateable historic record that can provide information on types of algae, relative chlorophyll *a* concentrations, and nutrient concentrations (Whitmore and Riedinger-Whitmore, 2004). For a natural lake, the sediment record can reveal pre-impact conditions. For a more recent impoundment, the sediment record can reveal how conditions have changed over time, and potentially index acceptable target conditions to an estimate of nutrient loading rates.

3.1.2 EMPIRICAL APPROACHES

A number of empirical approaches to predicting lake response have been developed. In most cases these predict average chlorophyll *a* concentrations as a function of phosphorus (or phosphorus and nitrogen) loading and hydraulic residence time. Examples of these were discussed in Section 2. While expected to hold in general, these relationships have not, to our knowledge, been fully validated on California lakes.

3.1.3 MODELING APPROACHES

Response models for lakes and reservoirs are well developed and tested, with a long history of use. What is needed for nutrient criteria develpoment is a relatively simple tool that can be used to evaluate general conditions and responses for a lake or reservoir with a given set of general characteristics. The Army Corps of Engineers (ACOE) BATHTUB model (Walker, 1987) appears to provide an appropriate tool for evaluation of eutrophication responses in lakes. BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll-*a* concentrations (or algal densities), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance or nutrient loading model approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day or seasonal variations in water quality. Algal concentrations are predicted for the summer growing season when water quality problems are most severe. Annual differences in water quality, or differences resulting from different loading or hydrologic conditions (e.g., wet vs. dry years), can be evaluated by running the model separately for each scenario.

BATHTUB first calculates steady-state phosphorus and nitrogen balances based on nutrient loads, nutrient sedimentation, and transport processes (lake flushing, transport between segments). Several options are provided to allow first-order, second-order, and other loss rate formulations for nutrient sedimentation that have been proposed from various nutrient loading models in the literature. The resulting nutrient levels are then used in a series of empirical relationships to calculate chlorophyll-*a*, oxygen depletion, and turbidity. Phytoplankton concentrations are estimated from mechanistically based steady-state relationships that include processes such as photosynthesis, settling, respiration, grazing mortality, and flushing. Both nitrogen and phosphorus can be considered as limiting nutrients, at the option of the user. Several options are also provided to account for variations in nutrient availability for phytoplankton growth based on the nutrient speciation in the inflows. The empirical relationships used in BATHTUB were derived from field data from many different lakes, including those in EPA's National Eutrophication Survey and lakes operated by the Army Corps of Engineers. Default values are provided for most of the model parameters based on extensive statistical analyses of these data. Once a target level of chlorophyll *a* is assigned to the support a specific use (as an intermediate measure), these relationships can be used to derive loading criteria.

3.2 METHODS FOR RIVERS AND STREAMS

3.2.1 STATISTICAL AND REFERENCE SITE APPROACHES

The use of reference sites is more promising for streams than reservoirs in most of California, if only because unimpacted reference streams are somewhat easier to find. The approach is still subject to the general limitation of not providing an exact criterion. For developing criteria, the generalized reference site or statistical method proposed by EPA is of some relevance, but only when the analyses are conducted at an appropriate sub-ecoregional level.

3.2.2 EMPIRICAL APPROACHES

A number of investigators have proposed stream nutrient criteria based on empirical analyses of the relationship between periphyton chlorophyll *a* (for which a target density is assumed) and nutrient concentrations (Dodds *et al*., 1997; Dodds *et al*., 2002; Biggs, 2000). These approaches were used in large part because reliable modeling tools were not available. Their success in predicting observed benthic chlorophyll *a* density has been somewhat limited; however, they do appear to do a reasonable job of predicting the maximum potential density. Biggs (2000) showed that the prediction of maximum benthic chlorophyll *a* can be greatly improved by including a measure of days of accrual, indexing the frequency of scouring events in the stream. A detailed analysis of the use of these types of approaches in California was presented in Section 2.

3.2.3 MODELING APPROACHES

Nutrient response models in streams and rivers are less well-tested and less easy to generalize than lake models. For instance, the nutrient response in streams often involves a complex set of interactions between periphyton, macrophytes, stream scouring, and light availability. The general experience has been that predictive models require site-specific calibration for success.

For the purposes of criteria development relatively simple models are needed. An appropriately simple formulation for periphyton growth is available as part of the QUAL2K model (Chapra and Pelletier, 2003). On its own, QUAL2K has too many poorly characterized parameters to be applicable to criteria development without site-specific calibration. Tetra Tech has, however, recently demonstrated that the QUAL2K equations for periphyton growth can be "trained" to match the predictions yielded by the empirical approach of Dodds et al. (2002). Predictions can be further enhanced by adding a component to account for the effect of days of accrual, as described by Biggs (2000). This hybrid empirical/modeling approach shows considerable promise as a means for refining nutrient targets for streams and rivers.

3.3 NATURAL BACKGROUND NUTRIENT LOADING

Setting the nutrient criterion also requires estimation of natural background concentrations or loads. As with the target development, this can proceed via a weight-of-evidence approach using several different sources. The different approaches are summarized below.

Statistical and reference site approaches rely on observations of concentrations in unimpacted streams. As the determination of a background component is not directly linked to beneficial use support, this is probably the most appropriate use of the EPA statistical approach. However, such an approach should be undertaken on the sub-ecoregional level. Empirical methods refine the naïve statistical approach by taking into account a variety of stratification factors that help determine differences in natural background concentrations.

A modeling approach to natural background focuses on nutrient load generation and transport: Given the natural land cover, topography, geology, and meteorology of a watershed, what is the expected distribution of nutrient concentrations and loads at various points in the stream network? Answering this question requires combining a watershed load generation model with a stream transport model that can account for losses in transport. For the interpretation of reference watershed loading, the SWAT model from USDA's Agricultural Research Service (USDA/ARS) (Neitsch *et al*., 2001) is attractive because it is explicitly designed to take land cover into account. Stream transport is a highly site-specific phenomenon that can be difficult to capture reliably in a generic application of a process-based mechanistic model. For a general analysis of natural background, it is preferable to use the empirical/statistical description of transport losses contained in the USGS SPARROW model (Smith *et al*., 1997), which describes in-stream losses as a function of flow regime and travel time, plus a factor for reservoir retention.

3.4 TIERED CRITERIA APPROACH

One of the major challenges to the development of nutrient criteria is finding a balance between the protection of uses and imposition of economic hardship on dischargers and agriculture. Responses vary greatly among waterbodies. Setting a generic criterion number that is too low can result in high costs for an action that may not have any associated benefits in a given setting. Setting a generic criterion number that is higher may result in more realistic and cost-effective management in less sensitive areas, but could allow degradation to occur in more sensitive areas. Selecting a single ecoregional target criterion based solely on statistical analysis basically ensures that the target will not be appropriate for many individual streams.

Further, if the statistical target is picked as a low percentile, this approach would ensure that many waterbodies' uses are over-protected at high cost and with little benefit.

One way to solve this problem would be to undertake a detailed analysis that resulted in appropriate site-specific criteria. While technically ideal, this labor-intensive approach is obviously infeasible for the vast majority of the waters of the state.

Development of risk-based criteria, related to both the benificial uses and physical characteristics of individual streams – as described in the previous sections – will go a long way to addressing these types of problems. However, it will not completely avoid them. On the one hand, the technical ability to predict responses to nutrient loads and the understanding of confounding factors is clearly limited. On the other hand, the costs of significant reductions in nutrient loads from both point and nonpoint sources are usually high.

Most water quality criteria for the protection of aquatic life address acute or chronic toxic stressors. For acute toxicants, laboratory tests are used to establish a species sensitivity dose-response curve, showing how increased concentrations lead to incremental loss of sensitive species. This provides a clear and quantitative basis for setting water quality criteria. A similar procedure, although often involving much greater uncertainty, is used for chronic effects through analysis of concentration effects on recruitment, growth, or other factors.

This approach does not work well for the majority of the deleterious impacts of nutrients, largely because of the complex chain of interactions that connects nutrient loads to species response. In some cases it is feasible to construct a dose-response curve for intermediate measures (for instance, relating survival of sensitive benthic invertebrate species to density of algal biomass). However, the relationships back to the ultimate stressor – nutrient loads – become much more inexact and subject to confounding effects.

Given these issues, it is generally not appropriate to establish a single criterion number for nutrients, even on an ecoregional basis, and even when linkage analyses are used to help account for confounding factors that may allow more or less nutrients to be present without impairment. Instead, nutrient criteria are better evaluated via a tiered approach. In relation to a given use, there are three natural tiers for evaluation of nutrient impacts:

Tier I. Impacts unlikely (use is supported).

Tier II. Probably sustaining (but potentially threatened).

Tier III. Impacts likely (use is not supported or highly threatened).

There are two boundaries between the three tiers, both of which may be thought of as criterion numbers. These are:

Tier I/II Criterion: Conceived foremost as a concentration or load below which impacts are unlikely. However, the Tier I/II Criterion should also take into account the natural background concentration likely to be present, and should not be set less than the natural background. If natural background results in an impairment of a benificial use, then the use designation is inappropriate and should be refined.

Tier II/III Criterion. This is a concentration or load above which an impairment of the use is highly likely, and should be set at the subecoregional level where possible.

Selection of the Tier I/II and Tier II/III criteria (relative to a specific use) could be made based largely on expert opinion, with supporting modeling analyses. In addition, the Tier I/II criterion could incorporate an analysis of subecoregional background, based on a statistical analysis of minimally impacted sites. The objective in setting these criterion levels should be to achieve a clear consensus as to the risk of impairment being acceptably low (Tier I/II) or unacceptably high (Tier II/III). The boundaries should not be set so tightly that inappropriate assessments are made (e.g., a waterbody is assessed as impaired and costly nutrient management strategies are implemented when actual impairment of the specific waterbody is unlikely to occur.)

Many individual waterbody segments will fall into the “gray area” of Tier II, in which impairment is a reasonable possibility, but not a certainty. These are the waterbodies for which a more detailed and site-specific analysis is warranted. The first step in processing candidate Tier II waterbodies would be to develop site-specific targets, using linkage analysis tools (e.g., models) that help account for site characteristics. If such a site-specific target is exceeded, the waterbody could then be moved to the Tier III category (likely nonsupporting). Where the site-specific target is met, the waterbody could remain in the Tier II (probably sustaining) category, but an antidegradation analysis would be warranted in connection with any proposed changes in permitted discharges.

Figure 3-1 presents a decision diagram summarizing how a tiered approach would work, from the perspective of use assessment based on nutrient concentrations. It should be emphasized, however, that other direct indicators of nutrient-associated impairment (measured algal biomass above a target level for the use, diurnal excursions of the DO criterion attributed to algal respiration, and so on) should supersede the need to make an assessment of impairment based on nutrient concentrations. In such cases, the tiered approach would still remain relevant to evaluating the loading reductions necessary to achieve support of a use (through a TMDL).

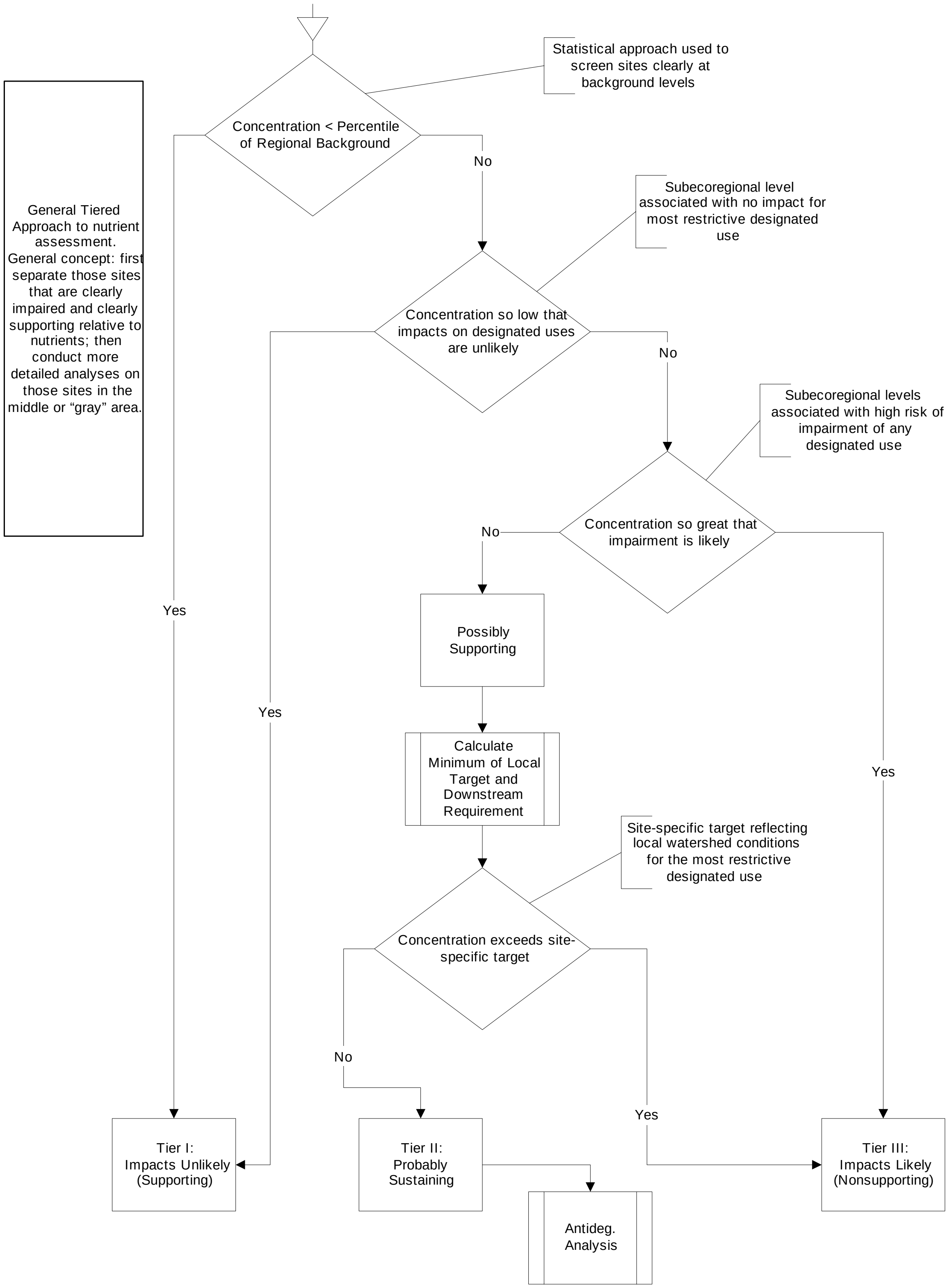


Figure 3-1 Tiered Approach to Nutrient Criteria Implementation

4.0 ANALYSIS OF NUTRIENT DATA FROM ECOREGION 6

4.1 DATA OBTAINED

Usable data was obtained from 712 unclassified stations and 79 minimally impacted stations over Ecoregion 6 distributed as shown in Figure 4-1. The main sources of the data are identified in Table 4-1. Important sources of data were the Regional Boards (2, 3, 4, 5, and 9) the USGS, the Central Coast Ambient Monitoring Program, and the Army Corps of Engineers. Although other sources of data no doubt exist within the ecoregion, we still feel confident that we have a fair representation of stations over this area, and that individual sub-regions are not under- or over-weighted. Future data collection may address some other remaining sources of data.

Table 4-1
Major sources of data in Ecoregion 6

Agency	Number of Stations
USGS	216
RWQCB 4	179
RWQCB 3	151
Central Coast Ambient Monitoring Program	134
RWQCB 5	55
Corps of Engineers	50
Orange County	25
Department of Water Resources	24
UCLA	18
RWQCB 9	12
RWQCB 2	6
Monterey County	6
Heal the Bay	2

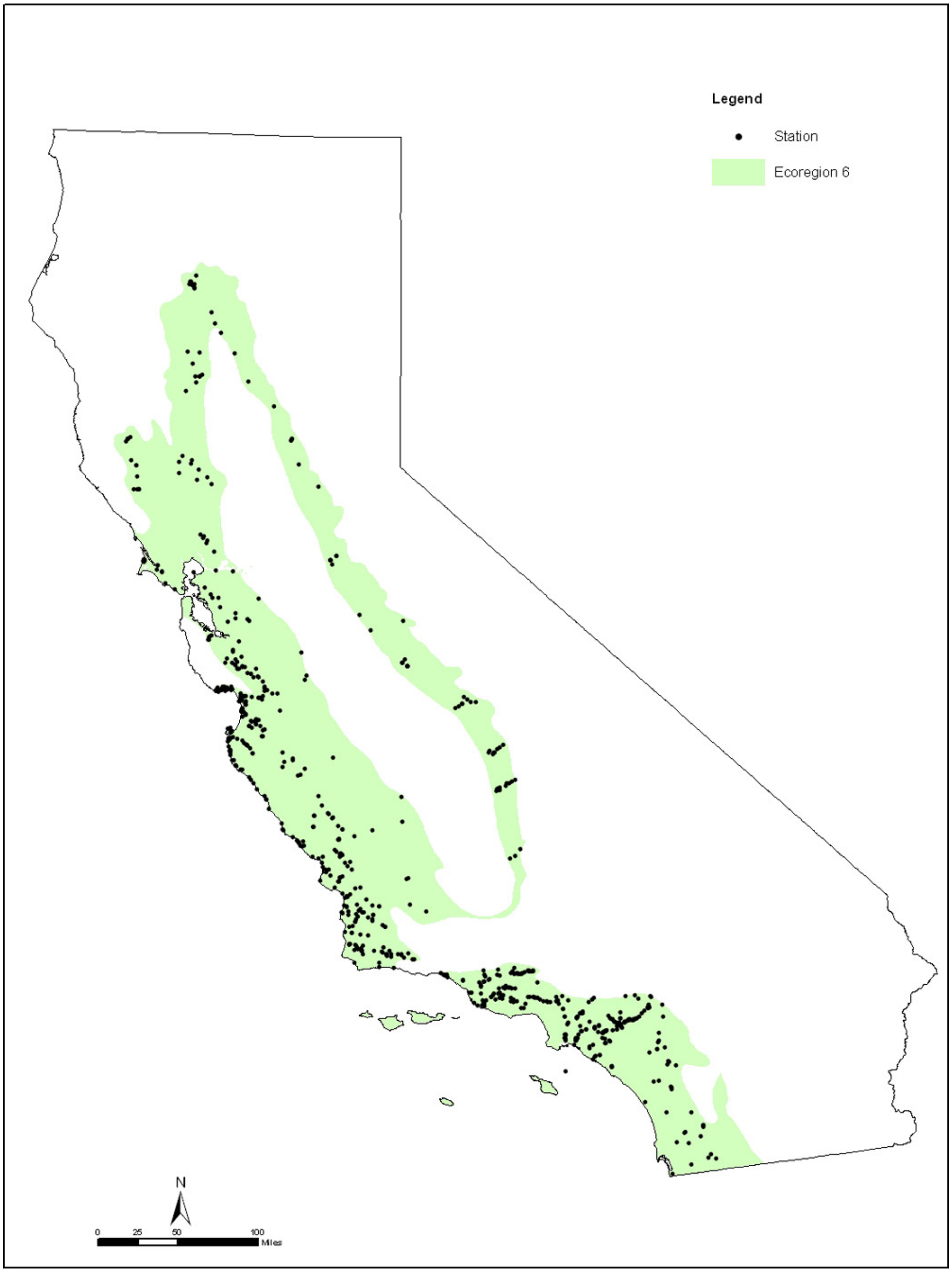


Figure 4-1 Distribution of stations with nutrient data in Ecoregion 6

Data on the following nutrient parameters were obtained with sufficient frequency to be used in the analysis that follows: NH_3, NO_2, NO_3, TKN, PO_4, and TP. All data were converted to represent the constituent in units of mg of nitrogen per liter or mg of phosphorus per liter. Thus, when data were reported for total nitrate in mg/l, this was multiplied by 14/62 to convert to units of nitrate as nitrogen in mg/l. This was done to obtain consistency across data sets from different sources.

4.2 DATA LIMITATIONS

Even though large quantities of data were collected through the effort described earlier, we must still point out some limitations inherent to this approach. These are likely to be significant in any nutrient-related data collection over a large region, and must be considered in future evaluations of nutrient criteria that are based on existing datasets from multiple sources.

Few stations had sufficient co-located information on biology. A small number of stations did report values of chlorophyll-*a*, dissolved oxygen, Secchi depth, and turbidity, but these data were insufficient to carry out a region-wide analysis. This finding is similar to what was observed in our earlier pilot study (Tetra Tech, 2000). Data on other metrics that characterize algal or benthic communities, such as diversity or percentage of diatoms, was even more rare, and it is unlikely that such data will exist except in small, localized research studies that limit their wider applicability in nutrient criteria development.

No information on watershed characteristics. Descriptions of stations (watershed, land use, residence times, etc.) were requested during data collection, but almost no sites had enough characterization information available. To a certain degree, this shortcoming can be addressed for stream stations in the future by the availability of high resolution digital elevation data, that can be used to calculate the watershed for each station in the database, and in conjunction with land cover and soil characteristics maps can be used to define the land use characteristics for each station. This was done in a preliminary manner for all the stations in the database, but will be the subject of more detailed analysis in the coming year. For lake stations, key nutrient-related parameters such as the depth and residence time must be obtained on a site-by-site basis.

Uneven data density. Because the sampling conducted by different entities has different objectives, some stations are found to have substantially more data than other stations. If a population of all available data is pooled, there is the potential for the population to be biased by the existence of some stations with a large number of contributing datapoints. This is likely to be a feature of datasets across regions that combine data from multiple sampling programs with multiple objectives.

The same set of parameters was not measured. Although we have identified NH_3, NO_2, NO_3, TKN, PO_4, and TP as the parameters that are most widely reported, not all of these constituents are measured as often or as uniformly. For example, a subset of stations may have a lot of data on NH_3, but less information on the other constituents.

This is again a consequence of there being multiple objectives underlying the data collection.

Data do not cover the same time period. All available data from stations within Ecoregion 6 was requested from the individuals/agencies contacted. For some stations the data record goes back several decades, but more commonly the data record single stations for a finite duration, and different stations may have been monitored over different time periods. For example, one station in the database may contain data from data from 1975 to 1985, another station may contain data from 1990 to the present. When data from such stations are pooled, there may be unknown influences such as weather or changing land use that are not easy to account for in a database of several hundred stations.

Limited number of stations identified as minimally impacted. One of the requests made of the contacts was for a list of stations that could be classified as minimally impacted, as defined earlier. These stations could be used as a comparison against the general population of stations. However, this effort did not yield a large number of stations. In part this may be because there are few minimally impacted stations in Ecoregion 6, and in part it may be because agencies/individuals do not have the basis to perform this characterization.

Not using the same methods. It was assumed for the purpose of this study that the data provided to us used commonly accepted methods for analyzing various nutrient constituents. It is possible that there are systematic differences across agencies that use slightly different methods. However, such an analysis was beyond the scope of this work.

Some of the limitations identified following the data collection have been addressed in the data screening procedures defined below. However, other limitations are fundamental to the data collection process, and must be considered as regulatory or policy decisions are based on them.

4.3 Screening of Data

Before any analysis, the data were subject to the following transformations:

- Data records reporting values below the minimum detection limit (MDL) for a constituent were replaced by the MDL for that constituent as reported by the source collecting the data.

- Data records prior to 1980 were not considered in the analysis because of possible confounding effects because of the use of different methods and/or changing conditions in the water body.

- Data records that reported concentrations in excess of 50 mg/l for a single constituent were treated as outliers and discarded from the analysis. This resulted in the removal of about 20 data points.

- In the original dataset, the number of data points per station is highly uneven, with some stations reporting thousands of measurements, and some stations reporting less than 10 measurements. To account for this unevenness in the data, we used one data point per nutrient metric per station per month. Although this cannot completely correct for the bias in the uneven data collection, it does prevents undue weight being given to a small number of stations. The number of monthly data points for different stations is mapped in Figure 4-2. The map shows that there are a large number of stations with sufficient data in the Central Coast. A handful of stations in the San Francisco Bay area and the Los Angeles area seem to be well characterized with a large number of data points.

4.4 CLASSIFICATION OF STATIONS

Stations were first classified as to whether they fell in a stream or lake because this information was not always provided with the source data. This was done by comparing the station coordinates with a GIS layer of water bodies in California. The site description associated with each station, where available, was also used for verification. As a result of this process, of the general population of streams, 101 stations could not be classified, 28 stations fell in bays, 98 stations were in lakes, and 484 stations fell in streams. Of the minimally impacted stations, 2 stations were not classified, 5 were in lakes, and 71 were in streams.

Stations that were classified by the data providers as being minimally impacted were considered separately in the analysis. The remaining stations were classified into three categories: unimpaired (i.e., meeting all beneficial uses), impaired by nutrients, and impaired by factors other than nutrients. This was based on a GIS mapping of the station coordinates over maps of impaired and unimpaired waters obtained from California and US EPA sources. Points that were <1 mile from the water body were considered to lie within the water body to account for errors in geographic positioning of various geographic data sources. This classification is shown in maps in Figures 4-3 through 4-6, where the station locations are overlaid on 1) a land use map of the state of California, and 2) a map identifying water bodies as unimpaired or impaired (by nutrients or non-nutrient factors). The minimally impacted stations are also shown on these maps.

The numbers of minimally impacted, impaired, and unimpaired stations, for each water body type are as follows:

Water Body Type	Stream	Lake	Bay
Minimally Impacted	71	5	-
Unimpaired	218	75	21
Impaired (nutrients)	81	2	-
Impaired (non-nutrients)	185	21	8

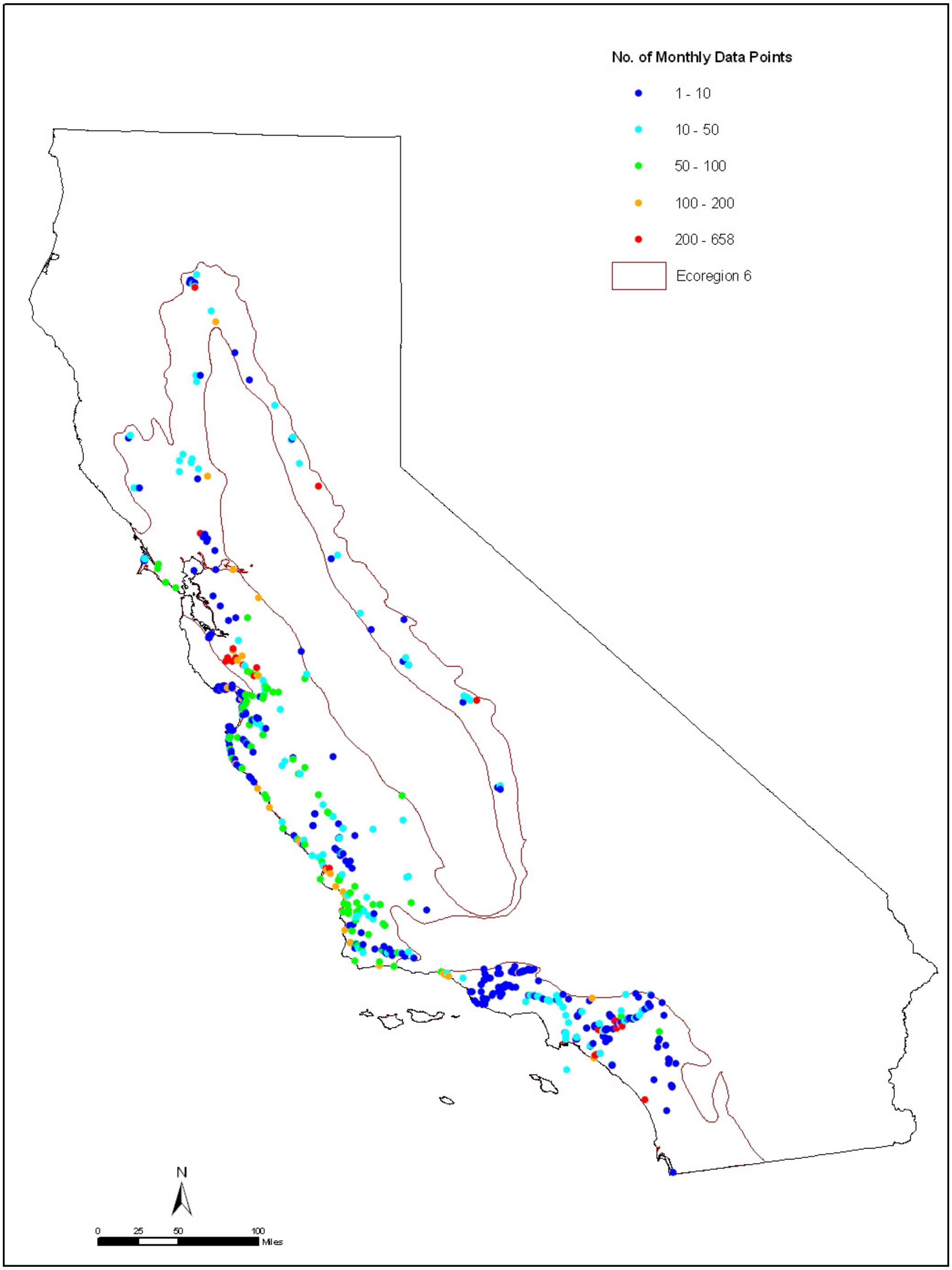


Figure 4-2 Number of monthly data points with nutrient data in different stations of Ecoregion 6

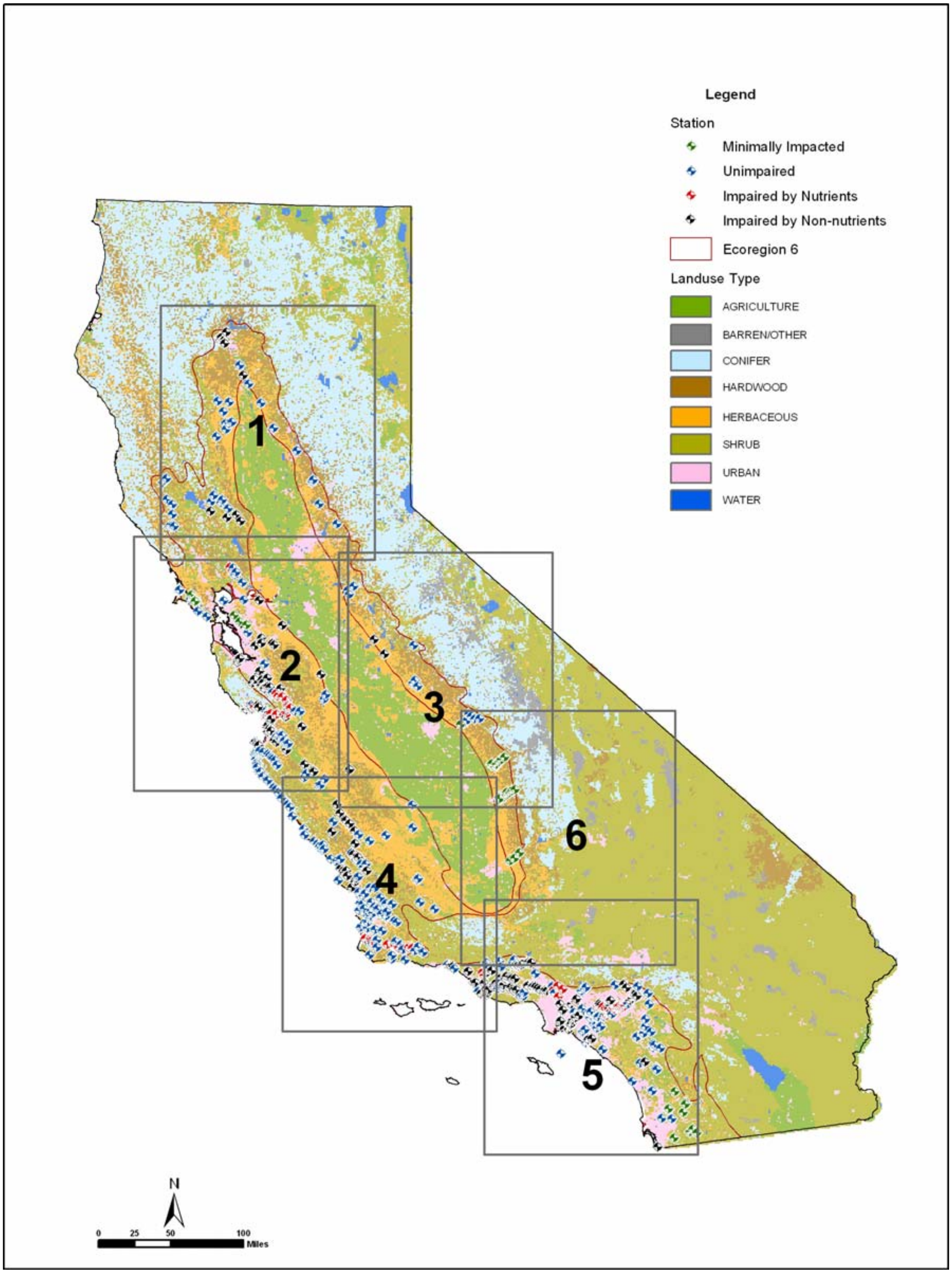


Figure 4-3 Stations classified as minimally impacted, unimpaired, impaired by nutrients, and impaired by non-nutrients across Ecoregion 6, overlaid on a map of land use.

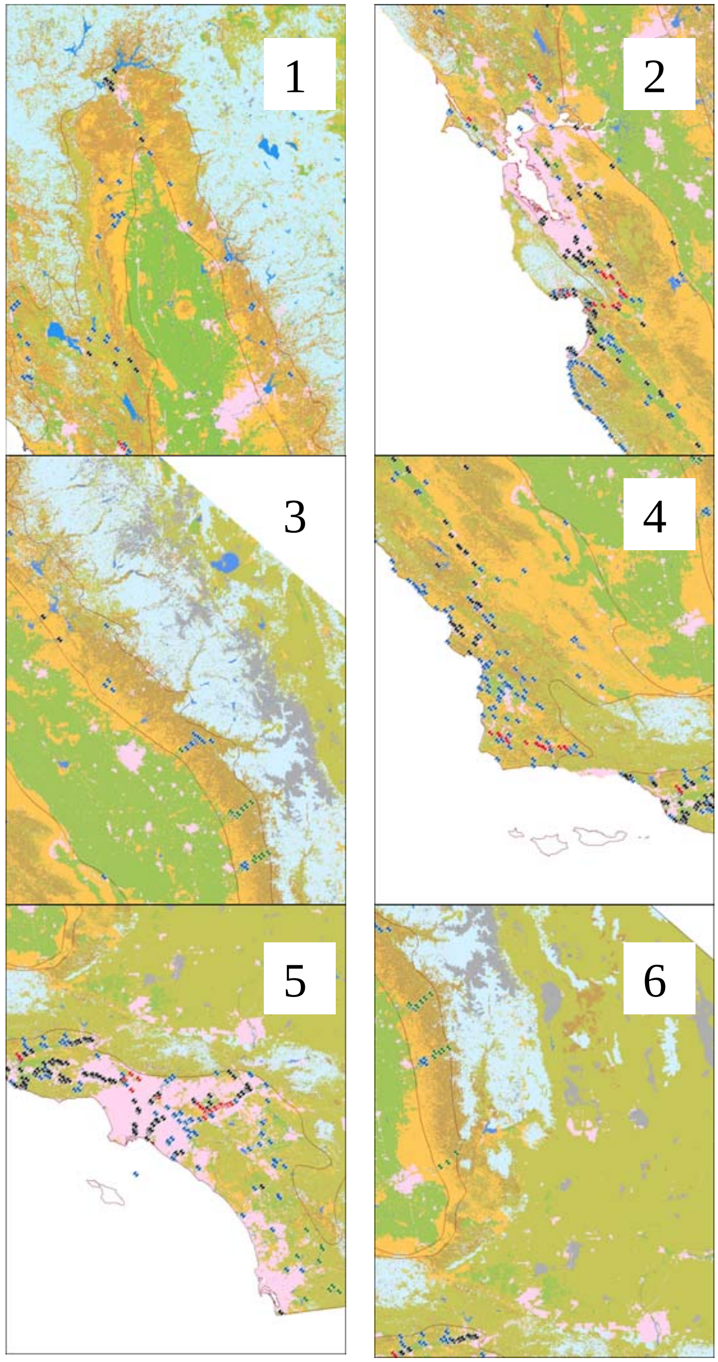


Figure 4-4 Stations classified as minimally impacted, unimpaired, impaired by nutrients, and impaired by non-nutrients across Ecoregion 6. Same as map in Figure 4-3 but focused on different parts of Ecoregion 6. The colors associated with the symbols and the land use coverages are shown in Figure 4-3. The map numbers correspond to the areas shown in Figure 4-3.

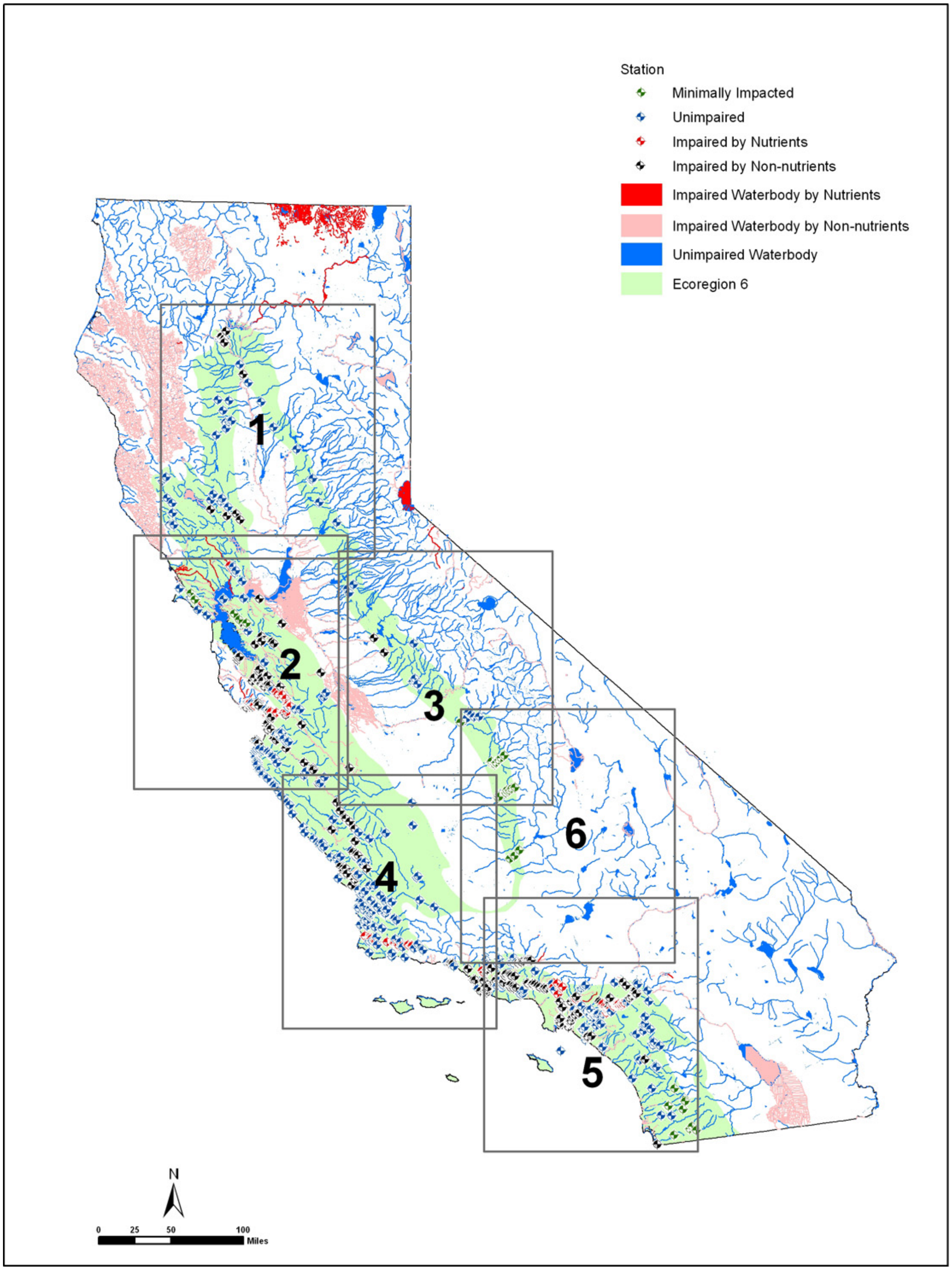


Figure 4-5 Stations classified as minimally impacted, unimpaired, impaired by nutrients, and impaired by non nutrients across Ecoregion 6, overlaid on a map of identifying water bodies as unimpaired and impaired by nutrients and non-nutrients.

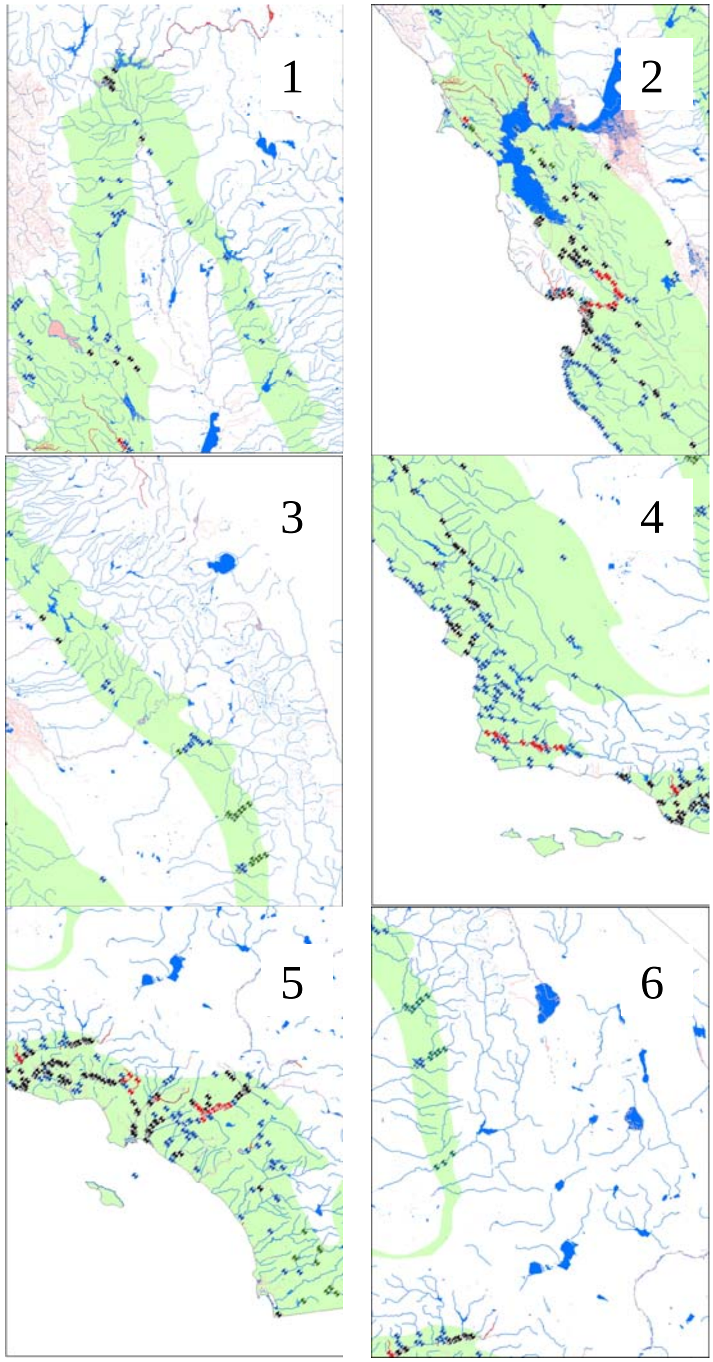


Figure 4-6 Stations classified as minimally impacted, unimpaired, impaired by nutrients, and impaired by non nutrients across Ecoregion 6, overlaid on a map identifying unimpaired and impaired water bodies. Same as map in Figure 4-5 but focused on different parts of Ecoregion 6. The colors associated with the symbols and the water bodies are shown in Figure 4-5. The map numbers correspond to the areas shown in Figure 4-3.

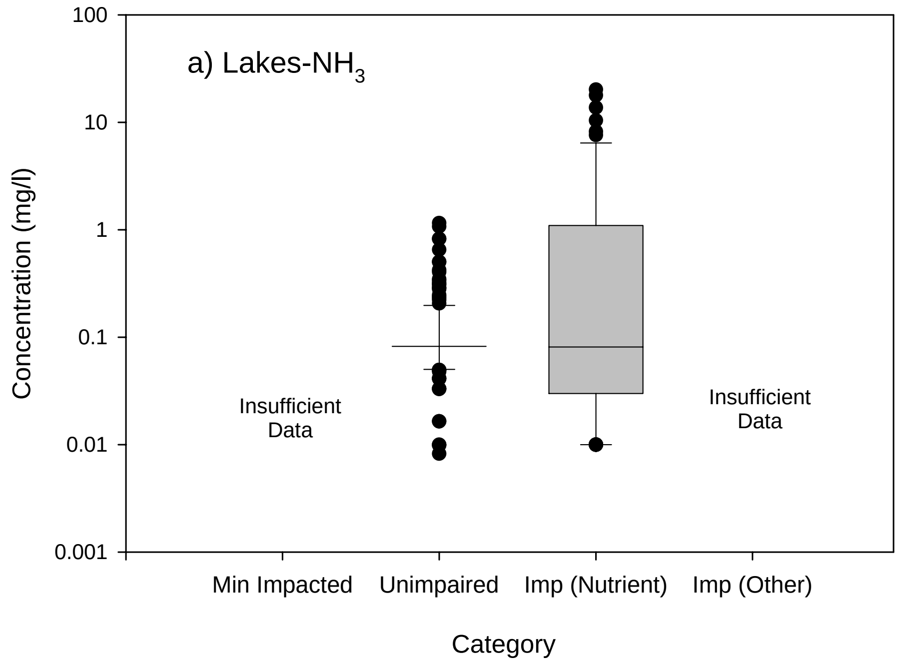
4.5 BOX PLOTS OF DATA

Utilizing the data screened as described above, and using only the stations in lakes and streams that have been categorized as minimally impacted, impaired, and unimpaired stations, results in a subset of nearly 22,000 data points with NH_3, NO_2, NO_3, TKN, PO_4, or TP data. To present these data in a way that aids comprehension, values for each nutrient constituent in each category of water body were represented by box plots. These plots are useful because they show key features of the distributions that have earlier been considered important in nutrient criteria development, i.e., the 25th percentile, the median, and the 75th percentile of the data.

Data are shown in Figures 4-7 through 4-18 for NO_3, NO_2, NH_3, TKN, PO_4 and TP for either lakes or streams with stations being classified as minimally impacted, unimpaired, impaired by nutrients, and impaired by agents other than nutrients. These data are also summarized in Tables 4-2 through 4-5. The main findings from this analysis are as follows:

- The data are highly variable for all categories of water bodies and for all nutrient constituents, spanning several orders of magnitude in many cases. Given the variety of natural and anthropogenic sources of nutrients, and the role of runoff in transport, this result is not surprising. From the standpoint of nutrient criteria development, this result is important because it provides a basis to relate the unimpaired and minimally impacted station variability into the criteria. Figure 4-19 shows the variation in standard deviations across different classifications of streams. Unimpaired water bodies have lower standard deviations of nitrate and TP, but standard deviations of TKN and PO_4 do not differ significantly.

- If we believe that nutrient concentrations can be directly related to impairment, we expect to see a pattern in these plots, with the lowest concentrations in minimally impacted water bodies, higher nutrient concentrations in unimpaired water bodies, and still higher concentrations corresponding to impaired water bodies. For streams, this relationship was found to be strong for the case of NO_3 and PO_4, somewhat significant for TKN and NH_3, but was not seen for the other two constituents, TP and NO_2. For lakes, the dataset we were working with was much smaller, and the trends were harder to discern. There appeared to be an effect of nitrate concentrations on impairment, albeit weaker than what was observed for streams. The behavior with respect to phosphorus was counterintuitive, with lower concentrations being seen in impaired water bodies.



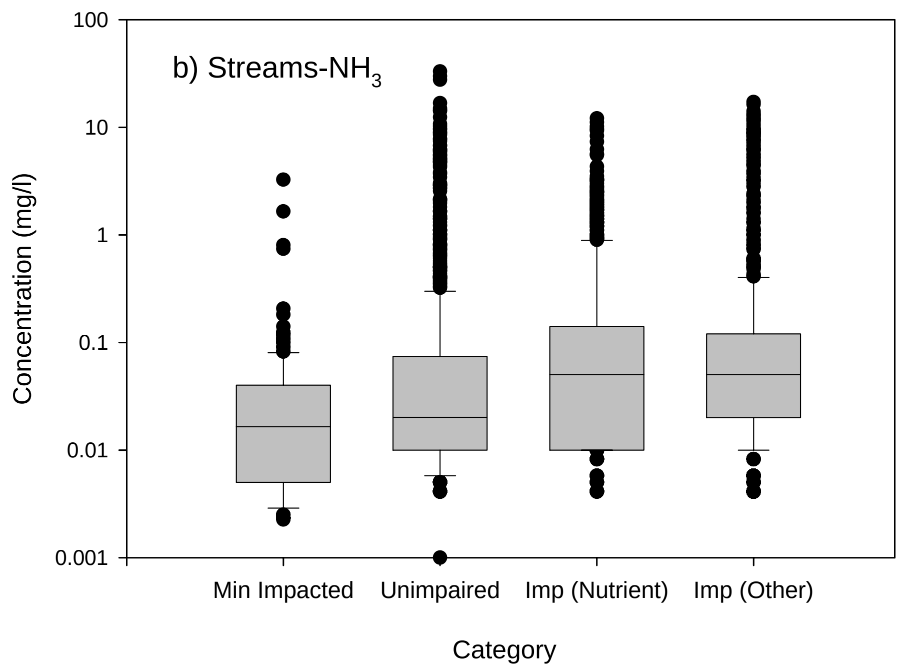
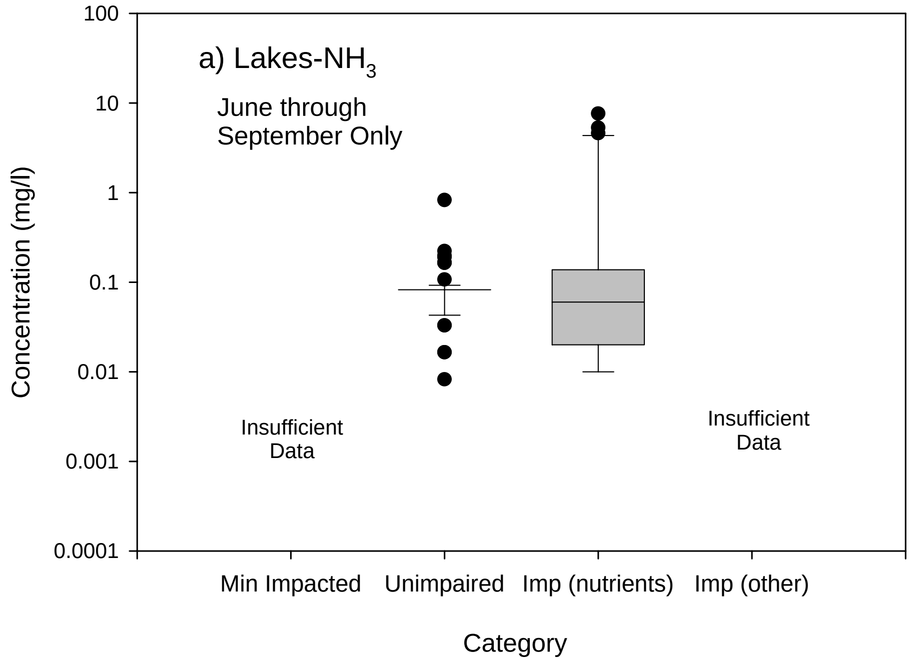


Figure 4-7 NH_3 levels across the whole year in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10^{th} and 90^{th} confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.



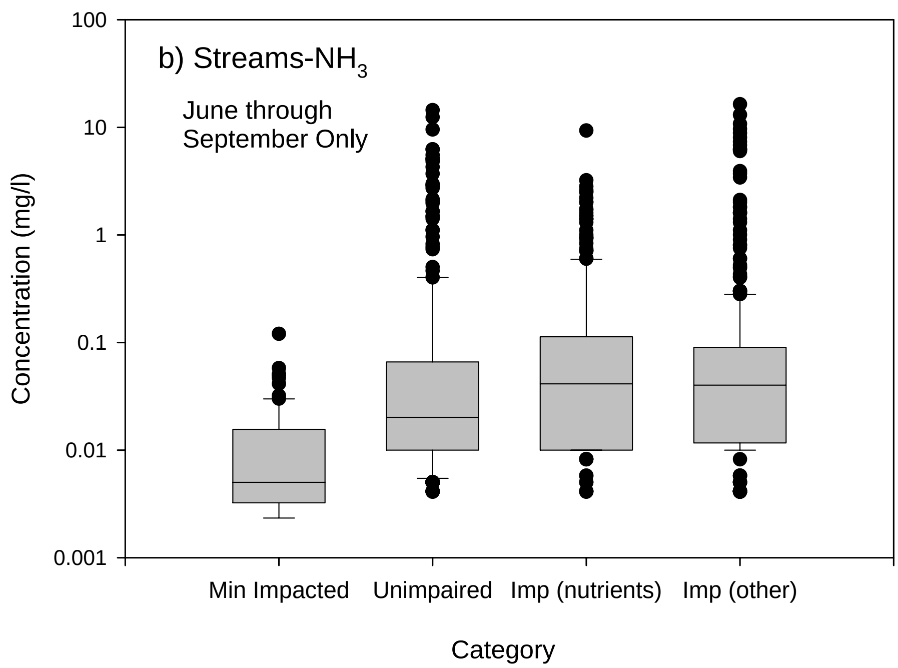


Figure 4-8 NH_3 levels from June to September in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

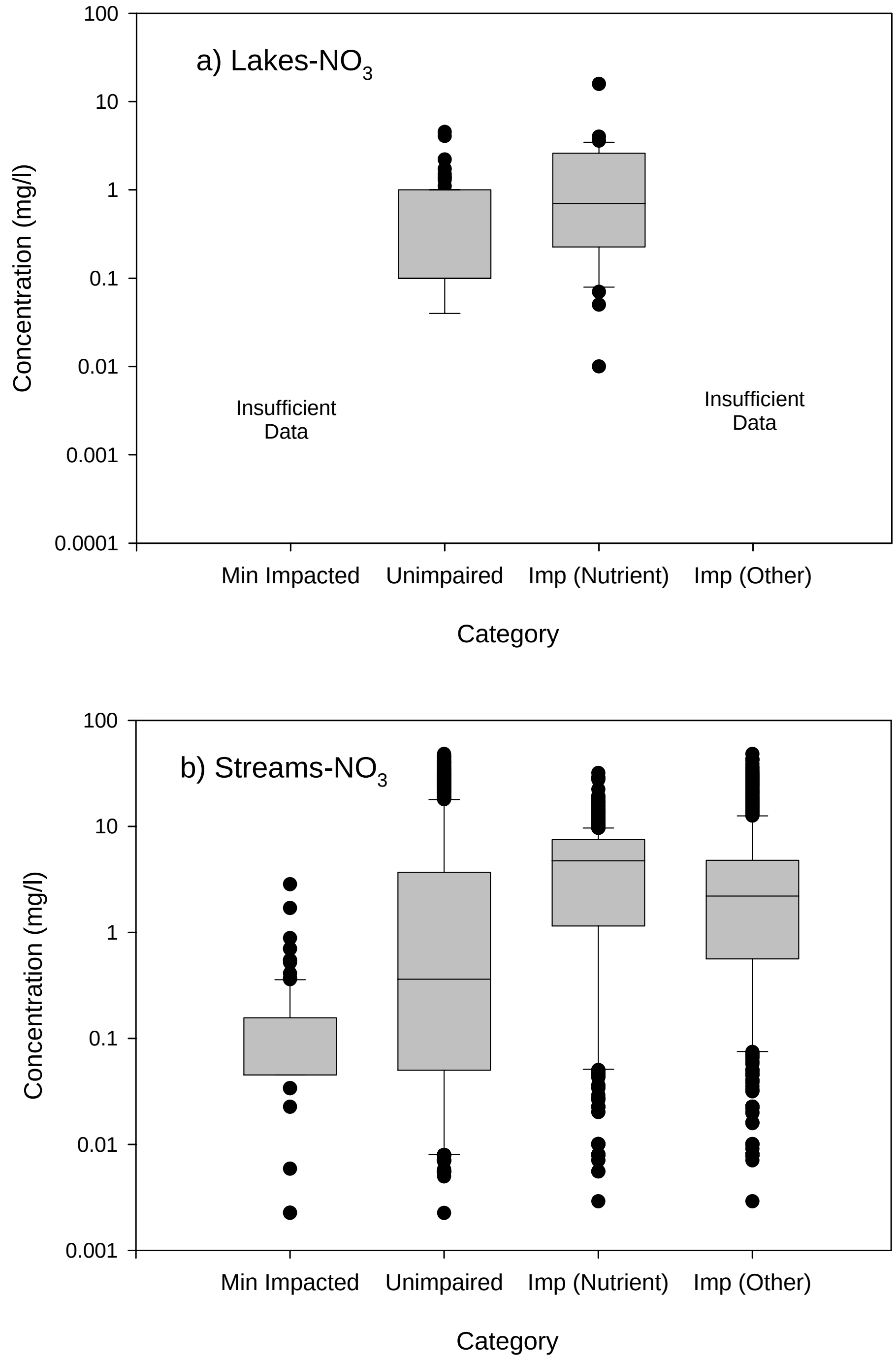
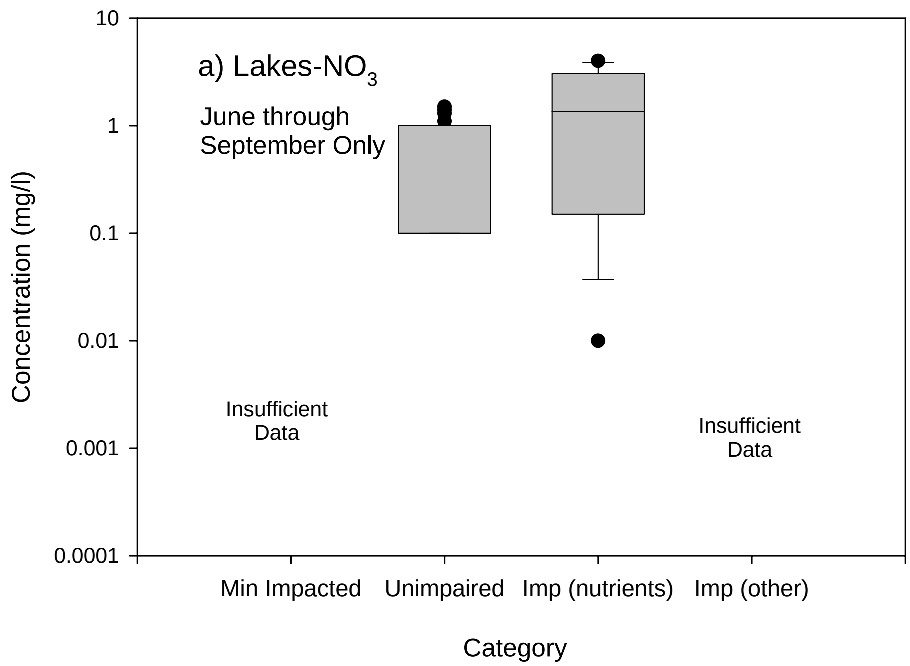


Figure 4-9 NO_3 levels across the whole year in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.



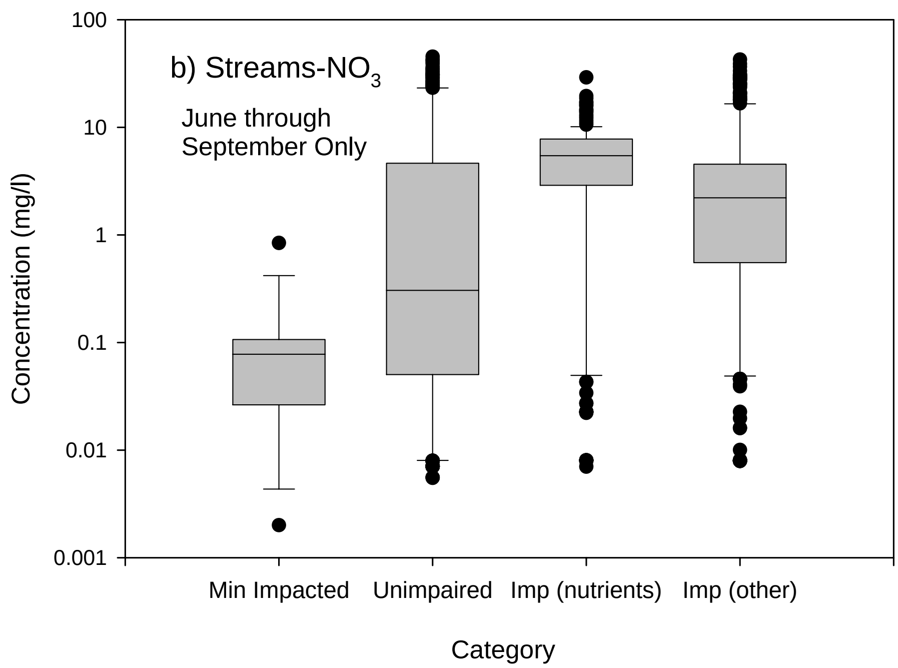


Figure 4-10 NO_3 levels from June to September in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

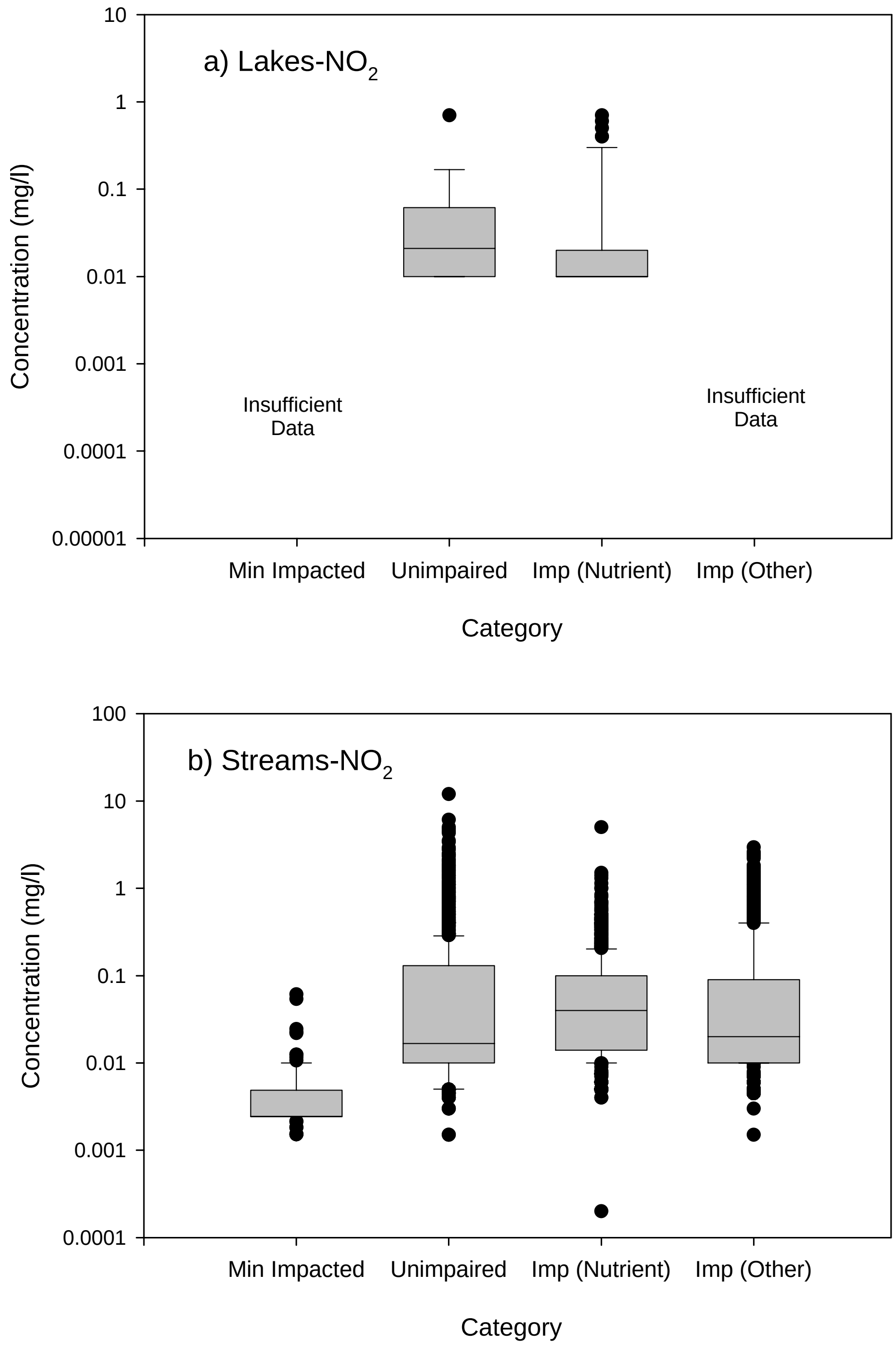


Figure 4-11 NO_2 levels across the whole year in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10^{th} and 90^{th} confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

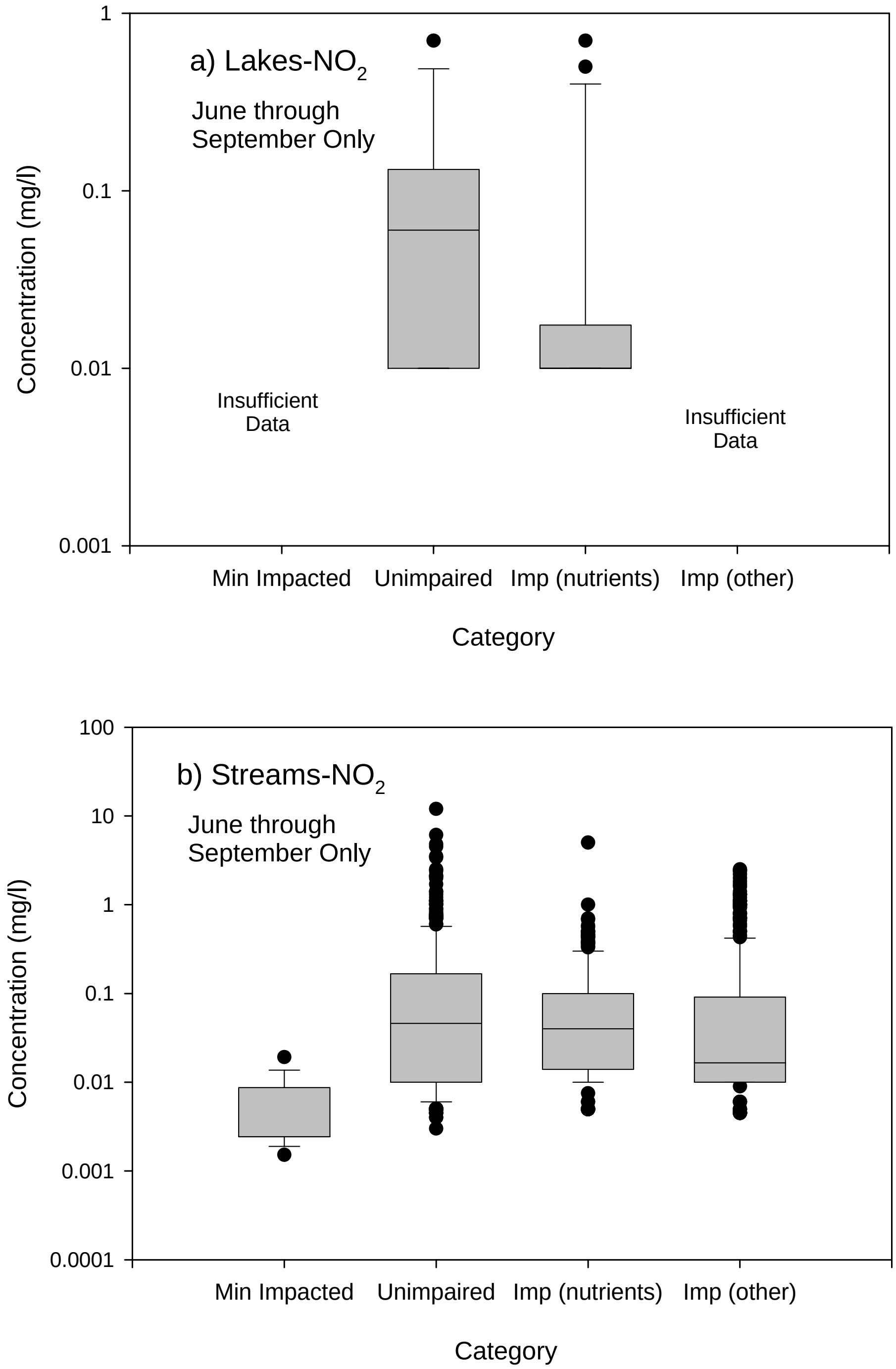
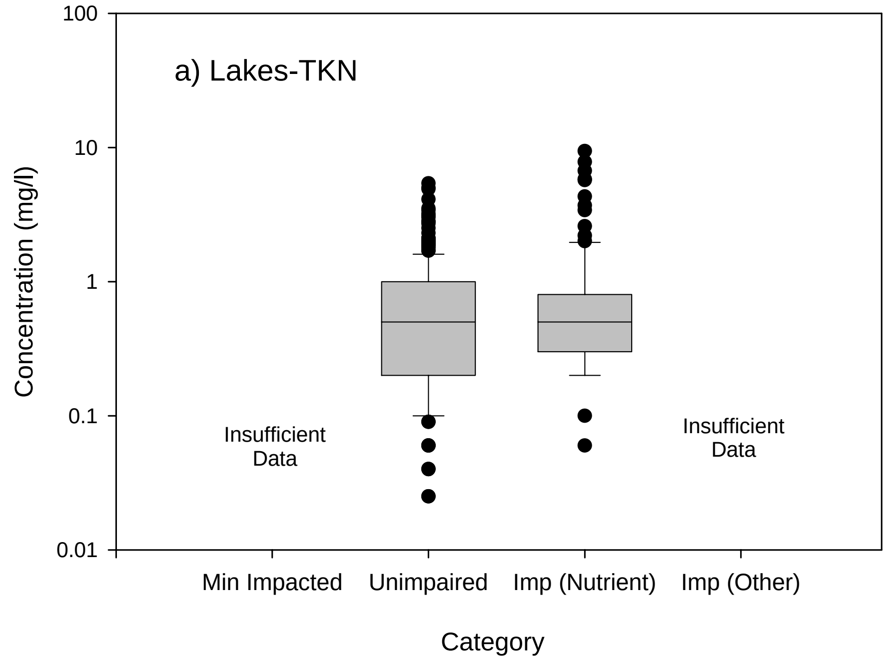


Figure 4-12 NO_2 levels over June through September in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.



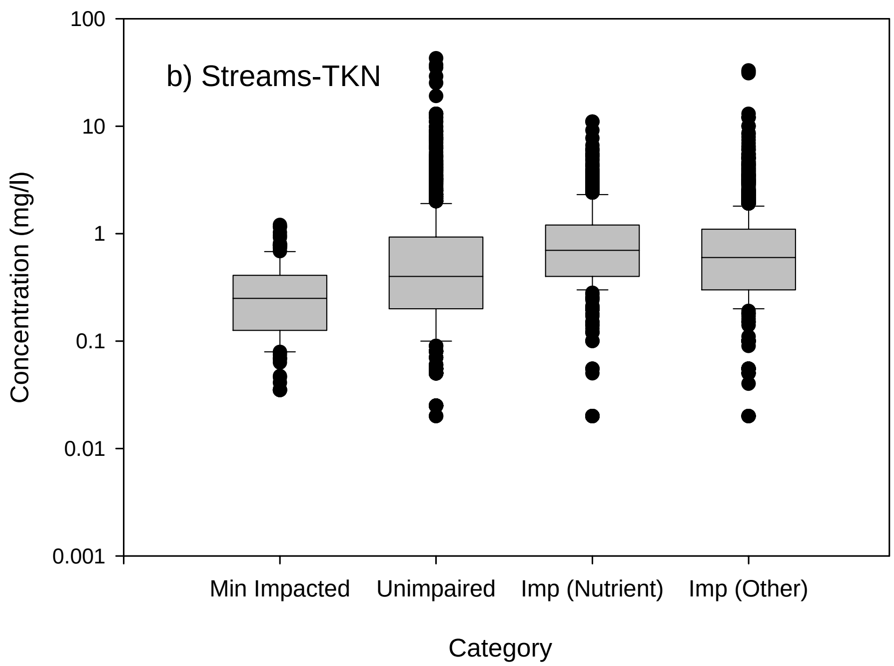


Figure 4-13 TKN levels over the whole year in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

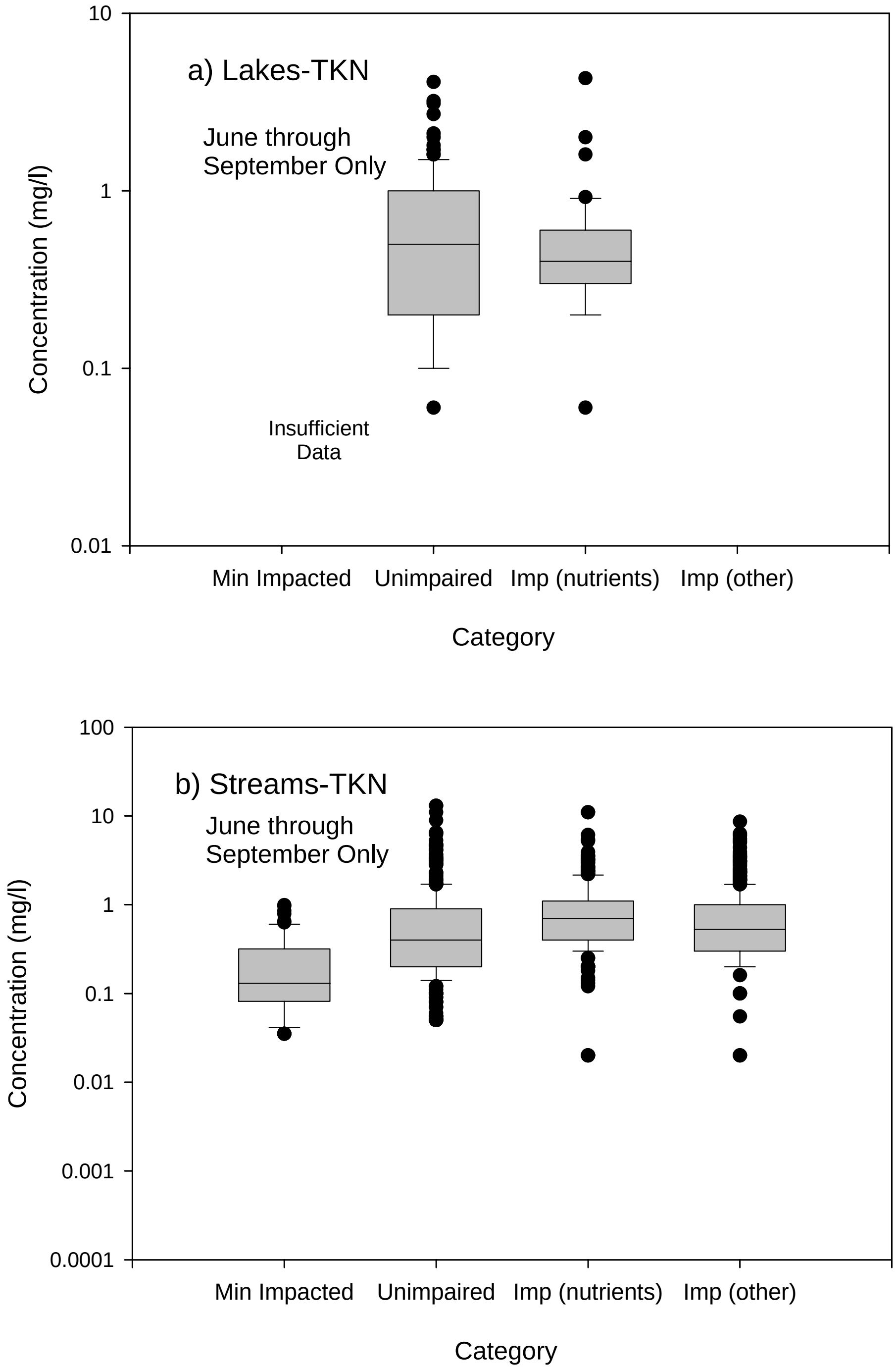


Figure 4-14 TKN levels over June through September in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

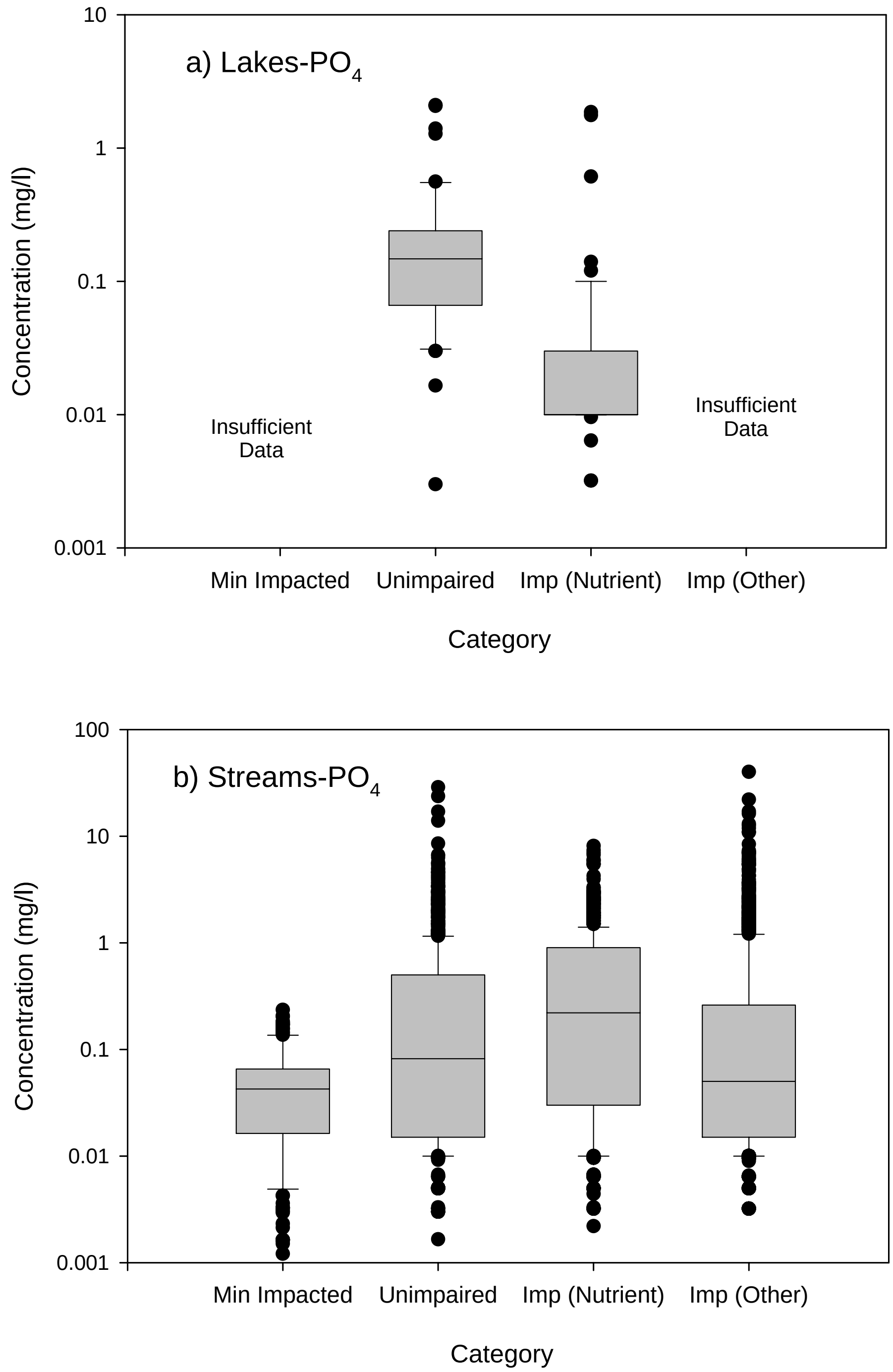


Figure 4-15 PO_4 levels across the whole year in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10^{th} and 90^{th} confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

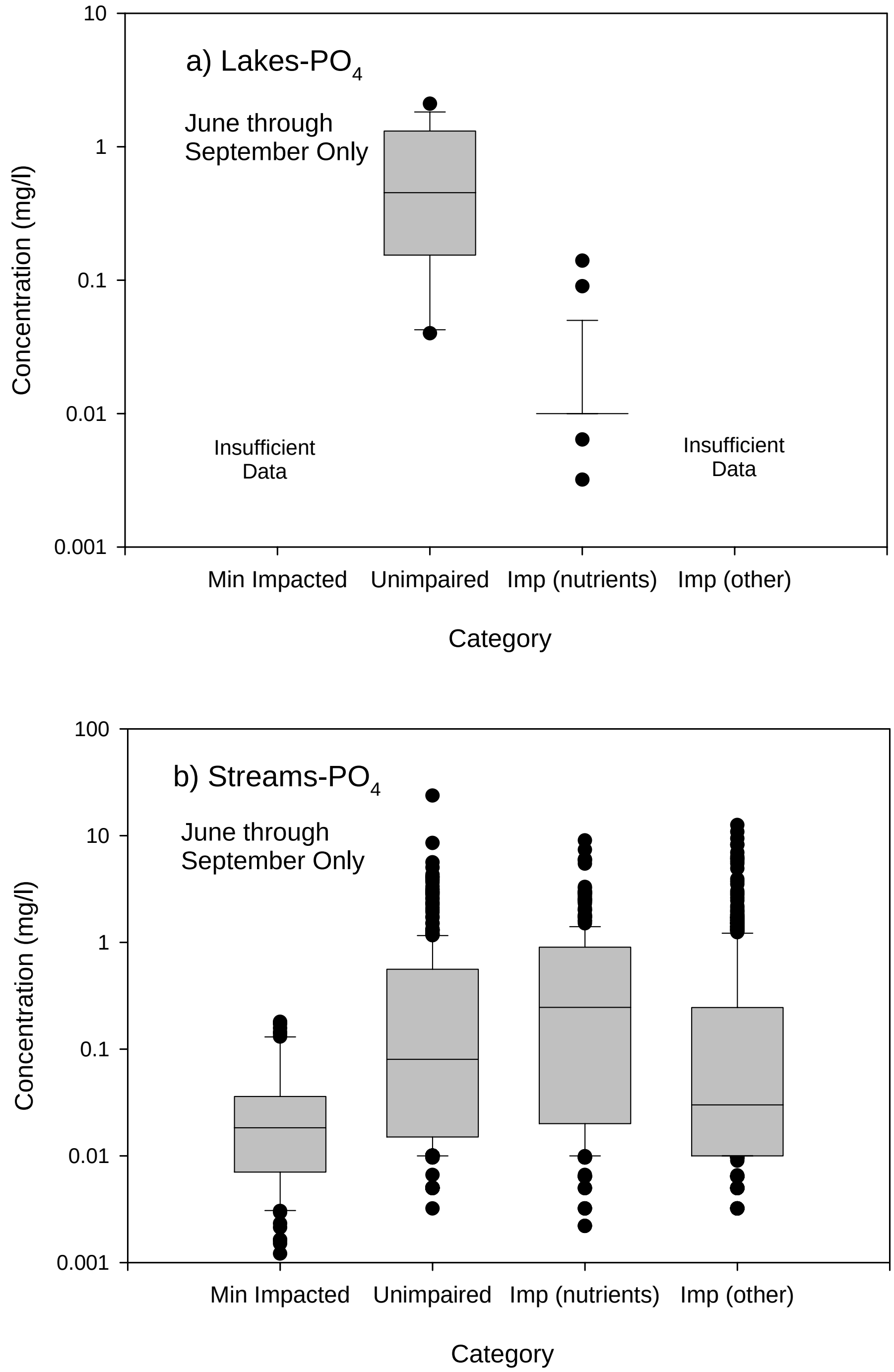
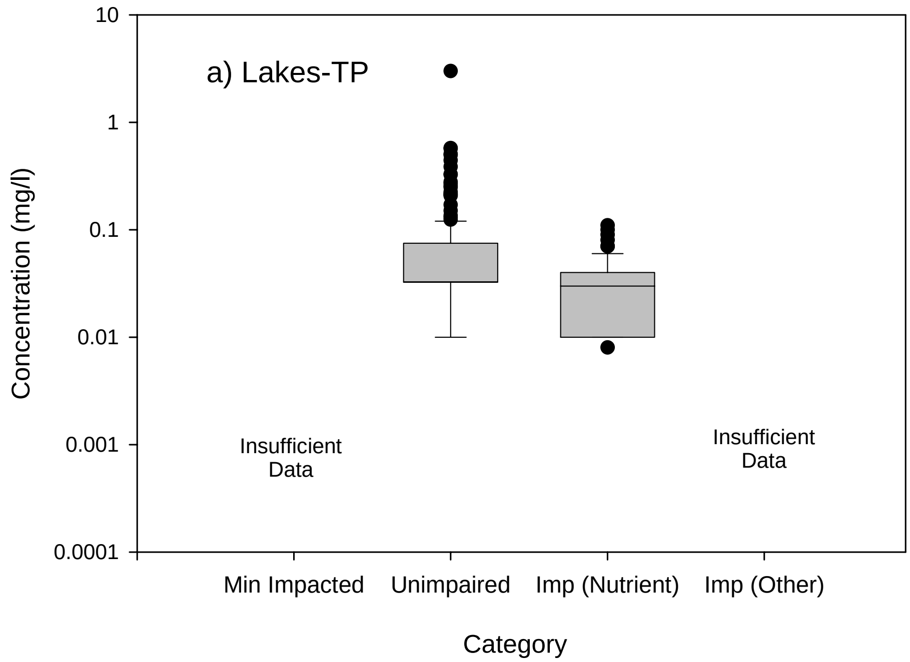


Figure 4-16 PO_4 levels over June through September in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.



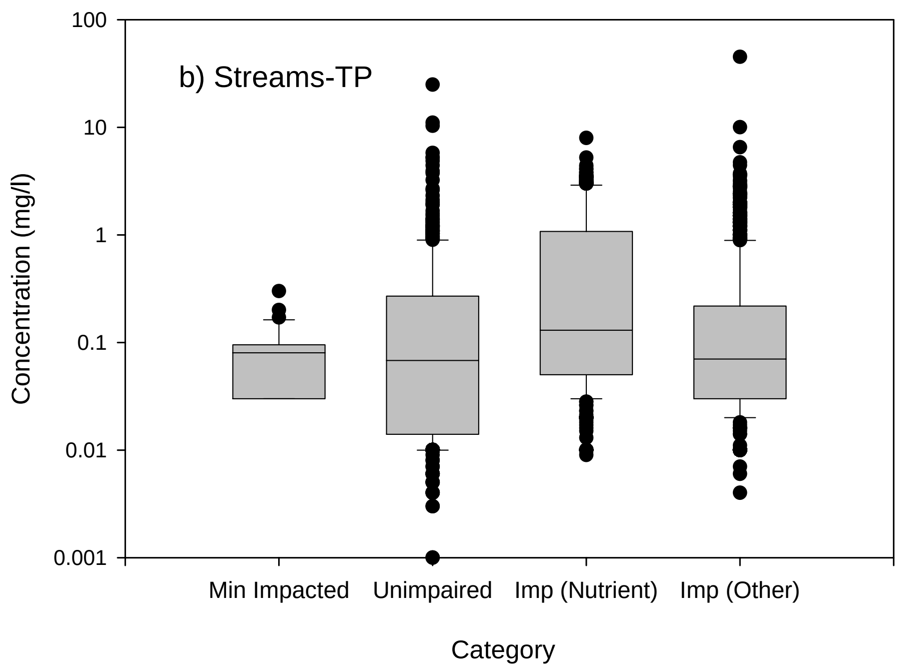
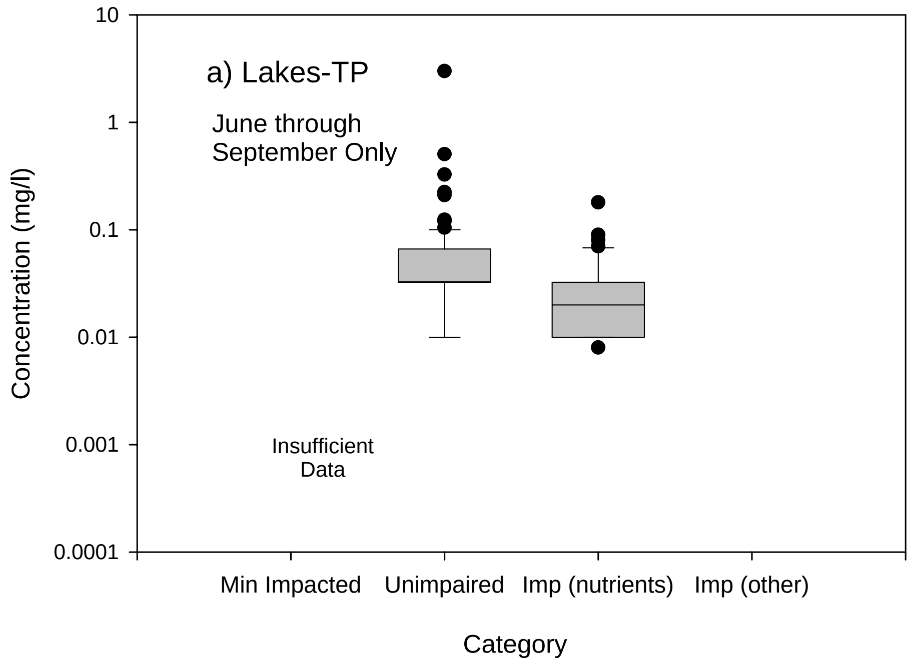


Figure 4-17 TP levels across the whole year in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.



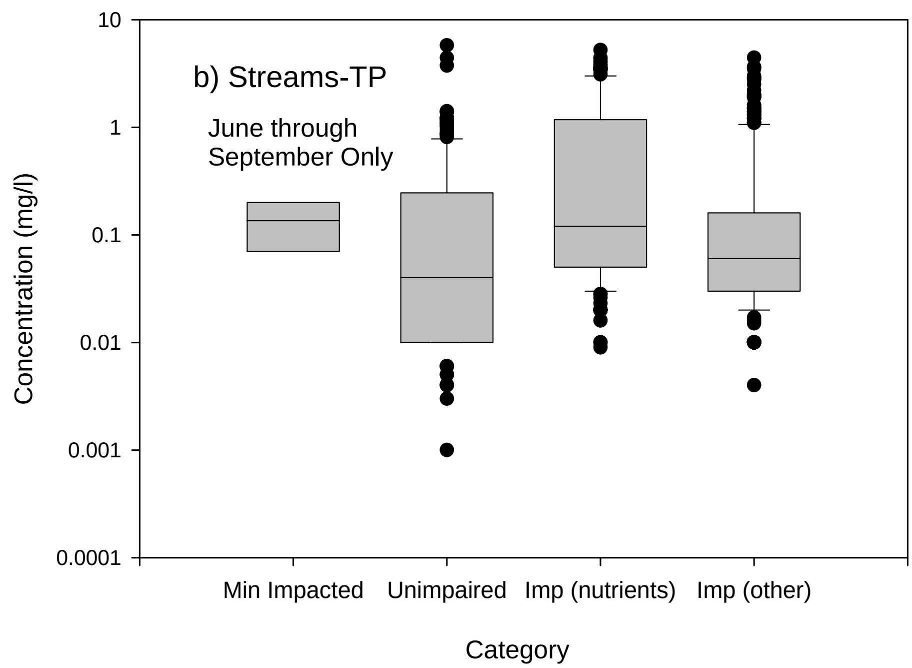


Figure 4-18 TP levels over June through September in lakes and streams, classified as minimally impacted, unimpaired, nutrient impaired, and non-nutrient impaired. The horizontal line in the middle of the each box represents the median, the lines are the 10th and 90th confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

Table 4-2
Nutrient Concentrations in Lakes (All Year)

Chemical	Stream Type	Median	Average	First Quartile	Second Quartile	Third Quartile	Fourth Quartile	No of Datapoints
NH_3	Unimpaired	0.08	0.12	0.08	0.08	0.08	1.15	228
	Impaired (other)	0.08	1.86	0.03	0.08	0.94	20.10	64
NO_2	Unimpaired	0.02	0.07	0.01	0.02	0.06	0.70	37
	Impaired (other)	0.01	0.07	0.01	0.01	0.02	0.70	62
NO_3	Unimpaired	0.10	0.43	0.10	0.10	1.00	4.52	190
	Impaired (other)	0.70	1.88	0.23	0.70	2.60	15.81	28
TKN	Unimpaired	0.50	0.73	0.20	0.50	1.00	5.40	315
	Impaired (other)	0.50	0.96	0.30	0.50	0.80	9.40	107
PO_4	Unimpaired	0.15	0.29	0.07	0.15	0.24	2.10	46
	Impaired (other)	0.01	0.10	0.01	0.01	0.03	1.86	55
TP	Unimpaired	0.03	0.08	0.03	0.03	0.08	3.00	252
	Impaired (other)	0.03	0.03	0.01	0.03	0.04	0.11	81

Table 4-3
Nutrient Concentrations in Lakes (June through September)

Chemical	Lake Type	Median	Average	First Quartile	Second Quartile	Third Quartile	Fourth Quartile	No of Datapoints
NH_3	Unimpaired	0.08	0.09	0.08	0.08	0.08	0.82	81
	Impaired (other)	0.06	0.83	0.02	0.06	0.13	7.60	29
NO_2	Unimpaired	0.06	0.13	0.01	0.06	0.12	0.70	9
	Impaired (other)	0.01	0.08	0.01	0.01	0.02	0.70	31
NO_3	Unimpaired	0.10	0.49	0.10	0.10	1.00	1.50	81
	Impaired (other)	1.36	1.64	0.18	1.36	2.78	4.00	8
TKN	Unimpaired	0.50	0.70	0.20	0.50	1.00	4.10	117
	Impaired (other)	0.40	0.59	0.30	0.40	0.60	4.30	42
PO_4	Unimpaired	0.45	0.70	0.19	0.45	1.28	2.10	9
	Impaired (other)	0.01	0.02	0.01	0.01	0.01	0.14	27
TP	Unimpaired	0.03	0.08	0.03	0.03	0.07	3.00	109
	Impaired (other)	0.02	0.03	0.01	0.02	0.03	0.18	37

Table 4-4
Nutrient Concentrations in Streams (All Year)

Chemical	Stream Type	Median	Average	First Quartile	Second Quartile	Third Quartile	Fourth Quartile	No of Datapoints
NH_3	Minimally Impacted	0.02	0.05	0.01	0.02	0.04	3.25	261
	Unimpaired	0.02	0.41	0.01	0.02	0.07	32.94	1229
	Impaired (nutrient)	0.05	0.34	0.01	0.05	0.14	12.10	907
	Impaired (other)	0.05	0.47	0.02	0.05	0.12	17.10	1279
NO_2	Minimally Impacted	0.00	0.01	0.00	0.00	0.00	0.06	110
	Unimpaired	0.02	0.15	0.01	0.02	0.13	12.00	1500
	Impaired (nutrient)	0.04	0.09	0.01	0.04	0.10	5.00	861
	Impaired (other)	0.02	0.14	0.01	0.02	0.09	2.95	1160
NO_3	Minimally Impacted	0.05	0.16	0.05	0.05	0.15	2.85	112
	Unimpaired	0.36	4.45	0.05	0.36	3.70	48.09	1301
	Impaired (nutrient)	4.74	5.02	1.17	4.74	7.50	31.84	600
	Impaired (other)	2.2	4.71	0.56	2.20	4.80	48.10	1037
TKN	Minimally Impacted	0.25	0.31	0.13	0.25	0.41	1.20	156
	Unimpaired	0.40	1.01	0.20	0.40	0.93	42.70	1425
	Impaired (nutrient)	0.7	1.06	0.40	0.70	1.20	11.00	868
	Impaired (other)	0.6	0.97	0.30	0.60	1.10	33.00	1486
PO_4	Minimally Impacted	0.04	0.05	0.02	0.04	0.07	0.23	260
	Unimpaired	0.08	0.49	0.02	0.08	0.50	28.73	1671
	Impaired (nutrient)	0.22	0.60	0.03	0.22	0.90	8.10	1056
	Impaired (other)	0.05	0.45	0.02	0.05	0.26	40.00	1793
TP	Minimally Impacted	0.08	0.08	0.03	0.08	0.09	0.30	34
	Unimpaired	0.07	0.36	0.01	0.07	0.27	24.80	633
	Impaired (nutrient)	0.13	0.77	0.05	0.13	1.07	7.94	525
	Impaired (other)	0.07	0.34	0.03	0.07	0.22	45.10	1069

Table 4-5
Nutrient Concentrations in Streams (June through September)

Chemical	Stream Type	Median	Average	First Quartile	Second Quartile	Third Quartile	Fourth Quartile	No of Datapoints
NH_3	Minimally Impacted	0.01	0.01	0.00	0.01	0.02	0.12	88
	Unimpaired	0.02	0.38	0.01	0.02	0.07	14.40	331
	Impaired (nutrient)	0.04	0.23	0.01	0.04	0.11	9.30	313
	Impaired (other)	0.04	0.36	0.01	0.04	0.09	16.30	459
NO_2	Minimally Impacted	0.00	0.01	0.00	0.00	0.01	0.02	11
	Unimpaired	0.05	0.26	0.01	0.05	0.17	12.00	390
	Impaired (nutrient)	0.04	0.11	0.01	0.04	0.10	5.00	288
	Impaired (other)	0.02	0.15	0.01	0.02	0.09	2.50	415
NO_3	Minimally Impacted	0.08	0.13	0.03	0.08	0.10	0.84	11
	Unimpaired	0.30	5.18	0.05	0.30	4.59	45.16	326
	Impaired (nutrient)	5.43	5.69	2.90	5.43	7.75	28.99	185
	Impaired (other)	2.20	5.05	0.58	2.20	4.53	42.45	312
TKN	Minimally Impacted	0.13	0.24	0.08	0.13	0.31	0.99	61
	Unimpaired	0.40	0.85	0.20	0.40	0.90	13.00	393
	Impaired (nutrient)	0.70	1.01	0.40	0.70	1.10	11.00	309
	Impaired (other)	0.52	0.81	0.30	0.52	1.00	8.60	537
PO_4	Minimally Impacted	0.02	0.04	0.01	0.02	0.04	0.18	88
	Unimpaired	0.08	0.52	0.02	0.08	0.56	23.60	433
	Impaired (nutrient)	0.25	0.66	0.02	0.25	0.90	8.98	354
	Impaired (other)	0.03	0.42	0.01	0.03	0.24	12.55	624
TP	Minimally Impacted	0.14	0.14	0.10	0.14	0.17	0.20	2
	Unimpaired	0.04	0.25	0.01	0.04	0.24	5.77	220
	Impaired (nutrient)	0.12	0.82	0.05	0.12	1.17	5.20	197
	Impaired (other)	0.06	0.29	0.03	0.06	0.16	4.42	420

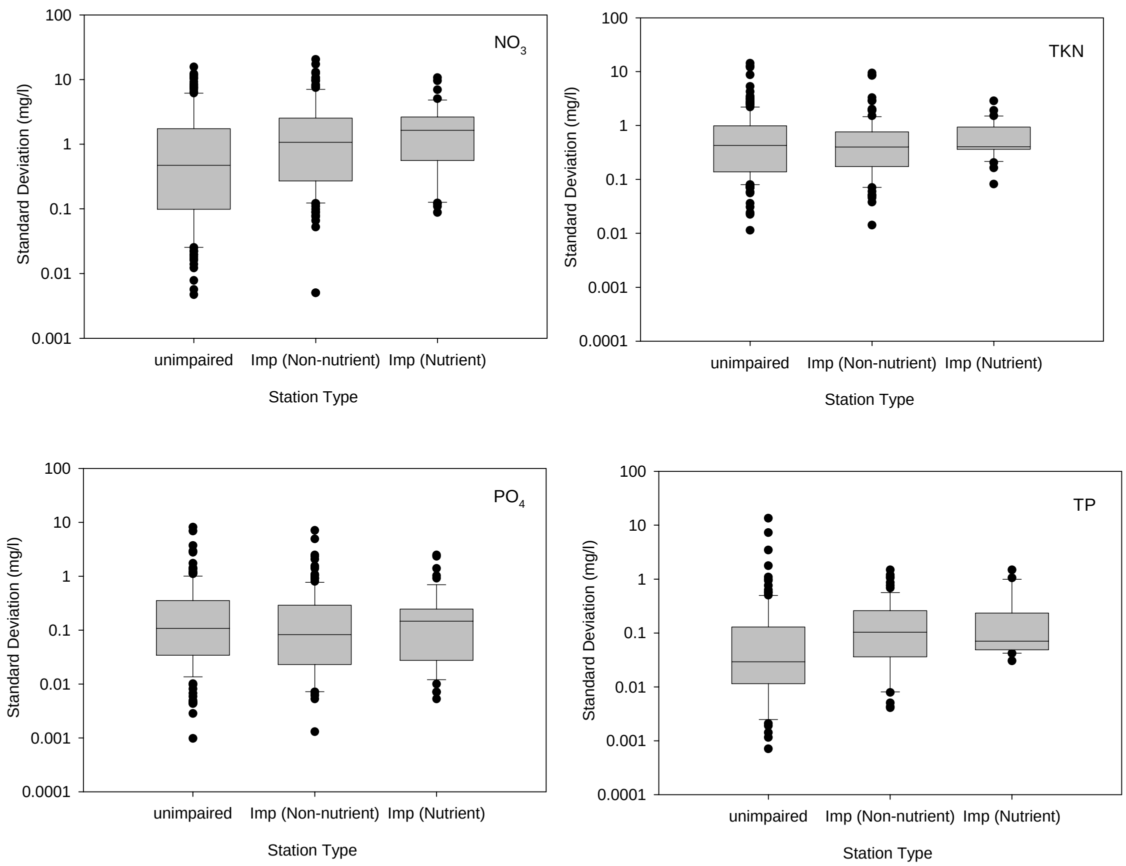


Figure 4-19 Standard deviations of NO_3, TKN, PO_4 and TP across different types of streams in Ecoregion 6. The horizontal line in the middle of the each box represents the median, the lines are the 10^{th} and 90^{th} confidence intervals and the black circles are the outliers. Note the log-scale on the y-axis of the plots.

- The seasonal effect of nutrients, particularly during the growing season was also considered in these analyses. Data were plotted separately for the months of June through September during which temperatures are expected to be warm, and algal growth likely to be significant. The results of these analyses are presented alternately with the whole-year analysis in Figures 4-8, 4-10, 4-12, 4-14, 4-16, and 4-18. These results amplify the findings of the whole year plots, especially for NO_3, where the difference between the minimally impacted and impaired stations is greater. For the other parameters the results are supportive of the whole year analysis. These results demonstrate that it may be possible to focus the criteria on nutrient concentrations during the warm, growing months of the year.

- Data summaries for all waterbody types, presented in Tables 4-2 through 4-5, can be used to supplement the box plots to identify the median and upper and lower quartile of constituent concentrations for impaired and unimpaired water bodies. Thus, over June through September, median concentrations of nitrates in streams vary from 0.08 mg/l for minimally impacted water bodies, to 0.3 mg/l for water bodies that are unimpaired and meet their beneficial uses, and increase to 5.43 mg/l in nutrient-impaired water bodies. Likewise, PO_4 concentrations increase from 0.02 mg/l in minimally impacted water bodies to 0.08 mg/l in unimpaired water bodies, increasing to 0.25 mg/l in nutrient impaired water bodies. In contrast, median TP levels in minimally impaired streams are almost as high as in nutrient-impaired streams (0.14 mg/l vs. 0.12 mg/l).

- The N-P ratio provides one basis for suggesting that nitrogen species may be more strongly correlated to impairment. When the molar ratio of nitrogen: phosphorus is greater than 16, the expected ratio of these elements in algal biomass, a water body is thought to be phosphorus limited, and when this ratio is less than 16, a water body may become nitrogen-limited. Co-located nitrogen and total phosphorus values (same station, date, and time) were plotted in Figure 4-20 to determine which element is most likely to be limiting. Most of the stream stations in Ecoregion 6 appear to be nitrogen limited as indicated by the Redfield ratio, although the lake stations appear to be limited by both nutrients. This finding may explain why we see a strong relationship between impairment and nitrate levels in streams in Ecoregion 6.

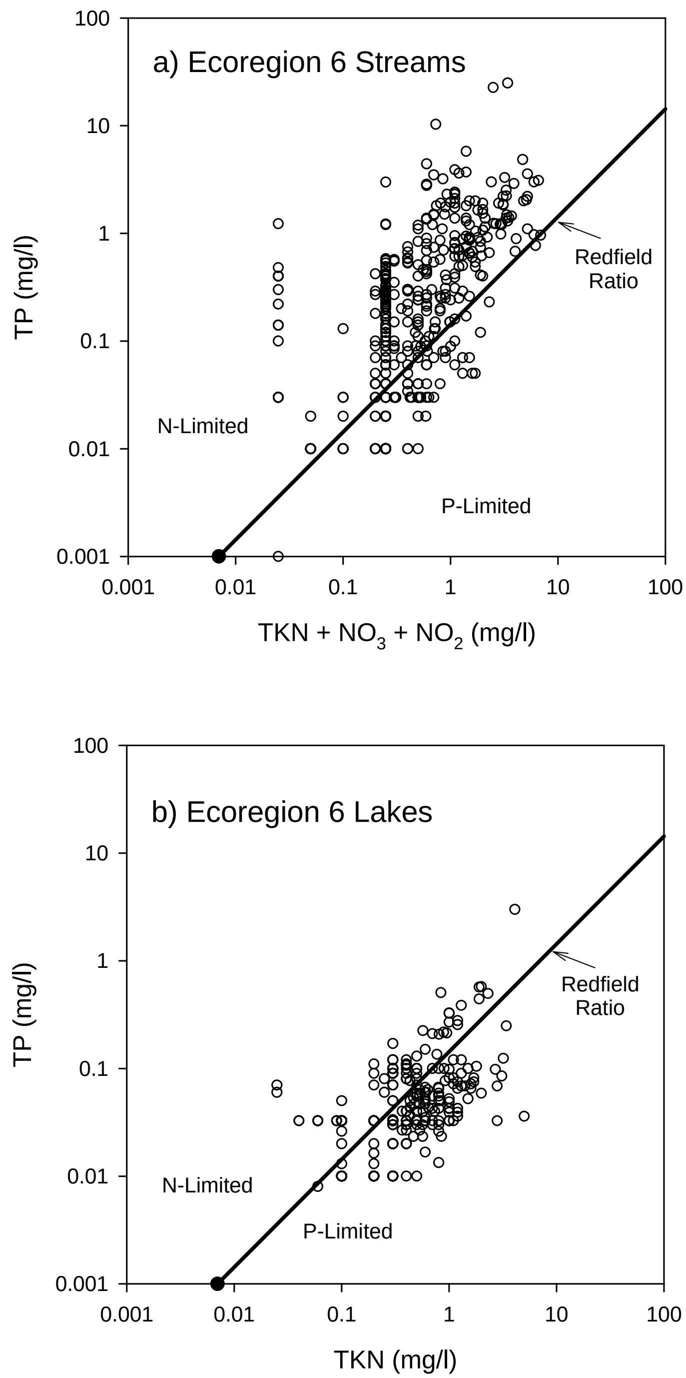


Figure 4-20 Co-located (same station, date, time) measurements of total nitrogen (sum of TKN, NO_3, and NO_2) and total phosphorus in Ecoregion 6 streams and lakes.

4.6 NUTRIENT LAND-USE RELATIONSHIPS

It is well understood that the presence of developed land in a watershed can lead to increased nutrient levels in downstream water bodies, as a result of various anthropogenic point and non-point sources. To understand nutrient levels in the absence of anthropogenic inputs, we are also interested in the distribution of values for stations where the percentage of developed land is low. To evaluate the effects of land use on nutrient concentrations, we performed a preliminary analysis where we looked at the proportion of different land uses with the CALWATER watersheds in which each of the study stations fell. The relationship between percentage of developed land (either percent of agricultural land or percent of urban land) and nutrient concentrations are shown in Figure 4-21 for streams and in Figure 4-22 for lakes. Although the relationships are noisy in all cases, more can be inferred from the stream plots because of the larger number of data points. In most instances it can be seen that for higher levels of developed land, nutrient concentrations are elevated. Interestingly, however, when the percentage of developed land is low, nutrient concentrations can be both high and low. The general relation is strongest for NO_3 data. This is indicative of a) possible inaccuracy in the analysis, because the land use in the CALWATER watershed for a station may not represent the land use in its entire watershed, or b) the effect of background sources of nutrients. In future work with these data, this question will be considered in greater detail by using the calculated watershed for each station in Ecoregion 6. For the urban land use, it appears that there is a decrease in some nutrient concentrations at high percentages of urban land. This is an interesting finding although possibly not important from the viewpoint of nutrient criteria development.

4.7 STREAM LEVEL AND NUTRIENT CONCENTRATION

Streams in EPA's RF1 database are characterized by level from 1 through 8; streams at the highest level (Level 8) are small streams with no tributaries, which feed into lower level streams. At the other extreme, Level 1 streams are expected to be large streams/rivers than drain into oceans. As we move from Level 8 to Level 1 streams we expect increases in catchment area and flow. The progression to larger streams is expected to reduce nutrient concentrations because of removal processes in streams. The relationship between nutrient chemistry and stream levels for unimpaired streams is shown in Figure 4-23. Based on the existing dataset there do not appear to be strong relationships between stream level and concentrations of NO_3, TKN, and TP. The pool of data for the minimally impacted stations was not large enough to perform a robust analysis, although such an analysis is recommended for future work.

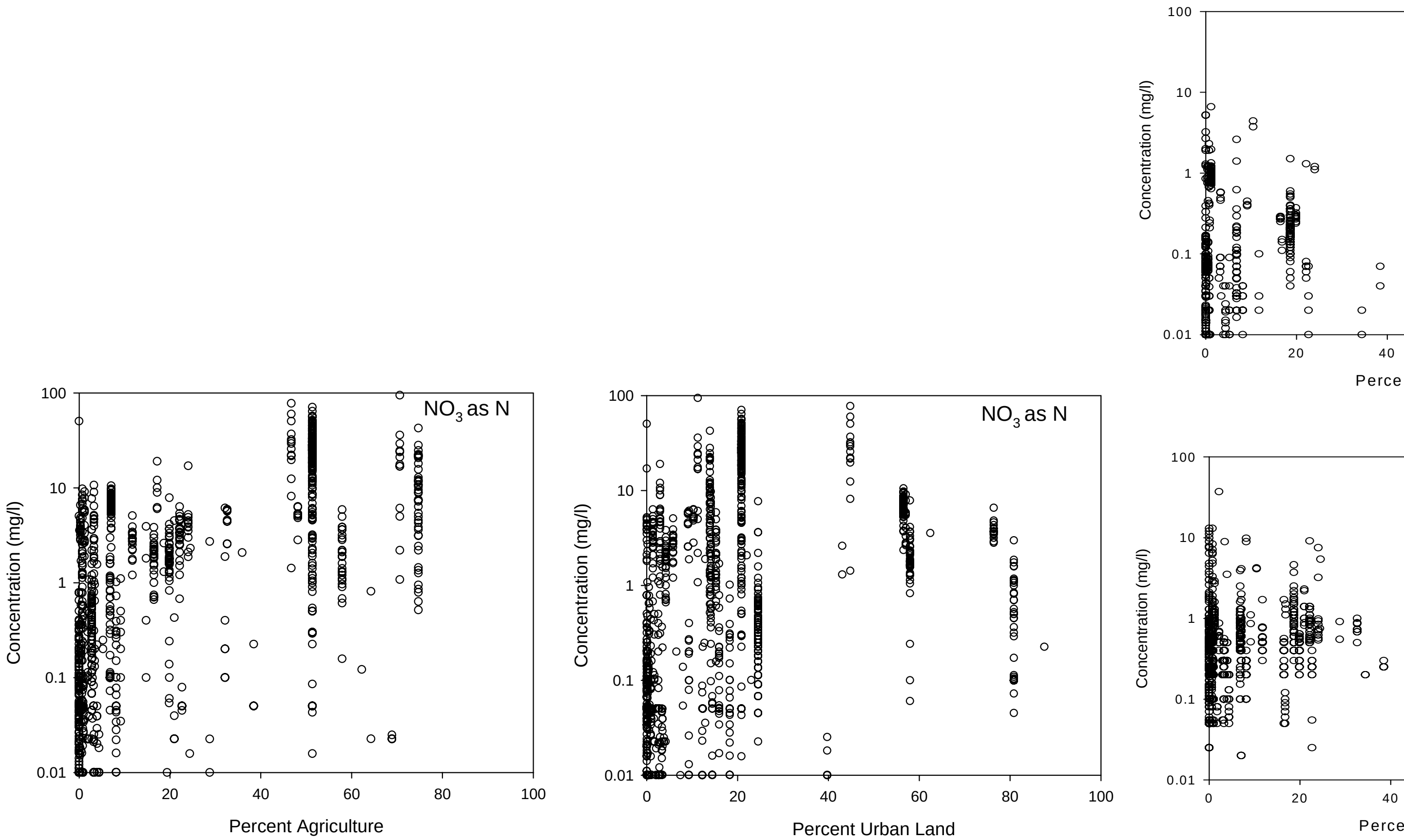
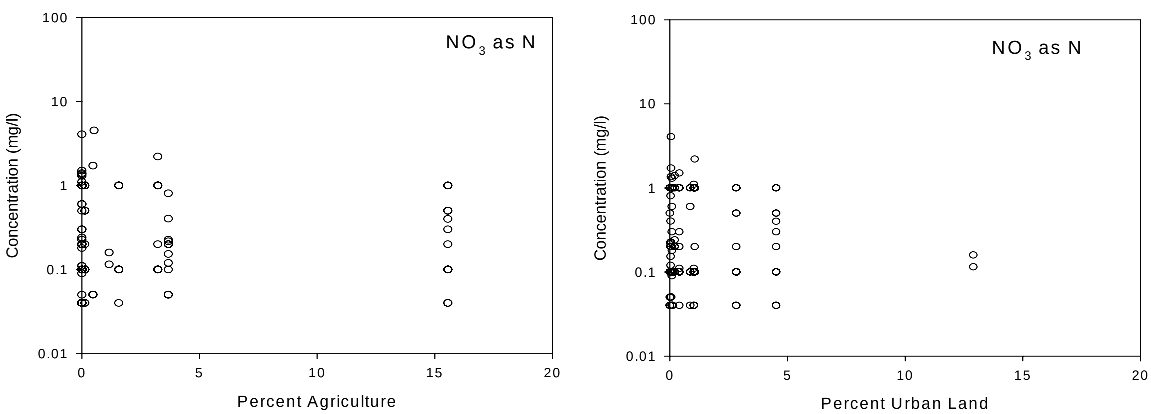


Figure 4-21 NO3, TP, and TKN measurements related to land use in the CALWATER watershed of th



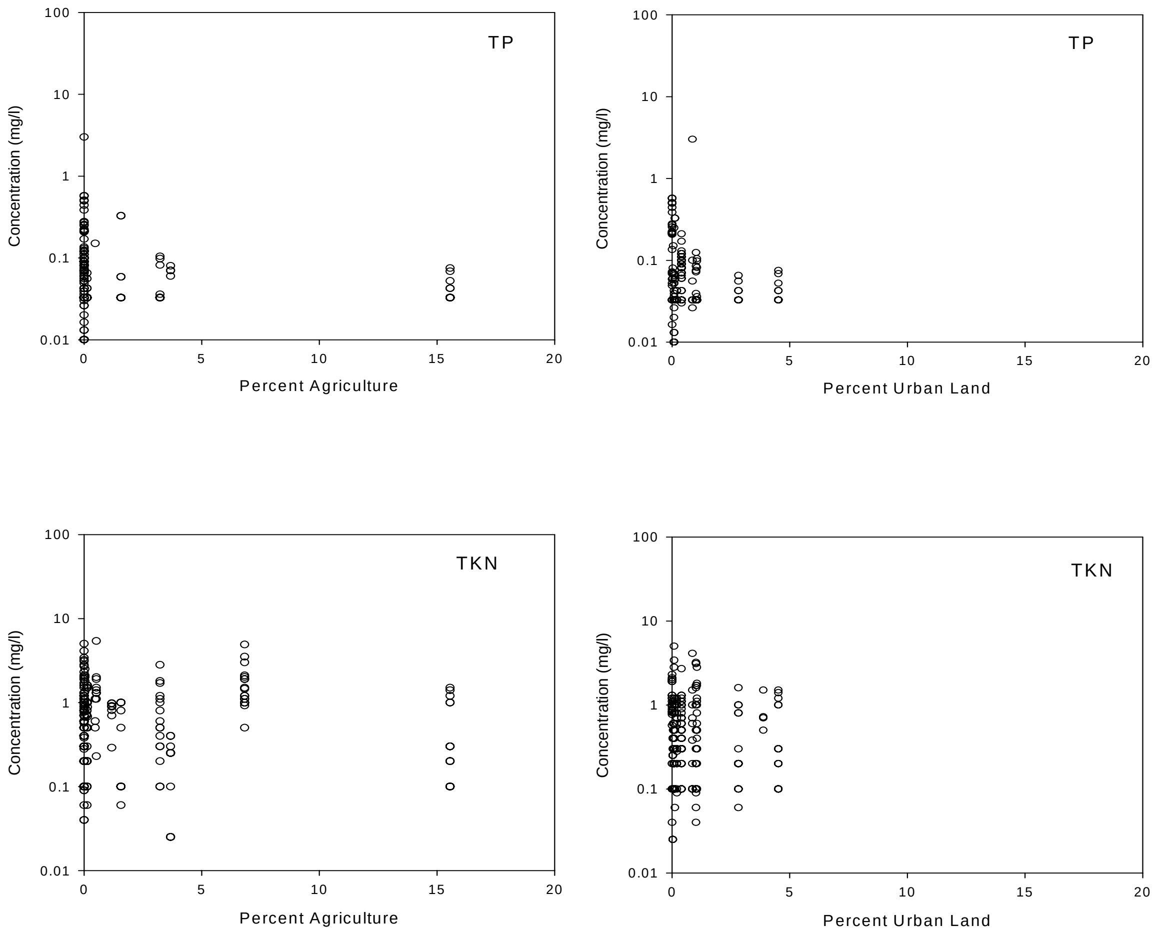


Figure 4-22 NO_3, TP, and TKN measurements related to land use in the CALWATER watershed of the corresponding station for lake stations.

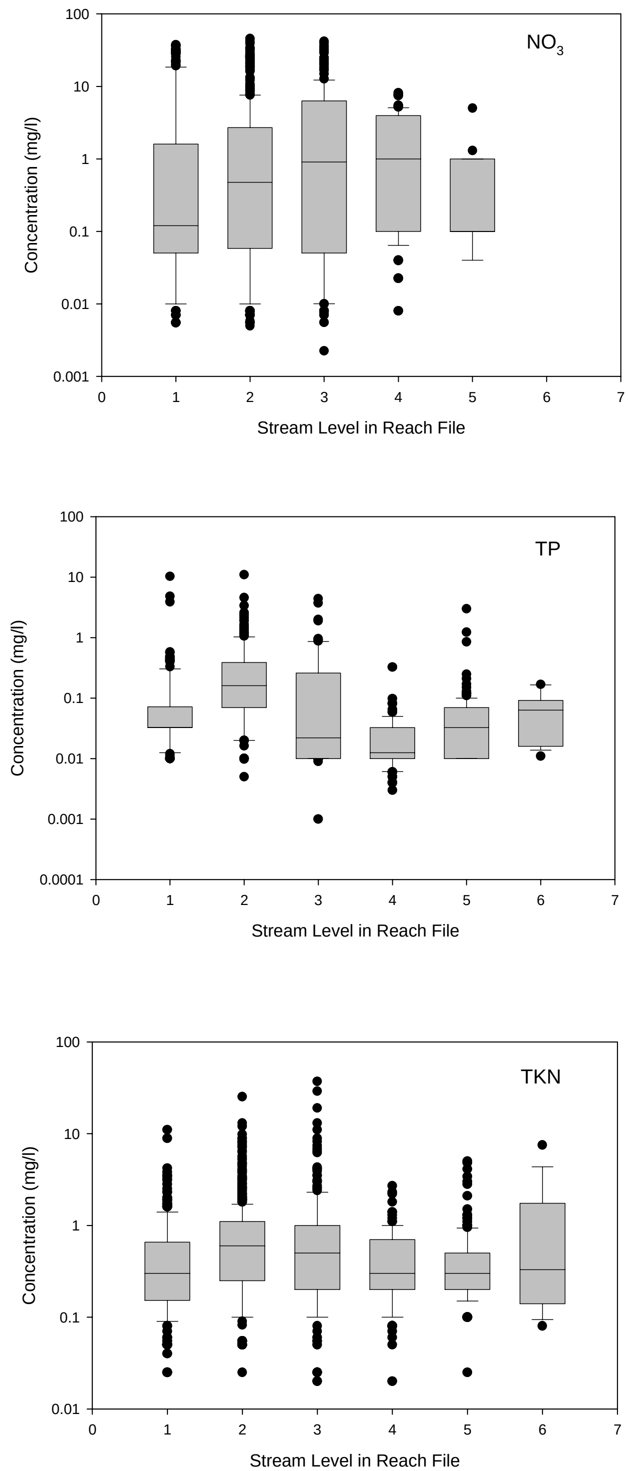


Figure 4-23 NO_3, TP, and TKN measurements in unimpaired streams of Ecoregion 6 related to stream level in the Reach File 1 Database.

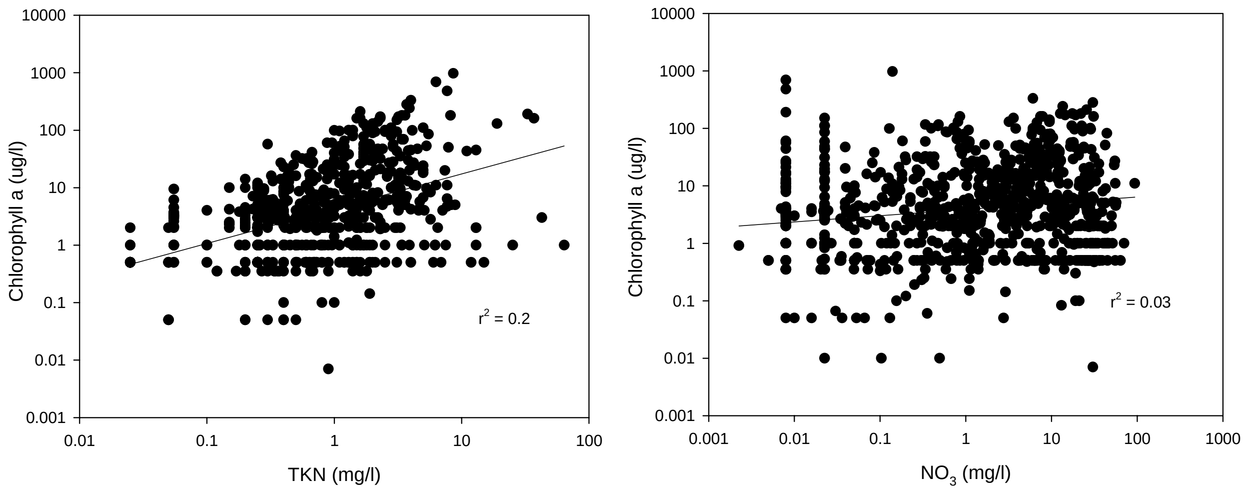
4.8 CHLOROPHYLL-*A* AND NUTRIENTS

In general, chlorophyll-*a* values were relatively sparse in the data collected for Ecoregion 6 and insufficient for a region-wide analysis. An exception, however, is the dataset obtained from Regional Board 3, which does contain a large number of co-located measurements of nutrient chemistry and chlorophyll-*a* concentrations. These data were used to study the nature of the relationship between chlorophyll-*a* and nutrients as shown in Figures 4-24 and 4-25. The data show a correlation between TKN and chlorophyll-*a*, and somewhat weaker correlations for NO_3 and PO_4. TP data were insufficient to draw any conclusions. This association of chlorophyll-a with TKN is in line with our finding earlier that most streams in Ecoregion 6 are nitrogen limited. Where sufficient data are available, chlorophyll target concentrations can be used to determine a corresponding range of nutrient concentrations that can be used to guide criteria development. At present these data are limited to a small part of the Ecoregion and cannot be extrapolated to the entire area. Future data collection of this nature over more stations is strongly recommended.

4.9 ASSESSMENT OF SUBSETS OF DATA

To evaluate the spatial relationships of nutrient constituents over Ecoregion 6, medians of key parameters were plotted on a map of the region. Plots of NO_3, TKN, PO_4, and TP are shown in Figures 4-26 through 4-29. These maps permit a different assessment of the same data that have been discussed in earlier sections. By far the largest number of stations with usable parameters are in the coastal regions of Ecoregion 6. In particular, it appears that the Central Coast Region south of Monterey Bay has low concentrations of all four constituents that have been mapped, whereas the areas further south such as those near San Luis Obispo, Santa Barbara, and Los Angeles, all high consistently higher concentrations of all four nutrients. The Monterey Bay area has high concentrations of nutrients, especially NO_3 and TKN. The area south of San Francisco Bay has high concentrations of TKN, but relatively low concentrations of the phosphorus species. The coastal areas north of San Francisco Bay have low concentrations of TKN, TP, and PO_4. Despite a large number of stations overall, it seems we still have insufficient data to characterize nutrient concentrations in the northern part of Ecoregion 6. This will be a focus of future data collection.

Data for All Year



Data for May Through September

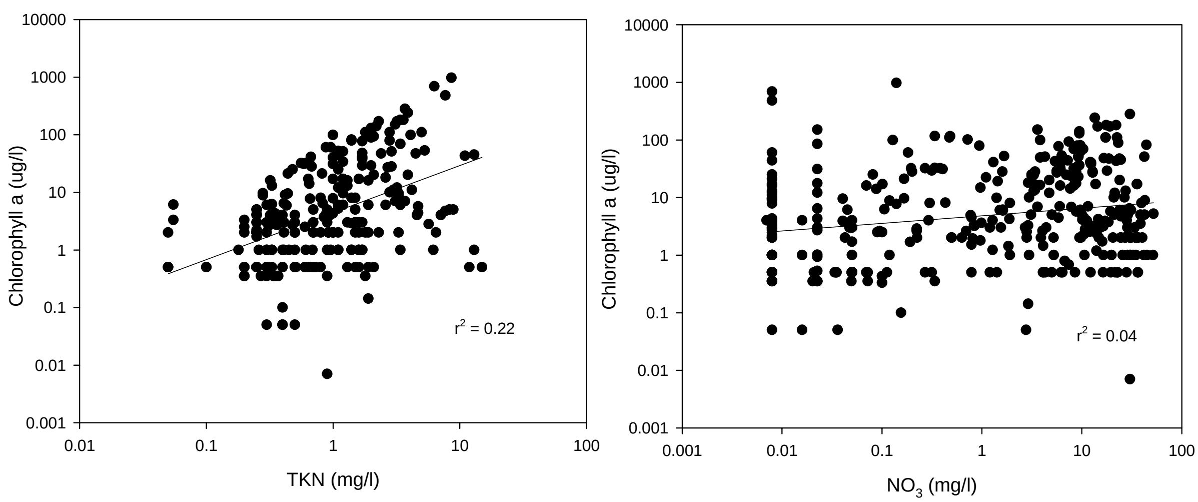


Figure 4-24 NO_3 and TKN measurements in RWQCB 3 streams related to chlorophyll *a*. Data are shown for (a) the whole year and (b) for May through September.

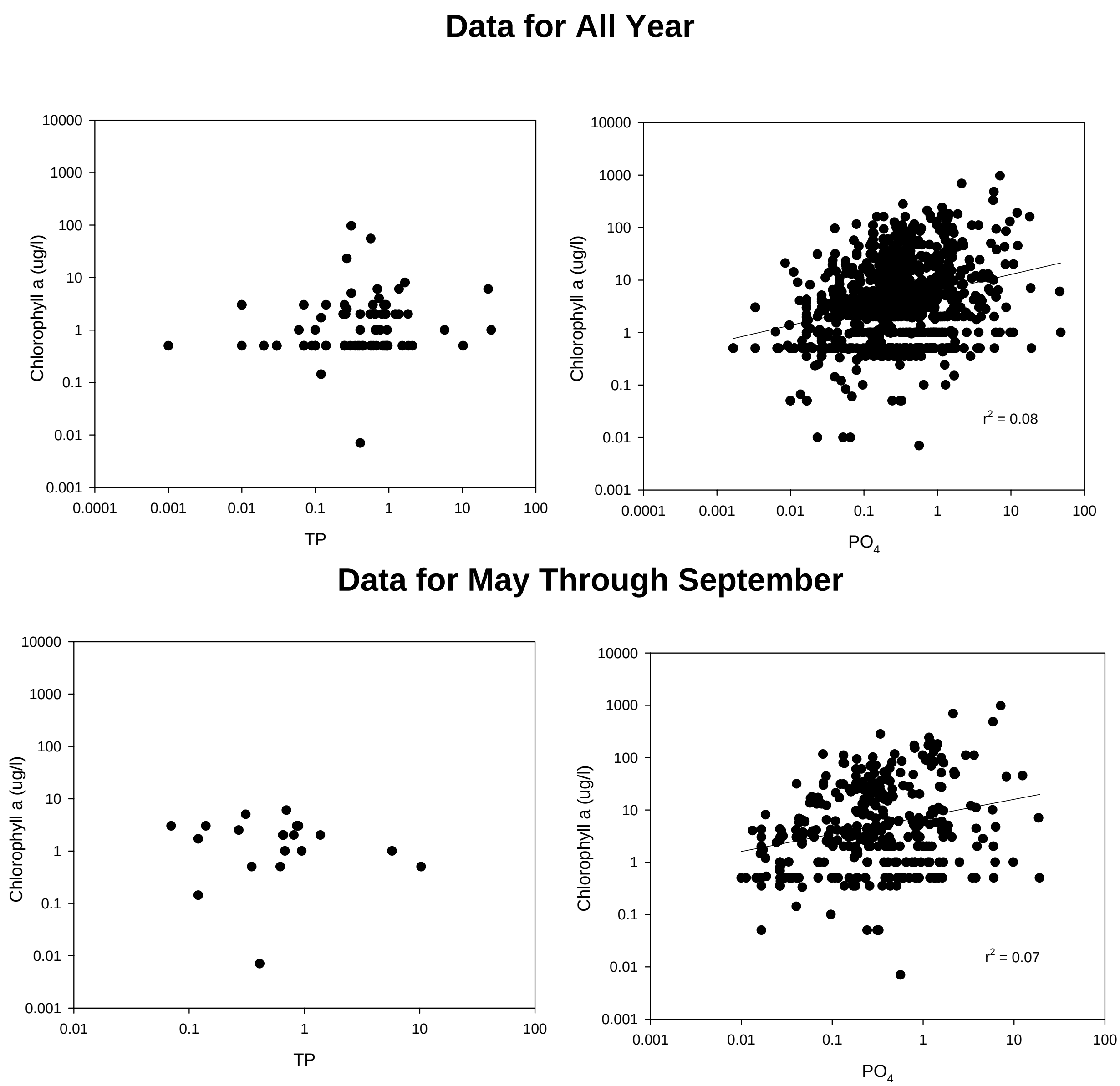


Figure 4-25 PO_4 and TP measurements in RWQCB 3 streams related to chlorophyll *a*. Data are shown for (a) the whole year and (b) for May through September.

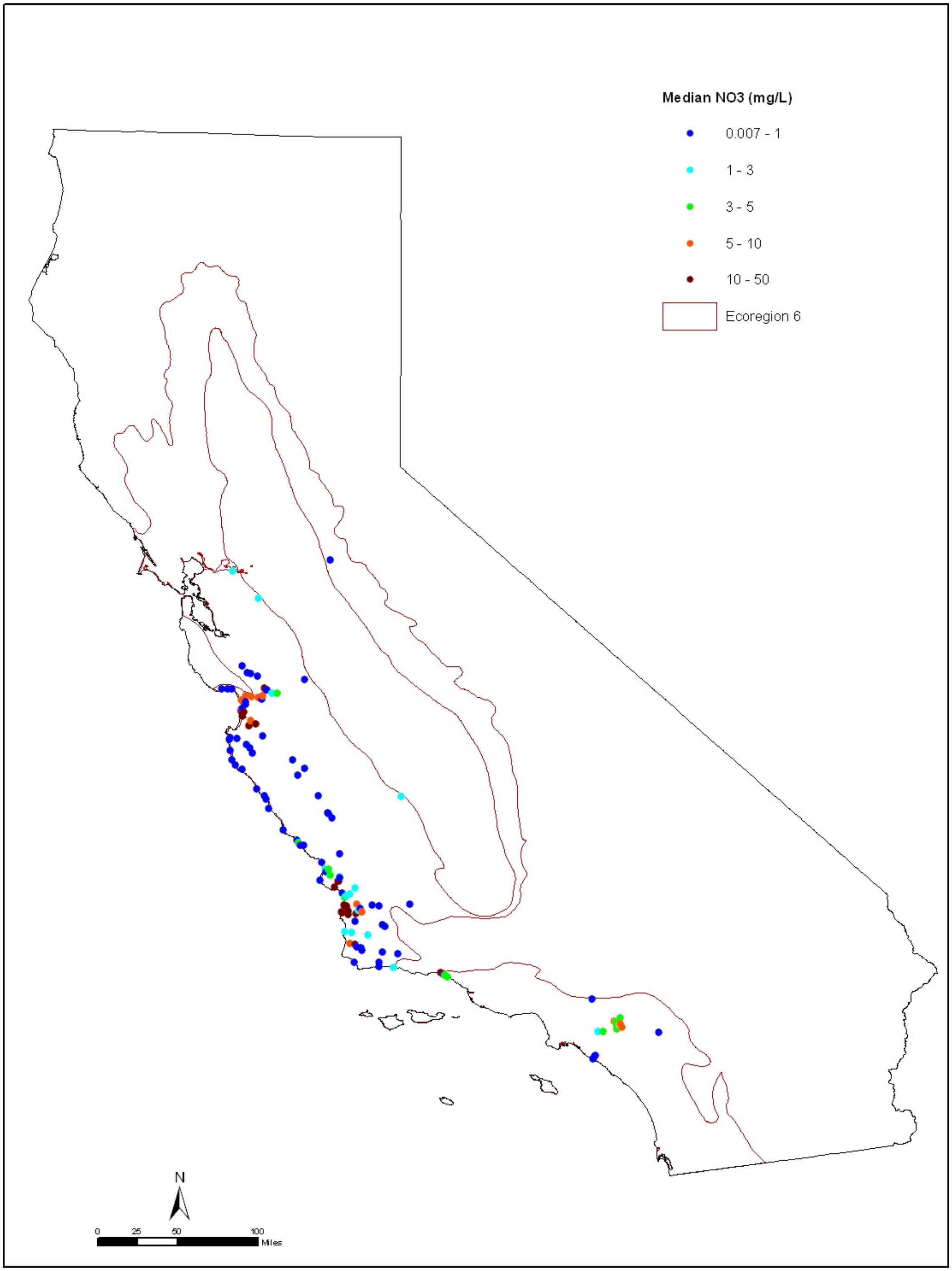


Figure 4-26 Median concentrations of NO_3 across Ecoregion 6.

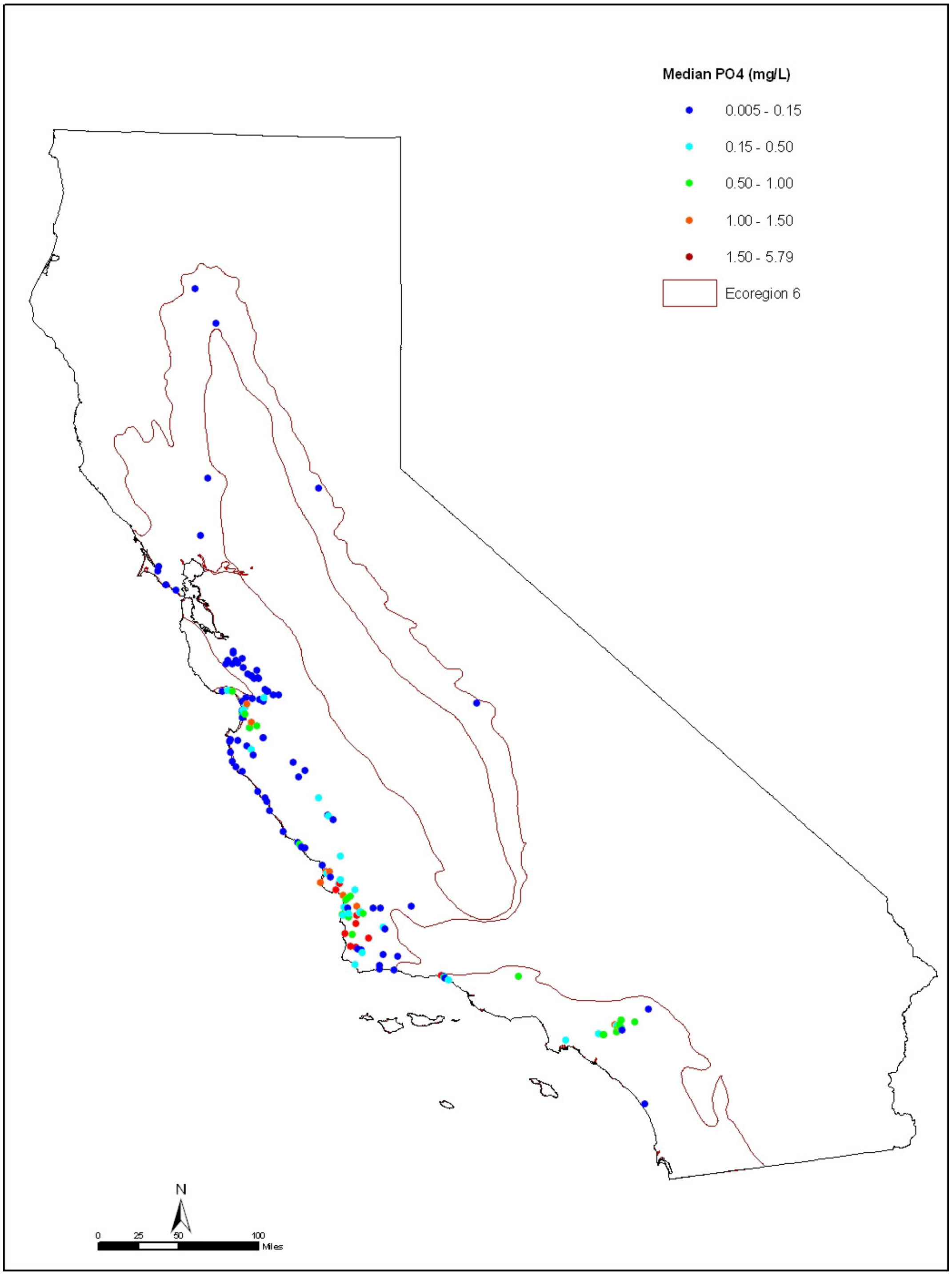


Figure 4-27 Median concentrations of PO_4 across Ecoregion 6.

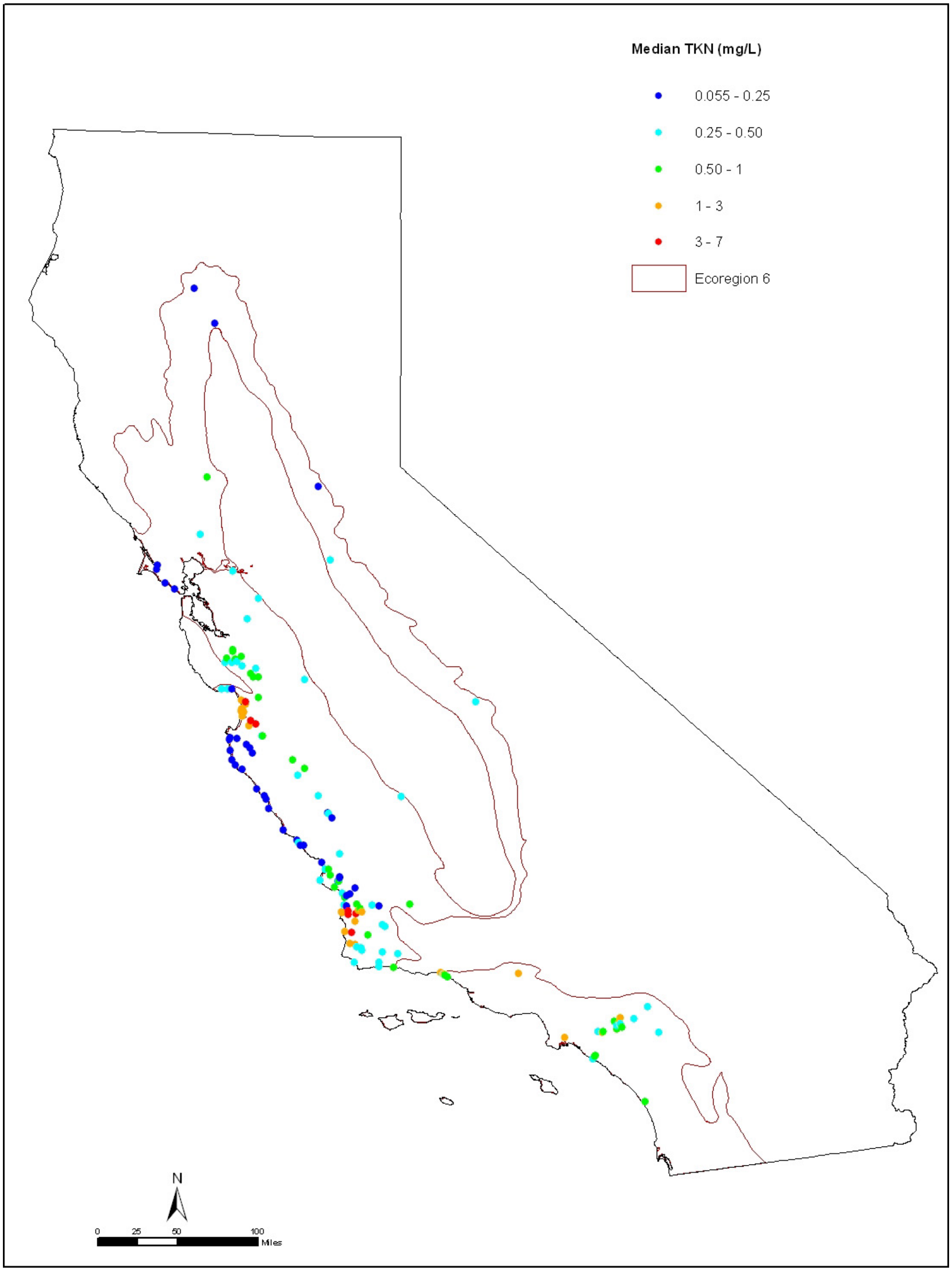


Figure 4-28 Median concentrations of TKN across Ecoregion 6.

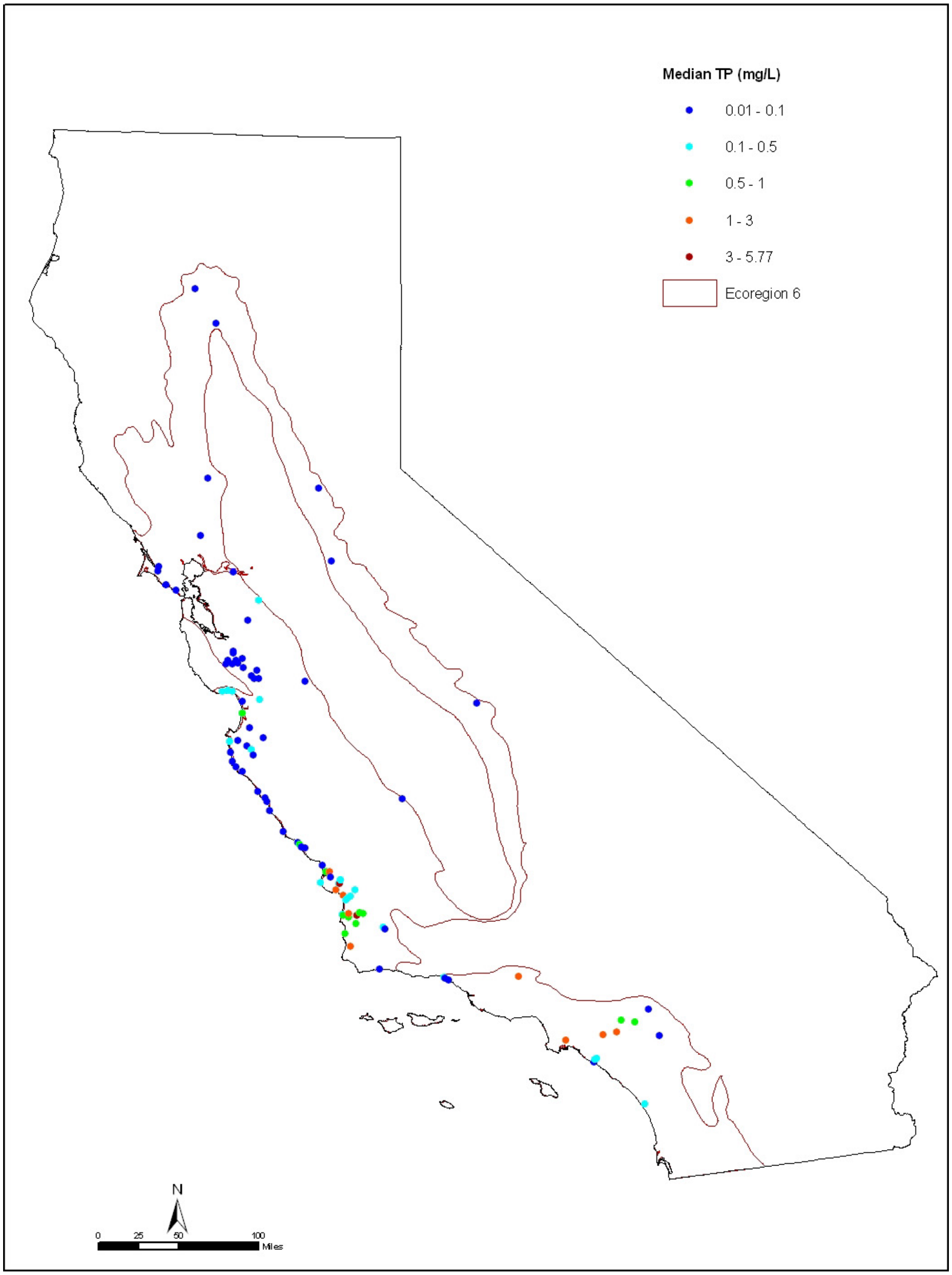


Figure 4-29 Median concentrations of TP across Ecoregion 6.

4.10 THE DODDS EMPIRICAL APPROACH TO STREAM CRITERIA

Dodds *et al*. (1997) developed nutrient criteria to address nuisance growth of benthic algae in the Clark Fork River (Montana), which have been widely cited. The criteria were developed based on empirical regression relationships between benthic chlorophyll *a* and nutrient concentrations. While some site-specific data from the Clark Fork are included, it is important to realize that the analysis is based primarily on a compilation of data from 205 sites throughout North America and New Zealand. In addition, the regressions rely on seasonal mean data, not point-in-time observations.

The best predictive regressions identified by Dodds *et al.* (1997) were nonlinear log-log regressions, in which the log (base 10) of mean benthic chlorophyll *a* and maximum benthic chlorophyll *a* were predicted from log(TN), the square of log(TN), and log(TP). TN and TP were found to be better regressors than inorganic N and inorganic P.

The relationships that were identified were relatively weak, with a maximum adjusted R^2 value of 0.430 in log space (Figure 4-30).

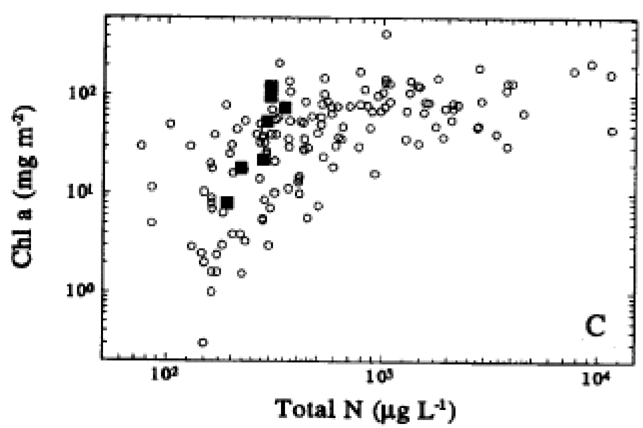


Figure 4-30 Relationship between Benthic Chlorophyll a and Total N shown in Dodds *et al.* (1997; Figure 2C)

Biggs (2000) found a similar degree of fit for New Zealand data, with a best reported R^2 of 0.325 for log-log regressions based on nutrient concentrations only. However, he was able to increase the R^2 value to 0.741 by including days of accrual in the relationship. It is likely that incorporating days of accrual into the Dodds *et al*. (1997) dataset might result in a similar improvement in predictive ability.

The two regression relationships recommended by Dodds *et al*. (1997) are:

$$\log(\text{mean Chl } a) = -3.22360 + 2.82630 \log(TN) - 0.431247 (\log(TN))^2 + 0.25464 \log(TP),\ R^2=0.430$$

and

$$\log(\text{max Chl } a) = -2.70217 + 2.78572 \log(TN) - 0.43340 (\log(TN))^2 + 0.30568 \log(TP),\ R^2=0.354.$$

The nutrient criteria recommendations given by Dodds *et al*. (1997) were created by fixing the N:P ratio at the Redfield ratio and solving the regression equation for appropriate concentrations of TN and TP to meet a benthic algae density target. This yields a central estimate in which approximately half of the observed sites would be expected to have an algal density greater than the target and the prescribed nutrient concentration value. Because the regression relationship is relatively weak and in log space, a high level of uncertainty is associated with the estimated nutrient target values. For instance, to obtain a target maximum chlorophyll *a* concentration of 100 mg/m^2, Dodds *et al*.'s regression analysis yields a target nitrogen concentration of 275 µg/L – but predictions associated with this TN concentration have a 95 percent confidence interval on maximum chlorophyll *a* of 7.8 to 407 mg/m^2. Dodds *et al*. therefore buttressed their arguments with a weight of evidence approach, noting (1) that the observed data indicate that when mean TN concentrations remained at or below 500 µg/L, mean benthic chlorophyll *a* densities exceeded 150 mg/m^2 in only 5 percent of cases, and (2) concentrations in an unimpaired reference station were similar.

Dodds *et al*.'s regression analysis also yields a TP target of 35 µg/L. Final recommendations were adjusted to 350 µg/L TN and 30 µg/L TP.

In addition to hydraulic effects, the observations used by Dodds *et al*. (1997) will be affected by light availability. No data on percent available light or canopy closure are provided with the data set. However, because they relied on well-studied periphyton sites it seems likely that the data set is biased toward streams in which light sufficient to promote ample periphyton growth is present.

Dodds *et al*. (1998) extended the analysis of the same data set used in the earlier work. In this second paper the authors appear to have abandoned the regression approach. Instead, they proposed trophic classification boundaries based on a simple division of the cumulative frequencies in the observed data into thirds, yielding the values summarized in Table 4-6.

Table 4-6
Boundaries for Stream Trophic Classifications Proposed by Dodds *et al*. (1998)

Variable	Oligotrophic-Mesotrophic Boundary	Mesotrophic-Eutrophic Boundary
Mean benthic chlorophyll *a* (mg/m^2)	20	70
Maximum benthic chlorophyll *a* (mg/m^2)	60	200
TN (µg/L)	700	1500
TP (µg/L)	25	75

The values shown in Table 4-6 are less than satisfactory for use as nutrient criteria for two reasons. First, they represent a naïve statistical tabulation into equal thirds that are not in any way tied to actual impairment. Second, they do not account for regional differences in nutrient background levels, light, or temperature. A somewhat better (but still very naïve) use of these data might be made as follows: Rather than pick breakpoints at arbitrary thirds of the distribution, determine the percentile of a desired chlorophyll target, then associate the corresponding percentile of the N and P distributions. For instance, a mean benthic chlorophyll *a* of 100 mg/m^2 appears to occur at about the 87^{th} percentile of the frequency distribution. From the graphs in Dodds *et al*. (1998), this looks to correspond to TP of about 250 µg/L and TN of about 2500 µg/L.

Dodds *et al*. (2002) further expanded the literature data set used in their earlier analyses and also examined the USGS National Stream Water Quality Monitoring Network stream data. Correlation analysis confirmed a positive relationship between mean and maximum benthic chlorophyll *a* and TN and TP concentrations. The authors also examined correlations to stream gradient, water temperature, and latitude – but not shading. They report a negative correlation of benthic chlorophyll *a* to gradient, consistent with Biggs (2000) work on scour/accrual effects. However, stream gradient was available for only a small subset of the data, and thus could not be included in the regressions.

Dodds *et al*. (2002, Table 5) provided new linear log-log regression models for mean and maximum benthic chlorophyll *a* in the augmented literature data set. These vary significantly from those reported in Dodds *et al*. (1997); however, it is not clear if the 2002 work evaluated (but rejected as not significant) the non-linear term (the square of log(TN)) as a potential variable or simply omitted it because of theoretical objections to the resulting hyperbolic form, which predicts declining algal concentrations at high TN concentrations. The new best-fit regressions for the literature data set are:

$$\log(\text{mean Chl } a) = 0.155 + 0.236 \log(TN) + 0.443 \log(TP),\ R^2 = 0.40$$

and

$$\log(\text{max Chl } a) = 0.714 + 0.372 \log(TN) + 0.223 \log(TP),\ R^2 = 0.31.$$

For both models, the reported R^2 values are slightly less than the best-fit models in Dodds *et al*. (1997).

Dodds *et al*. (2002) also developed regression models, for mean chlorophyll *a* only, in the USGS stream data set. The fit of these models was, however, uniformly poor, with the best reported R^2 equal to 0.18. It appears that this data set had fewer samples per site than the literature data set, and thus more uncertainty in the evaluation of means, which may account for the poorer performance. In addition, the USGS data set may include more sites where stream shading is a significant uncontrolled covariate relative to the literature data set, as noted above. This fits with Dodds *et al*.'s comment that no observations in this dataset exceeded 100 mg/m^2. If this interpretation is correct, the regression against the literature data set should provide an approximate upper bound on the USGS data.

Although not done by Dodds *et al*. (2002), the proposed new regression models can be analyzed for criteria recommendations in a manner analogous to that in Dodds *et al*. (1997), using the N:P Redfield ratio of 7.23. Obtaining 100 mg/m^2 maximum benthic chlorophyll *a* with the new equations corresponds to TN of 304 mg/L and TP of 42 mg/L, both slightly higher than the amounts (275 and 35) estimated with the earlier regression.

4.11 APPLICATION TO CALIFORNIA DATA

No data sets within California have been identified on which an approach similar to that employed by Dodds *et al*. could be developed. There are a few data sets that do provide measures of benthic biomass, but these generally have only a few measurements per site and thus cannot be used to estimate maximum chlorophyll *a* or even to obtain good estimates of mean chlorophyll *a*. In addition, most of these sites do not have long runs of nutrient data, so nutrient concentrations must also be estimated from a few data points.

There is thus not a reasonable prospect of recreating a Dodds-type analysis, which is based on data from a wide range of sites throughout temperate zones of the world, with California data only. What can be done, however, is to compare these datasets to the Dodds *et al*. results and check for approximate consistency. In making this comparison, we expect to find the following results if the relationships are valid:

- Dodds' equation for mean chlorophyll *a* should approximate the center of the distribution of observed data (in log space) *for those sites at which strong light limitation or frequent scour is not a major confounding factor.*

- Based on the comparison of the literature data set and USGS data in Dodds *et al*. (2002), the equation for mean chlorophyll *a* should generally lie above the center of the data for a mixture of sites where light and scour limitations are important.

- Dodds' equation for maximum chlorophyll *a* should approximate the upper bound envelope on the observed data.

Two relatively large data sets have been identified in California on which this comparison can be made. These are provisional RWQCB 6 data and EMAP data.

4.11.1 PROVISIONAL RWQCB 6 DATA

California Regional Water Quality Control Board 6 has collected periphyton chlorophyll *a* for numerous streams since 2000. Provisional data for 2000 through 2002, which are still undergoing quality review, were provided to us on the condition that the numerical data not be released or attributed to specific geographic sites. Between 30 and 35 sites were sampled per year, and there are a total of 93 valid data points with both benthic chlorophyll *a* and nutrient data. These represent point measurements, rather than seasonal averages. Unfortunately, no data are available on stream hydrologic regime or light availability.

Figures 4-31 and 4-32 plot the RWQCB 6 benthic chlorophyll *a* observations against TN and TP, respectively. There appears to be a positive correlation with TN in the lower concentration range, but a potential negative correlation for TN concentrations above 1 mg/L. Less of a correlation is evident to TP, and nitrogen appears to be the limiting nutrient in many cases.

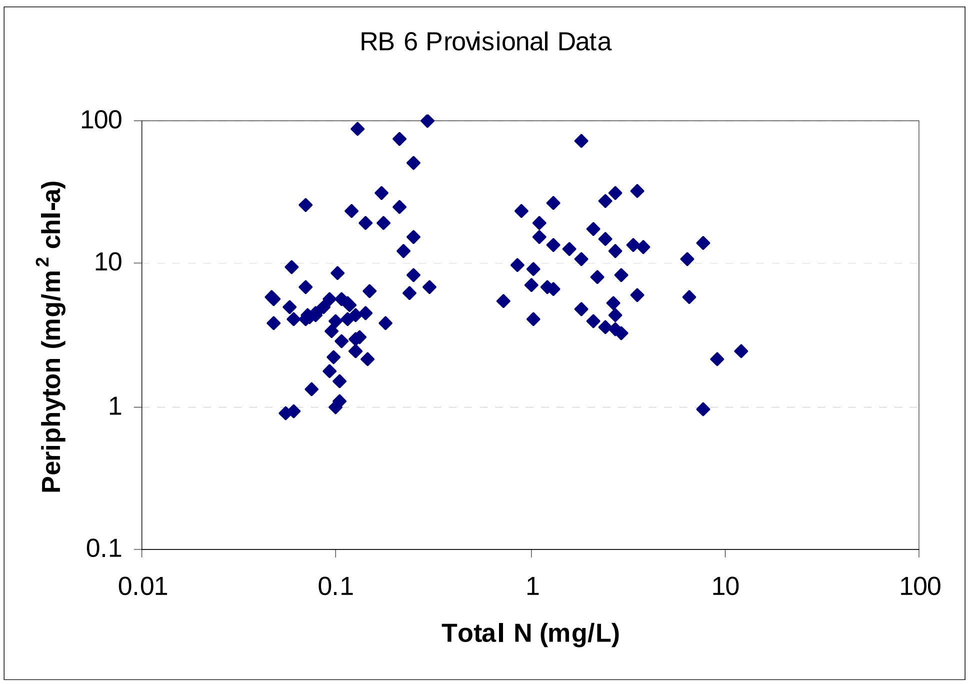


Figure 4-31 Provisional RWQCB 6 Benthic Chlorophyll *a* Observations vs. Total Nitrogen

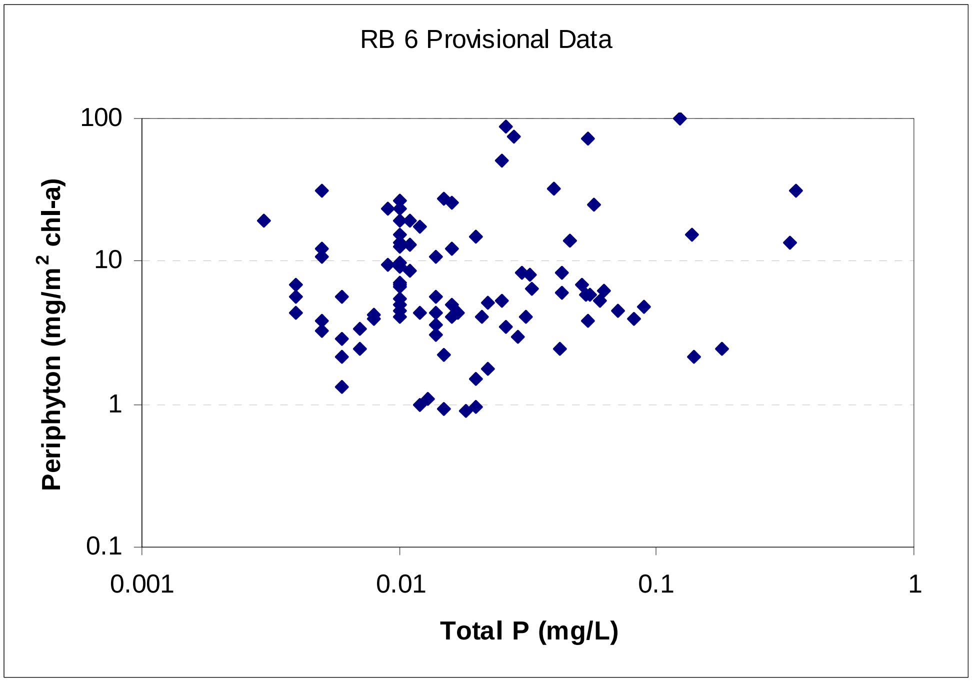


Figure 4-32 Provisional RWQCB 6 Benthic Chlorophyll *a* Observations vs. Total Phosphorus

None of the observations in the data set exceeded 100 mg/m^2 despite concentrations well in excess of the criteria recommended by Dodds.

We developed a nonlinear log-log regression equation for the RWQCB 6 data as a function of TN and TP. Because of the apparent hyperbolic relationship to TN, a nonlinear term on $[\log(TN)]^2$ was included, as in the Dodds *et al*. (1997) work. The resulting regression equation is

$$\log(\text{mean Chl } a) = -3.20 + 2.94 \log(TN) – 0.512 (\log(TN))^2 + 0.0914 \log(TP),$$

with an R^2 of 0.20. This low R^2 is similar to the results found by Dodds *et al*. (2002) in working with the USGS data. The coefficients on TN in this relationship are similar to those in the Dodds *et al*. (1997) model for mean chlorophyll *a*, but the coefficient on log(TP) is much lower – perhaps reflecting a situation in which phosphorus is not often limiting. The net result is a prediction that is lower than predicted by the various Dodds equations based on the literature data set. As noted above, this is the expected result because of the small sample size and the fact that many of these sites are likely subject to limitation by shading and scour.

In Figure 4-33, the RWQCB 6 data are plotted against TN, with the results of three regression equations (which also depend on TP) superimposed. These are the RWQCB 6 data regression and the two equations from Dodds (2002); results of the Dodds (1997) equations are not that different. The power regression against the RWQCB data is lower than both the Dodds equations. The Dodds equation for the mean is within the data, while the Dodds equation for the maximum lies above all but one of the data points, consistent with expectations.

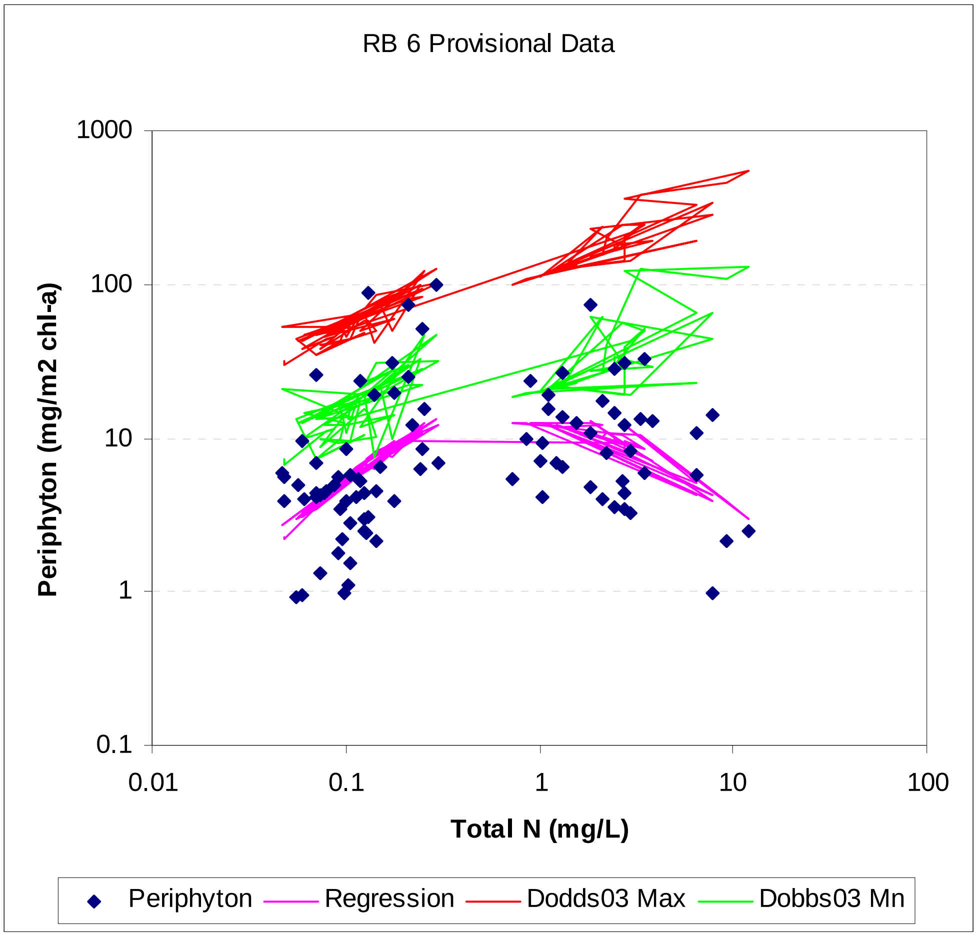


Figure 4-33 SWQCB 6 Periphyton Chlorophyll *a* Compared to Various Regression Equations

The relation of the SWQCB 6 data to the Dodds *et al*. (2002) equation for mean chlorophyll *a* is further explored in Figure 4-34, which shows the deviations of the data from the predicted mean, plotted against the observed value. Within the lower range of observed values, the deviations (predicted minus observed) are consistently greater than or equal to zero, consistent with the assumptions above. In the higher range, the deviations become negative, presumably representing cases in which the point-in-time algal response is greater than the seasonal mean.

Figure 4-35 compares the data to the maximum benthic algal regression from Dodds *et al*. (2002). In all but one case, the predictions are greater than observations, suggesting that the Dodds line does indeed provide an upper bound.

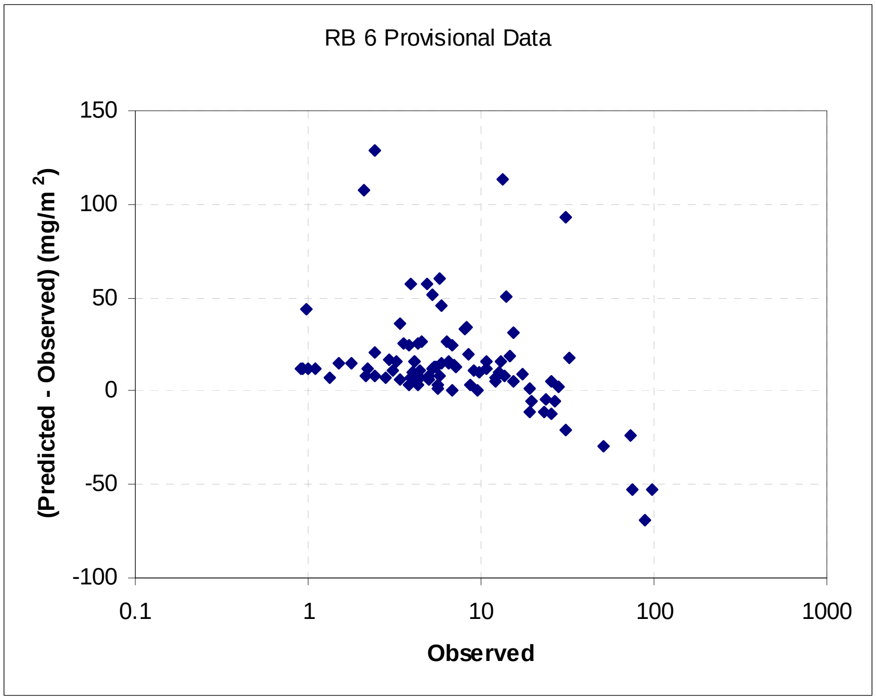


Figure 4-34 Deviations of RWQCB 6 Provisional Benthic Chlorophyll *a* Data (mg/m^2) from Mean Predictions using Dodds *et al*. (2002)

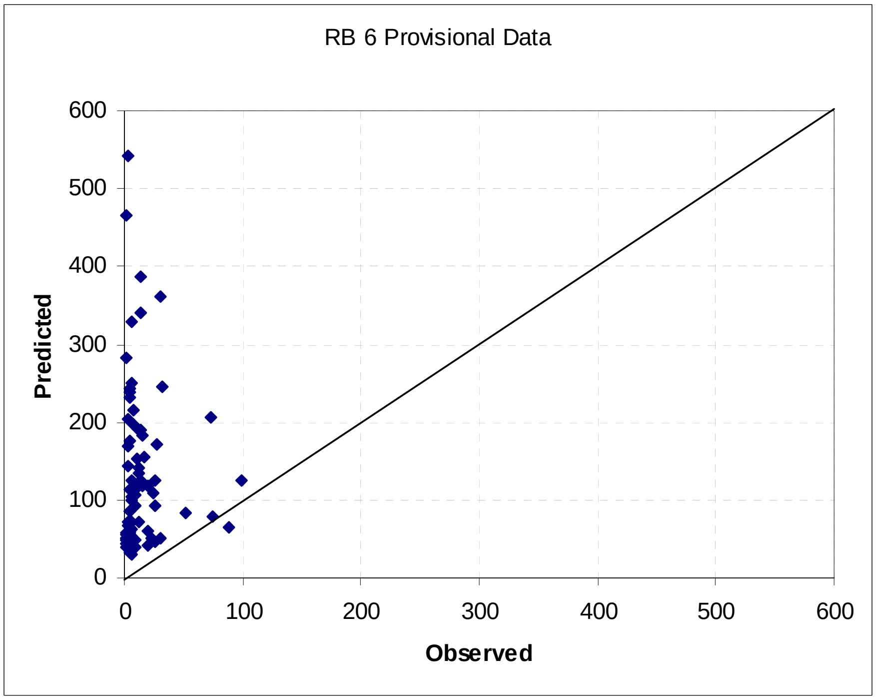


Figure 4-35 RWQCB 6 Provisional Data for Benthic Chlorophyll *a* Compared to Maximum Concentrations Predicted by Dodds *et al*. (2002) Regression

4.11.2 Provisional EMAP Data

Another useful data set was collected by EMAP. This contains 103 data points with both nutrients and benthic chlorophyll *a*, from sites throughout California in 2000-2002. These data are also in provisional form at the time of this writing, and a complete description of the individual site characteristics is not yet available.

Figures 4-36 and 4-37 plot the California EMAP benthic chlorophyll *a* data against TN and TP respectively. As with the RWQCB 6 data, there is an evident positive correlation to TN below concentrations of 1 mg/L (1000 µg/L), while the relationship to TP appears much weaker.

As was done with the RWQCB data, the benthic chlorophyll *a* results, plotted against TN, are shown with the Dodds *et al*. (2002) mean and maximum predictions superimposed in Figure 4-38, while deviations relative to the mean prediction are shown in Figure 4-39. Once again, the data lie near the mean prediction, while the maximum prediction appears to establish a reasonable upper bound. The plot of deviations against the mean shows that the difference between predicted and observed is greater than or equal to 0 at low observed concentrations (reflecting other limiting factors), and tends to be less than zero at high observed concentrations (where the observations are likely to more closely approach their maximum potential).

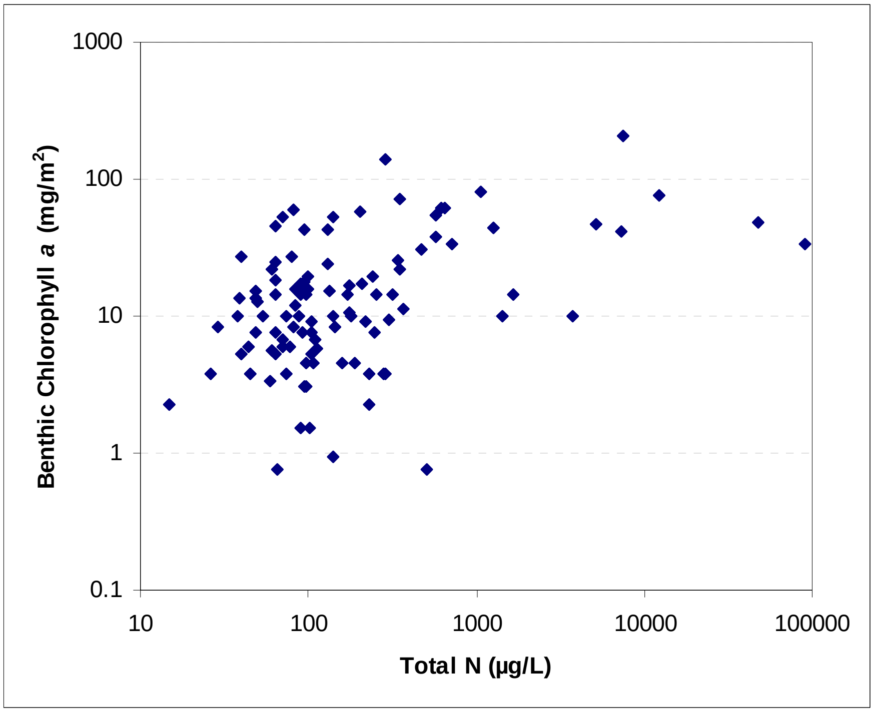


Figure 4-36 EMAP California Benthic Chlorophyll *a* Observations vs. Total Nitrogen

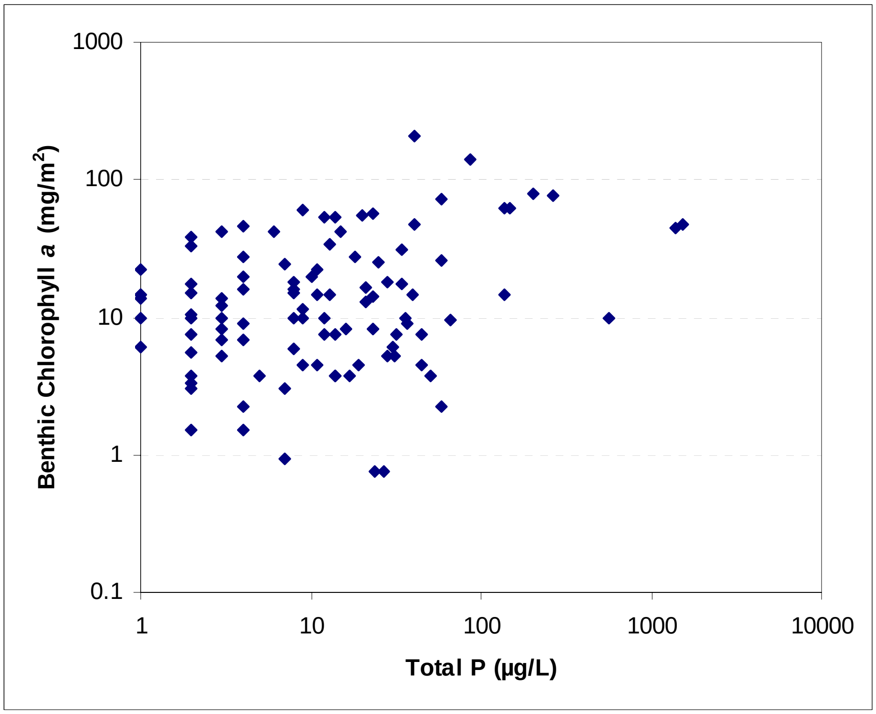


Figure 4-37 EMAP California Benthic Chlorophyll *a* Observations vs. Total Phosphorus

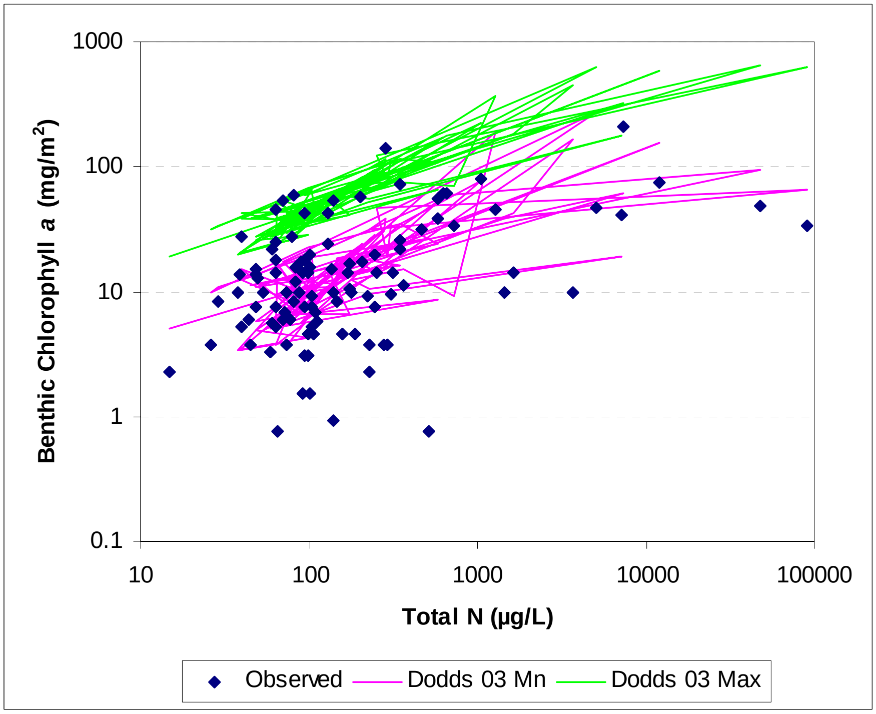


Figure 4-38 EMAP California Periphyton Chlorophyll *a* Compared to Various Regression Equations

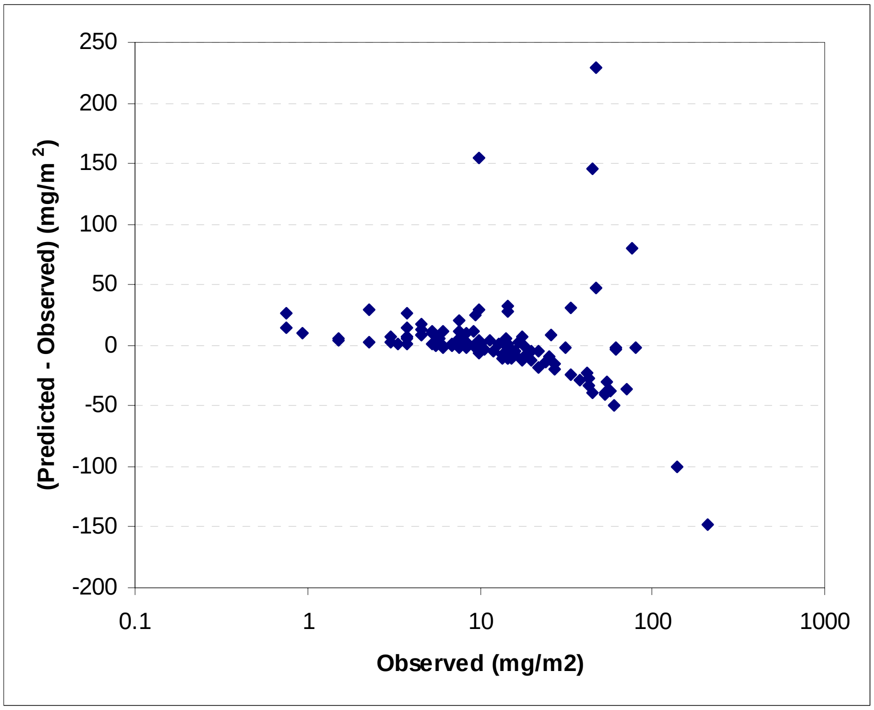


Figure 4-39 Deviations of EMAP California Benthic Chlorophyll *a* Data (mg/m^2) from Mean Predictions using Dodds *et al*. (2002)

4.12 DISCUSSION

Comparison to the RWQCB 6 and EMAP data suggests that the equations proposed by Dodds *et al*. (1997, 1998, 2002) are qualitatively reasonable for predicting mean and maximum potential growth of benthic algae in California streams in the absence of severe light or scour limitation. It should be noted, however, that the Dodds statistical relationships are quite weak, with R^2 values uniformly less than 50 percent. This is believed to reflect the fact that light and scour limitation play important roles in observed chlorophyll *a*. For New Zealand, Biggs (2000) demonstrated that the predictive ability of empirical regression equations could be substantially improved (from an R^2 of less than 0.40 to an R^2 greater than 0.70) by inclusion of a measure of average days of accrual. Presumably, inclusion of a measure of canopy closure might further improve results.

It would be of great interest to re-evaluate Dodds' data set with inclusion of information on average days of accrual, but it may not be possible to obtain the data. Otherwise, the results reported above suggest it is desirable to go to a simple parametric model of benthic algal response. Such a model should be calibrated to be in reasonable agreement with the Dodds regression results. In particular, the Dodds maximum should generally agree with model predictions under conditions of minimal light limitation.

5.0 MODELING IN SUPPORT OF NUTRIENT CRITERIA DEVELOPMENT

A set of models was used to help obtain quantitative estimates of background nutrient loads in different watersheds in the study region (Ecoregion 6 in California) and to relate nutrients levels to possible biological impacts. In both cases the modeling was motivated by the absence of adequate data to perform these assessments in Ecoregion 6. The models uses were:

- Surface Water Assessment Tool (SWAT) for estimating background land-derived nutrient loads
- USGS Spatially Referenced Regressions of Contaminant Transport on Watershed Attributes (SPARROW), for estimating background loads for comparison with SWAT and evaluating nutrient losses during transport through the stream network
- Army Corps of Engineers' BATHTUB model to analyze the water quality response in lakes and impoundments to different nutrient loading scenarios
- Stream periphyton analysis using the benthic algal component from the QUAL2K water quality model

The application of each of these models and their relevance to nutrient criteria development is presented in this section.

5.1 USE OF SWAT TO SIMULATE BACKGROUND NUTRIENT LOADS AND CONCENTRATIONS IN CALIFORNIA

5.1.1 THE SWAT MODEL

5.1.1.1 History

The Soil and Water Assessment Tool, version 2000 (SWAT2000) was developed by the USDA, ARS, and the Texas A&M Spatial Sciences Laboratory with funding from EPA and is incorporated into EPA's BASINS 3.0 water quality modeling system. SWAT, first developed in the early 1990s, is based directly on the Simulator for Water Resources in Rural Basins (SWRRB, Williams *et al*., 1985; Arnold *et al*., 1990) with features from several other ARS models, such as CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems) (Knisel, 1980), GLEAMS (Groundwater Loading Effects on Agricultural Management Systems) (Leonard *et al*., 1987), and EPIC (Erosion-Productivity Impact Calculator) (Williams *et al*., 1984).

SWAT has undergone a series of updates and improvements since its first release. In addition to revisions in model process representation, a complete ArcView GIS interface for the model was developed (Neitsch and DiLuzio, 1999) and subsequently incorporated into the EPA BASINS package (U.S. EPA, 2001). The SWAT model is available as a standalone at http://www.brc.tamus.edu/swat/swat2000.html, and as part of the BASINS package at http://www.epa.gov/waterscience/basins/b3webdwn.htm. However, some changes in the model code are recommended below for California application.

5.1.1.2 Characteristics

SWAT (Neitsch *et al*., 2001) is a long-term, continuous watershed simulation model. This model simulates land cover impacts with weather, soil, topography, and vegetation data. The SWAT simulation and output is organized by Hydrologic Response Units (HRUs), which are areas with homogeneous land cover and soil properties.

Within SWAT, runoff is simulated separately for each HRU using the SCS Curve Number approach, aggregated to the sub-watershed level, and then routed to calculate total runoff and pollutant delivery. The model considers moisture and energy inputs, including daily precipitation, maximum and minimum air temperatures, solar radiation, wind speed, and relative humidity. SWAT simulates a complete set of hydrologic processes including canopy storage and evapotranspiration. It uses the Modified Universal Soil Loss Equation (MUSLE) to model erosion and sediment yield with runoff. MUSLE variables include aboveground biomass, residue on the soil surface, and the minimum C factor for each species. The MUSLE approach differs from the Universal Soil Loss Equation (USLE) in that a delivery ratio to account for trapping within the watershed is not required. Instead, the USLE erosivity factor is replaced with a factor including watershed area, average flow depth, and peak flow that estimates both erosivity and delivery. The empirical parameters for this equation were developed on a set of small watersheds primarily located in Texas and Nebraska.

For channel sediment deposition and degradation, SWAT defines the maximum sediment transport from a reach segment as a function of peak channel velocity. SWAT simulates the nitrogen and phosphorus cycles, including plant uptake of nutrients and the mineralization of organic nutrients in plant residue. SWAT employs a detailed process-based simulation of plant growth and the effects of plant cover on nutrient balances, making it a useful candidate for evaluating unimpacted nutrient balances. The model differentiates between annual and perennial species as well as woody and non-woody species.

SWAT is particularly well suited for application to large river basins in semi-arid western areas, and has seen many applications to such systems. It has the advantage of describing processes using methods for which parameter information is readily available. This facilitates efficient extraction of parameters from land use and soil coverages. It also differs from simpler watershed loading functions in a number of important ways, most notably including a full simulation of instream flow and transport (including diversions and transmission losses) and simulation of irrigation. The model provides continuous simulation on a daily time step.

In comparison to other commonly used watershed models, SWAT has the following advantages:

- SWAT explicitly incorporates elevation or orographic effects on precipitation and temperature.
- SWAT was developed for and has been widely applied to simulation of watersheds in arid regions.
- SWAT explicitly incorporates routines for agricultural diversions and irrigation.
- SWAT includes routines designed to address the impacts on flow and pollutant loading of multiple small (or large) farm ponds within a basin.
- SWAT is designed to use either observed meteorological data or statistically generated meteorology, facilitating the development of long-term analyses.

Because the model is physically based and uses commonly available geographic data, it is claimed “Watersheds with no monitoring data…can be modeled”, allowing the efficient evaluation of “relative impact of alternative input data (e.g., changes in management practices, climate, vegetation, etc.) on water quality…” (USEPA CREM, 2004).

5.1.1.3 *Structural Limitations of SWAT*

While SWAT is a process-based model, it intentionally incorporates simplified representations of most processes so that many parameters can be obtained from readily available geospatial coverages. For upland generation of flow and sediment, SWAT relies on the well-tested, semi-empirical approaches of the SCS Curve Number and MUSLE (although the MUSLE implementation appears to be non-standard, as described below in Sub-Section 5.5.3). The basic time step of the model is one day (although runoff can be simulated at a finer scale using Green-Ampt infiltration); so actual flow hydrographs are not represented. The MUSLE approach is most applicable to the estimation of cumulative sediment loads, rather than loads from individual events.

In SWAT, organic and inorganic nutrients in dissolved and sorbed form are moved from uplands to streams with water and sediment, respectively. Nutrient balances in the soil (as well as the cover index for erosion calculations) are determined by the results of plant growth simulation – which is considerably more complex and difficult to validate. At best, the nutrient loads predicted by the model should also be considered as estimates of cumulative yield, rather than loads from individual events.

Routing within streams adds further limitations to SWAT predictions. Because both upland loads and instream routing are simulated at a daily time step, the model will not provide an accurate representation of intra-event concentrations of even conservative constituents in streams with rapid, “flashy” responses. Nutrient kinetic transformations in the stream were added to SWAT in 1996. The approach taken was to implement the kinetic description contained in the documentation for the QUAL2E model (Brown and Barnwell, 1987). The model tracks nutrients dissolved in the stream and nutrients adsorbed to the sediment. Dissolved nutrients are transported with the water while those sorbed to sediments are allowed to be deposited with the sediment on the bed of the channel. However, it is important to recognize that the current implementation of SWAT does not actually evaluate the time derivatives described in the theory. Rather (in subroutine watqual.f), the routing time for nutrients in a reach is forced to be equal to one day. This means that rate constants are actually implemented as step-function reductions. For instance, if a nutrient transformation rate is 40 percent per day, then 40 percent of the influent nutrients will be lost during transport through a reach, regardless of the actual travel time.

As a result of these compromises, the instream concentrations reported by SWAT are not necessarily realistic representations of instantaneous concentrations. Further, the mass transport through reaches of nonconservative parameters will be realistic only when the reach travel time approximates one day.

5.2 Selection of SWAT for California Background Nutrient Load Evaluation

5.2.1 Pilot Project Evaluation Needs

The SWAT model was selected for the Ecoregion 6 pilot study as one among several lines of evidence to evaluate unimpacted background concentrations and loads of nutrients, where "unimpacted" refers to natural vegetative cover without point source loads. The unimpacted background is important because it establishes a baseline for nutrient criteria. Another important line of evidence for unimpacted background is obtained from observations at unimpacted reference sites. However, truly unimpacted reference sites may be difficult to locate in some regions, and monitoring data for such sites is sparse. In addition, significant variability in nutrient dynamics at potential reference sites may occur due to variability in soils, slopes, precipitation, and vegetation type. Watershed modeling is expected to provide a basis for examining how background loads and concentrations may vary with these and other controlling factors.

SWAT was selected for the upland simulation based on the desirable characteristics cited in the previous section: the model uses readily available geospatial databases, is intended for uncalibrated application, and accounts for differences due to land cover type. In the pilot project, SWAT was applied to smaller headwater watersheds only. The instream transport components were not used, due to concerns identified above. Instead, load generation by SWAT was combined with the empirical transport component of the USGS SPARROW model (Smith *et al*., 1997) to provide an evaluation of accumulated nutrient delivery through larger watersheds.

5.2.2 Previous Experience

SWAT has received wide application in recent years, and a bibliography is available at http://www.brc.tamus.edu/swat/swat-peerreviewed-publications.htm. For example, SWAT has been applied to water balance studies of the entire contiguous United States (Arnold *et al*., 1999), using only geospatial data available at the 1:125,000 scale, and generally produced adequate results. The application did tend to underpredict runoff in mountainous areas (probably due to orographic effects).

Water quality applications with SWAT have been less thoroughly tested and peer reviewed. Santhi *et al*. (2001) recently reported on the successful calibration and validation of a SWAT model for sediment and nutrients for the Bosque River watershed in Texas. This was a calibrated application to a 4300 km^2 watershed dominated by pasture, range, and row crop land uses. The validation focuses on monthly cumulative load delivery, apparently reflecting the fact that point-in-time concentration estimates are less reliable.

The Tetra Tech team also has experience with the SWAT model. Of most relevance to simulation in the arid southwest was development of a complete hydrologic, sediment, and nutrient model of the 5500 square mile Verde River watershed in central Arizona (Tetra Tech, 2001). This application achieved an excellent hydrologic calibration and what appeared to be a good representation of nutrient loading from a wide variety of natural vegetation covers.

SWAT simulations of nutrient loading at the scale of 6-digit hydrologic units have been developed as part of the Hydrologic Unit Model for the United States (HUMUS) project (Srinivasan *et al*., 2000). Maps of predicted nitrogen and phosphorus yield (as kg/ha) are available in an online presentation (http://srph.brc.tamus.edu/humus/slides/index.html), but do not appear to have been formally published or validated against data. These are reproduced below (Figures 5-1 and 5-2).

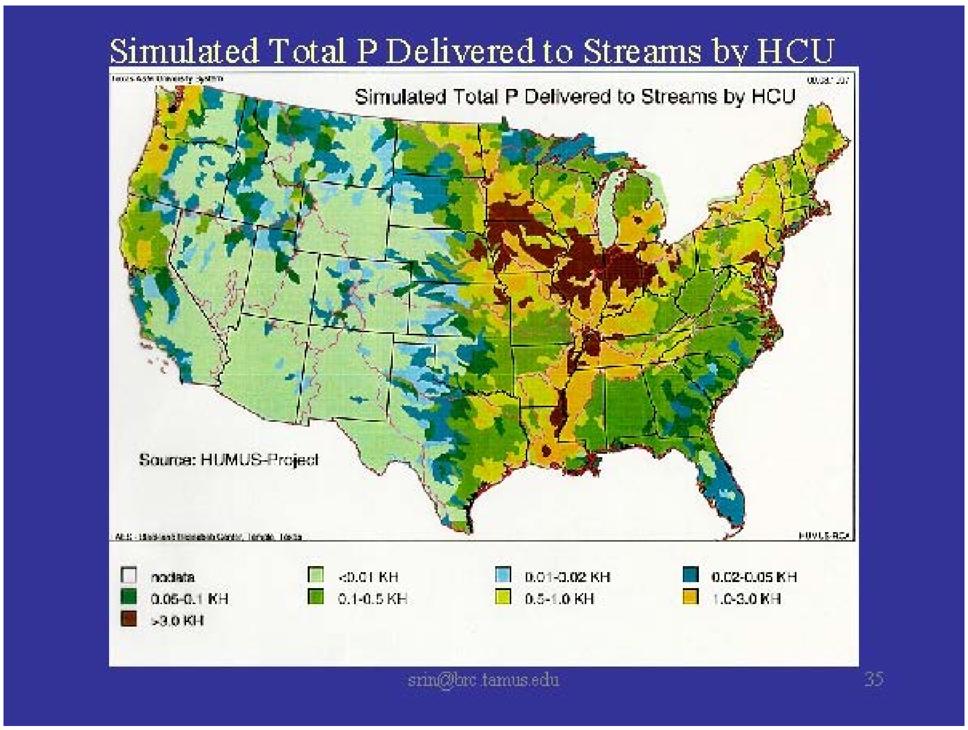


Figure 5-1 SWAT/HUMUS Predictions of Phosphorus Yield

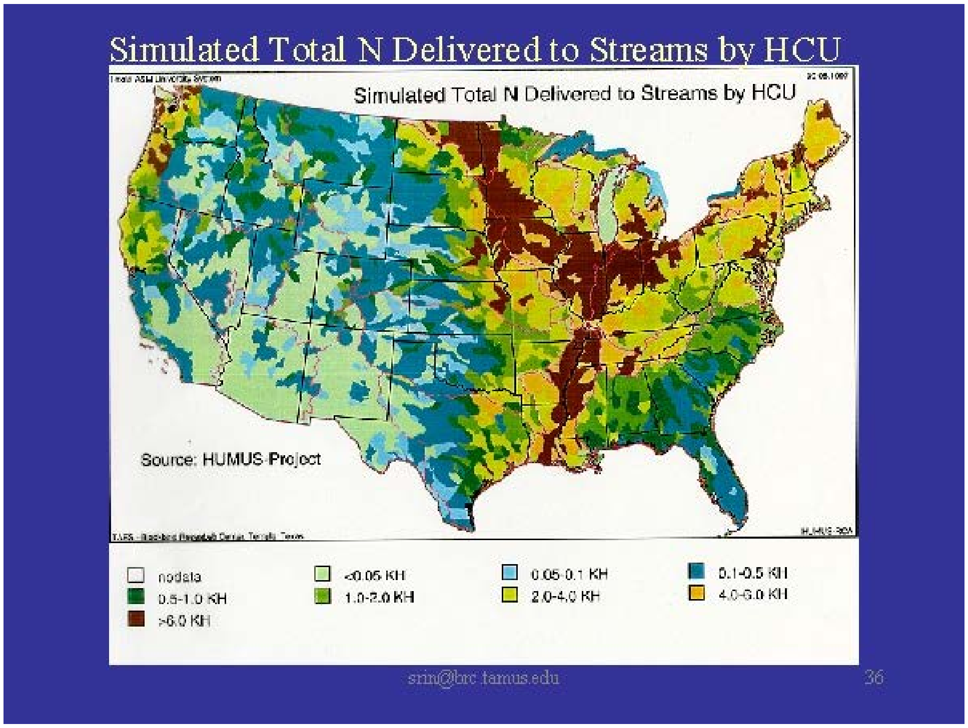


Figure 5-2 SWAT/HUMUS Predictions of Nitrogen Yield

5.3 SWAT Setup for Ecoregion 6 and Initial Testing

5.3.1 Initial Parameter Specification for Ecoregion 6

SWAT applications for natural background load estimation in Ecoregion 6 were initially documented in the 2003 Progress Report (Tetra Tech, 2003). These were further revised and reported in the white paper *Use of SWAT to Simulate Nutrient Loads and Concentrations in California*, prepared for the SWQCB. Both of these earlier efforts have now been superseded by the results presented herein.

The strategy for the application was to commence with SWAT default parameters, then diagnose and modify inputs as needed. Applications of SWAT used California land cover (derived from the GAP analysis), California soils (STATSGO), California meteorological stations, and an appropriate Digital Elevation Model. The BASINS-SWAT interface with used to generate all input files, with only minor modifications. This approach was consistent with the intention of using SWAT as a tool that enables rapid evaluation without detailed, site-specific calibration.

5.3.2 Physical/Chemical Parameters

Several groups of SWAT input must be input manually, as follows:

5.3.2.1 Elevations and Lapse Rates

In areas of high relief, precipitation and temperature can vary significantly with elevation. SWAT can simulate these effects using lapse rates, but does not set these up by default. To activate this simulation it is necessary to specify an "elevation band" and associated parameters as follows:

Table 5-1
Augmented input for elevation and lapse rates.

Description	Input	File	Default	Revised
Elevation at the center of the elevation band (m)	ELEVB(1)	.sub	Optional	ELEV
Fraction of subbasin area within elevation band	ELEVB_FR(1)	.sub	Optional	1.00
Precipitation lapse rate (mm/km)	PLAPS	.sub	0.0	1.06
Temperature lapse rate (°C/km)	TLAPS	.sub	0.0	-6.00

5.3.2.2 Groundwater and Rainfall Nutrient Concentrations

Significant nutrient fluxes – particularly for nitrate – may occur via groundwater pathways. SWAT simulates shallow (only) groundwater flow, but does not calculate nutrient mass balances in groundwater. The SWAT user must specify the nitrate and phosphate concentrations in groundwater, otherwise they default to zero.

Shallow groundwater concentration data are difficult to locate. The United States Geologic Survey maintains a database on the Internet (USGS, 2004) that the user can query by county, hydrologic unit, period of record, and well depth, among other attributes. However, well depths are typically large (as the focus is on potable water supplies), and so the data may not be representative of groundwater contributions to streamflow. Further, the concentration discharged into streams is often less than the concentration in adjacent ground water due to biological uptake in stream sediment.

For the current iteration of the validation study, the groundwater concentrations were modified slightly from those used in the initial pilot project. There, the concentration of nutrients in the groundwater contribution to stream flow was determined from California monitoring data for minimally impacted steams during base flow periods. The resultant average values were 0.03 mg N/L for nitrate concentration and 0.01 mg-P/L for soluble phosphorus concentration. These values were revised upwards to 0.04 mg-N/L and 0.015 mg-P/L (Table 5-2) In fact, base flow concentrations may differ significantly from stream to stream, and use of a regional average value may contribute significantly to uncertainty in model predictions of nutrient concentrations.

SWAT also simulates the contribution of nitrate in rainfall. Only nitrate flux (not ammonium) is simulated, and this is added directly to the soil moisture profile, not partitioned into direct runoff. The sum of nitrate plus ammonium concentration in rainfall can be entered as nitrate in the model to approximate the total atmospheric wet deposition load. The SWAT interface defaults this concentration to 1 mg/L (as N). Information on atmospheric deposition of nitrogen was gathered from the National Atmospheric Deposition Program (NADP, 2004). Isopleth maps of year 2002 nitrate ion concentration and ammonium ion concentration were downloaded from the NADP website. The maps provided data for six monitoring stations located in the state of California. Nitrate and ammonium concentration data were converted to concentration as nitrogen values and then summed to provide a total nitrogen concentration at each station. The final values used in the current validation modeling were 0.30 mg N/L for the northern portion of the ecoregion, and 0.50 mg N/L for the southern region, as shown in Table 5-2.

Table 5-2
Augmented input for groundwater and rainfall nutrient concentrations

Description	Input	File	Default	Revised
Ground Water Nitrate Concentration (mg N/L)	GW_NO3	.gw	0.0	0.04
Ground Water Soluble Phosphorus Concentration (mg P/L)	GWSOLP	.gw	0.0	0.015
Concentration of nitrate in rainfall (mg N/L)*	RCN	.bsn	1.0	0.3 northern; 0.5 southern

5.4 KINETIC FACTORS

Several kinetic factors also required correction from the values created by default in the SWAT GIS interface (Table 5-3). The soil compensation factors (which control the distribution by soil layer of withdrawal of water and phosphorus by plants) need to be updated from the default of zero to values that are more appropriate for arid climates with deep-rooted plants. Finally, the rate factor for humus mineralization of organic nitrogen appears to be set improperly by the interface to 0.003, whereas the manual states that the default should be 0.0003.

Table 5-3
Augmented input for kinetic factors

Description	Input	File	Default	Revised
Soil Evaporation Compensation Factor	ESCO	.hru	0	0.5
Plant Uptake Compensation Factor	EPCO	.hru	0	1.0
Rate factor for humus mineralization of active organic nitrogen	CMN	.bsn	0.003	0.0003

5.4.1 LAND COVER/CROP DATABASE

SWAT simulates the removal of water and nutrients from the root zone, transpiration, and biomass yield production based on the combination of soils and the biophysical properties of the land cover. The primary purpose of the SWAT application is to estimate nutrient loading characteristics under natural conditions of native vegetation cover in Ecoregion 6. Although it is a single ecoregion, native vegetation shows considerable variability within the ecoregion, and it is important to specify vegetation types that match the site-specific conditions. Such information is provided by the California GAP Analysis. The GAP land use dataset developed from satellite flyover analyses in the early 1990s was used to estimate the distribution of land cover within each delineated watershed. Human-influenced land uses (e.g., urban areas, agricultural land) in the watersheds were converted to undisturbed cover for simulation of unimpacted watersheds.

The National Gap Analysis Program (GAP) has developed this data to provide detailed information on the distribution of common species (USGS, 2003). The data for California provides a list of co-dominant species and habitat type for each small-scale polygon. It provides more detailed information about vegetation type compared to MRLC land cover data. About 5 to 10 polygons were found within each reference watershed.

The SWAT input parameters required to simulate the biophysical processes of each land cover are shown in Table 5-4.

Table 5-4
Input Variables for the SWAT Land Cover/Plant Growth Database

Variable	Description
BIO_E*	Radiation use efficiency in ambient CO_2 (kg/ha)/(MJ/m^2)
HVSTI	Potential harvest index for the plant at maturity given ideal growing conditions
BLAI*	Potential maximum leaf area index for the plant
FRGRW1	Fraction of the growing season corresponding to the 1st point on the optimal leaf area development curve
LAIMX1	Fraction of the maximum plant leaf area index corresponding to the 1st point on the optimal leaf area development curve
FRGRW2	Fraction of the growing season corresponding to the 2nd point on the optimal leaf area development curve
LAIMX2	Fraction of the maximum plant leaf area index corresponding to the 2nd point on the optimal leaf area development curve
DLAI	Fraction of growing season at which senescence dominates growth
CHTMX*	Plant's potential maximum canopy height (m)
RDMX*	Maximum rooting depth for plant (mm)
T_OPT*	Optimal temperature for plant growth (^oC)
T_BASE*	Minimum temperature for plant growth (^oC)
CNYLD	Fraction of nitrogen in the yield
CPYLD	Fraction of phosphorus in the yield
BN1*	Normal fraction of nitrogen in the plant biomass at emergence
BN2*	Normal fraction of nitrogen in the plant biomass at 50% maturity
BN3*	Normal fraction of nitrogen in the plant biomass at maturity
BP1*	Normal fraction of phosphorus in the plant biomass at emergence
BP2*	Normal fraction of phosphorus in the plant biomass at 50% maturity
BP3*	Normal fraction of phosphorus in the plant biomass at maturity
WSYF	Harvest index for the plant in drought conditions, the minimum harvest index allowed for the plant
USLE_C	Minimum value of USLE C-factor applied to the land cover or plant
GSI	Maximum stomatal conductance in drought conditions

Table 5-4
Input Variables for the SWAT Land Cover/Plant Growth Database

Variable	Description
VPDFR	Vapor pressure deficit corresponding to the fraction stomatal conductance defined by FRGMAX
FRGMAX	Fraction of maximum stomatal conductance that is achievable at a high vapor pressure deficit
WAVP	Rate of decline in radiation-use efficiency per unit increase in vapor pressure deficit (kg/ha)/(MJ/m2)
CO2HI	Elevated CO2 atmospheric concentration (ppmv)
BIOEHI	Radiation use efficiency at elevated CO2 atmospheric concentration value for CO2HI (kg/ha)/(MJ/m2)
RSDCO_PL	Plant residue decomposition coefficient

* The literature search focused on these variables.

To simulate the growth of Southern California vegetation, each land cover class required the input variables listed in Table 5-4. Therefore, each GAP polygon needed a unique land cover identifier associated with a series of input variables. The GAP species and plant types were aggregated into thirteen groups according to genus and plant type, and each genus-plant type had one set of SWAT input values. The genus-plant types were chosen so that each type included at least one species with SWAT input variables. The thirteen genus-plant types are the following:

1. Chaparral-Adenostoma
2. Chaparral-Arctostaphylos
3. Chaparral- Buckbrush (Ceanothus)
4. Chaparral- Hoaryleaf ceanothus
5. Chaparral- Desert ceanothus
6. Chaparral-Scrub Oak
7. Chaparral- Mountain mahogany
8. Chaparral- Sugarbush
9. Scrub-Shrub
10. Conifer
11. Hardwood-Blue Oak
12. Hardwood-Coast Live Oak
13. Herbaceous

The input values were acquired from the SWAT Land Cover/Plant Growth database and from literature values obtained for individual species. SWAT provided default values for several of the variables listed in Table 5-4. The SWAT Land Cover/Plant Growth database also contained input values for pine, oak, poplar, and honey mesquite trees as well as the more general deciduous, mixed, and evergreen forest classes. The GAP data defined annual grassland as including *Avena* and *Bromus* species; for this land cover, the SWAT input values for *Avena sativa*, *Bromus inermis*, and *Bromus biebersteinii* can be used.

The estimation of input values focused on radiation use efficiency, height, root depth, leaf area index, base and optimal growth temperatures, and nutrient biomass concentrations. A literature search was conducted for key species. Maximum heights were found for all species. Leaf area indices for most species were acquired, and several values for rooting depth and nutrient biomass concentration were found. According to the literature search and personal communication with SWAT co-developer Jim Kiniry, the literature does not provide estimates of radiation use efficiency (RUE; in biomass units) for the Southern California species. With Dr. Kiniry's advice, Tetra Tech estimated RUE by comparing leaf area indices of Southern California species with other species for which RUE values were available.

5.4.2 VALIDATION APPROACH

Results from uncalibrated SWAT applications to nine watersheds in Ecoregion 6 were reported in Tetra Tech (2003), and a number of additional watersheds were modeled subsequently. Results appeared generally reasonable, but some problems were noted, particularly in apparent over-estimation of erosion and sediment-associated nutrient loads. It is important, however, to go beyond these qualitative comparisons to examine with greater rigor whether results provided by the model are reasonable. To this end, a validation process was attempted. The validation exercise considered two lines of evidence: comparison to monitored data from unimpacted streams, and comparison of SWAT results to other regional estimates of nutrient loads.

Comparison of model results to monitored data for small, unimpacted watersheds would provide the strongest test of model performance; however, ability to conduct such tests is hampered by data availability. In California, the majority of streams that have been intensively monitored are impacted streams that have extensive agricultural or urban land use and/or hydrologic modification. Much of the monitoring is focused downstream of point sources, which is useful for evaluating wasteload allocations, but of limited use for assessing unimpacted background loads. Where unimpacted streams are monitored, the data are often sparse and intermittent, and frequently do not include a complete representation of nutrient species. This presents a particular problem for comparison to SWAT, where we expect the long-term loads of total nutrients to be more accurate than estimates of the concentration of individual nutrient species.

An alternative source of information on expected nutrient loads is provided by the SPARROW work conducted by USGS (Smith *et al*., 2003; Smith *et al*., 1997). The complete SPARROW model includes empirical regression models of upland load generation, based on detailed analysis of USGS NASQAN data. Two versions are available. Smith *et al*. (2003) present estimates of upland loading that are based on runoff regime, a regional indicator, and a correction for atmospheric deposition of nitrogen. Smith *et al*. (1997) provide a more complex approach in which the nutrient yield from the land surface is a function of fertilizer application, livestock waste production, atmospheric deposition, and area of nonagricultural land, while delivery to water depends on soil permeability, stream density, and (for nitrogen only) average temperature. While calibrated to similar data sets, the two estimators can produce rather different results for individual watersheds.

5.4.3 SELECTION OF VALIDATION WATERSHEDS

Validation watersheds were chosen in a manner consistent with the pilot study, in which the SWAT model was used to simulate loadings from small headwater watersheds in unimpacted areas. To this end, validation watersheds were chosen based on their size (ideally 10,000 to 40,000 acres) and their location in areas unimpacted by urban or agricultural loads. Additionally, watersheds were chosen to provide a representative sample across Ecoregion 6 and where there was as much monitored data as possible. Nine validation watersheds were chosen for the study; their locations and summary information are shown in Figure 5-3 and Table 5-5, respectively.

Most of the monitoring stations for which large data sets were available are located on impacted streams, as discussed above. Excluding these watersheds left a much smaller subset of stations from which to choose, and the criteria listed above were found to be too restrictive to obtain a reasonable population of examples. For example, watershed 455 is 128,000 acres, but was included in the validation subset because it was the only available station choice located in the Sierra Nevada foothills. Note that the number of available water quality nutrient samples is quite small for these watersheds, ranging from 7 to 31 data points.

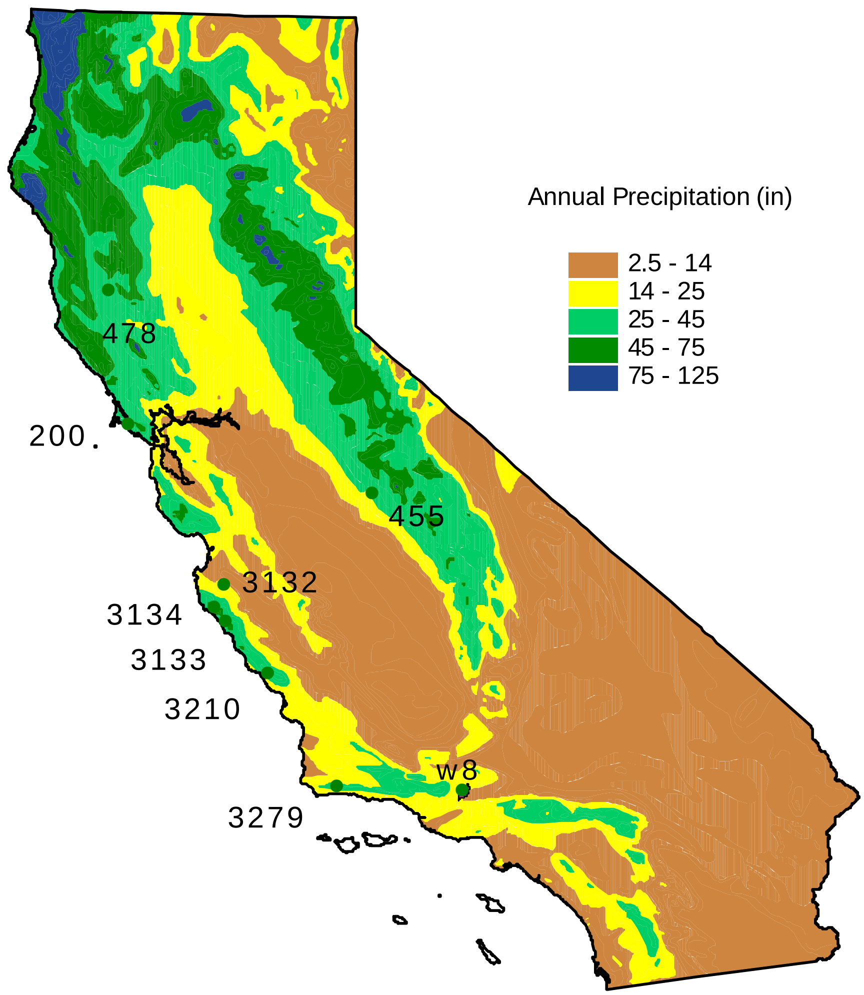


Figure 5-3   Location of Validation Watersheds

Table 5-5
SWAT Validation Watersheds

Watershed	County	Area (ac)	Average Elevation (m)	Dominant Cover	Observation Count
200	Marin	8,030	134	Conifer	19
455	Mariposa	128,223	672	Hardwood, Conifer	7
478	Mendocino	59,972	275	Chaparral, Pasture, Hardwood	7
3132	Monterey	16,294	722	Grass, Hardwood, Chaparral	22
3133	Monterey	14,216	1,067	Chaparral, Hardwood, Redwood	23
3134	Monterey	45,640	406	Hardwood, Chaparral	31
3210	San Luis Obispo	16,294	346	Grass, Hardwood	20
3279	Santa Barbara	16,924	263	Grass, Hardwood	23
w8	Ventura / Los Angeles	28,936	524	Grass, Shrub	*

* Summary information only available from Systech (2002)

5.4.4 INITIAL VALIDATION RESULTS

Initial validation comparisons using SWAT default parameters revealed only limited correlation between SWAT output, monitoring data, and the two SPARROW methods (Smith *et al*., 1997; 2003). In many cases, the monitoring data also deviated significantly from the SPARROW estimates. Initial applications showed load to streams in the range of 1.5 to 7 kg/ha total N and 0.2 to 1.5 kg/ha total P, which appear high for this region. Smith *et al*. (1997) suggest that half of the watersheds in California (including impacted watersheds) should yield less than 5 kg/ha total N, while data in Smith *et al*. (2002) imply that estimates of natural background load delivered to water courses in the xeric west (prior to channel losses) are unlikely to exceed about 3 kg/ha total N or 1 kg/ha total P as an upper bound, with delivered loads at downstream monitoring stations much smaller. In addition, sediment yield estimates appeared too high.

Some of the discrepancies in the SWAT output were determined to result from problems in the setup of parameters for simulation of erosion and biomass, largely related to deficiencies in the SWAT model setup interface. These contributed to an over-estimation of sediment load, but were not the sole cause. In addition, the simulated water yield for some watersheds deviated significantly from that expected.

Comparison of SWAT results to concentrations from monitoring data is problematic, as noted above. Not only is SWAT expected to be a better predictor of yield than concentration, but also the monitoring data are very sparse and potentially biased by small sample sizes. In addition, complete nutrient loads are lacking for many of the validation watersheds, particularly the organic phosphorus component. In most cases the SWAT median concentration for total nutrient was less than the observed mean, while the SWAT flow-weighted mean was greater than the observed mean concentration.

Concentration results for individual nutrient species and for wet and dry seasons showed considerable discrepancies between the model and data, and many of the differences between individual watersheds are not captured. These results may, however, be due to uncertainties in the limited observed data as well as to uncertainties in the model.

In general, the initial attempt at validation with SWAT defaults was judged to be unsatisfactory. Therefore, a series of detailed analyses were undertaken to improve simulation, as described in Section 5.5.

5.5 MODIFICATIONS TO SWAT DEFAULTS FOR CALIFORNIA APPLICATION

The initial validation runs revealed a number of potential problems with the SWAT model setup from defaults, even after the site-specific data specifications described in Section 5.3.1. Diagnostic work with the model determined that erosion tended to be over-estimated, while biomass was often unrealistically low, and water yield often did not match expectations. These problem areas and proposed solutions are described below.

5.5.1 PRECIPITATION AND FLOW SIMULATION

The default SWAT setup assigns a precipitation station for long-term statistical simulation of weather to a watershed based on proximity. A total of 53 weather stations are provided for the State of California. A correction is made for elevation of a watershed relative to a weather station to account for higher precipitation rates typically occurring at higher elevations (lapse rate). In the California terrain, however, precipitation can be highly variable locally, depending on orientation relative to the Pacific and rain shadow effects, and the lapse rate correction is often not sufficient to extrapolate correct precipitation to a watershed from a relatively sparse network of weather stations.

Consistency in runoff simulation was checked by comparison of 50-year simulation runs to the expected average annual runoff estimated by USGS (Gebert *et al*., 1987).

Potential problems are readily apparent in the area around validation watersheds 3132 and 3134 in Monterey County (Figure 5-4). For both of these watersheds, the nearest weather generator station is Salinas 3SE. Expected annual water yield at Salinas is between 1 and 2 inches per year; however, expected water yield at 3132 is 5 inches, and water yield at 3134 is greater than 20 inches per year. Obviously, simulation of these watersheds using the Salinas precipitation record is likely to yield erroneous results.

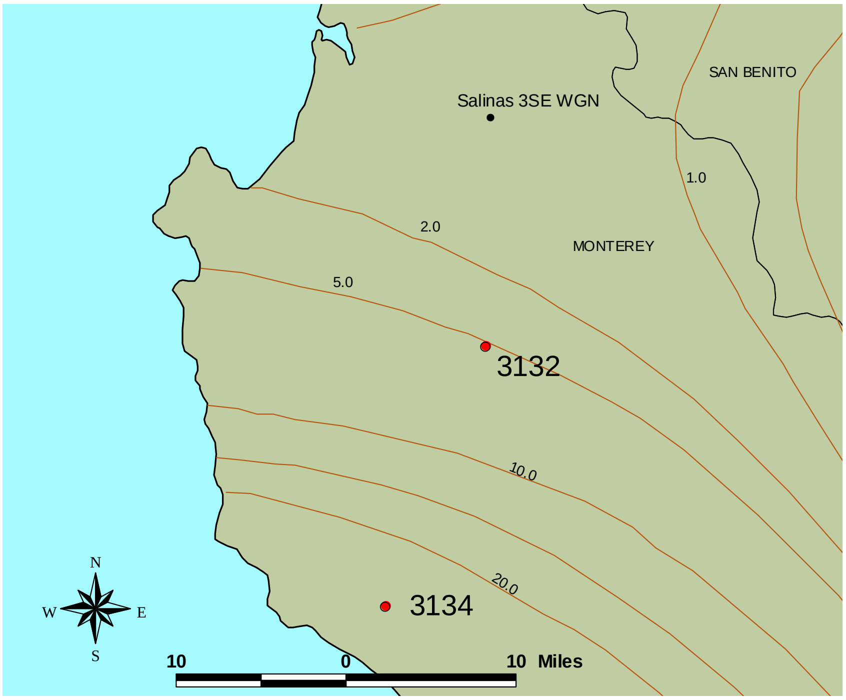


Figure 5-4 Expected Runoff (in/yr) in the Salinas Area (Gebert *et al*., 1987)

This problem can be addressed by using SWAT's options for climate change studies. Within the .sub file, the variable RFINC specifies a percentage change in precipitation, which can be used to adjust the long-term average runoff into the range expected from Gebert *et al*. Five of the ten validation watersheds required adjustments of this sort to fall near the expected range, as shown in Table 5-6.

Table 5-6
Adjustments to SWAT Runoff Simulation

Watershed	Weather Station	Expected Runoff (in/yr)	Unadjusted Runoff (in/yr)	Rainfall Adjustment Percentage	Adjusted Runoff (in/yr)
200	San Francisco	5 - 10	9.0	0	9.0
455	Yosemite	10 – 15	20.0	-25	12.2
478	Willits	~20	36.9	-35	19.7
3132	Salinas	~5	7.0	0	7.0
3133	Salinas	~20	10.8	+75	20.9
3134	Salinas	~20	11.2	+75	23.4
3210	Paso Robles	10- 15	1.7	+100	11.4
3279	Lompoc	2 – 5	3.4	0	3.4
w8	Fairmont	2 – 5	5.8	0	5.8

Note: Expected runoff derived from Gebert *et al*. (1987). SWAT adjustments are made through variable RFINC in the .sub file.

5.5.2 EROSION AND SEDIMENT YIELD

Initial simulations predicted very high sediment yields, with loads from some combinations of cover and soil greater than 100 metric tons/ha/yr. A number of factors contributed to the unrealistic sediment yield simulation. Problems were first identified with SWAT parameterization of initial residue cover and slope length. These issues proved, however, to be only part of the problem. A more significant issue occurs with the SWAT implementation of MUSLE, which must be addressed with a modified and recompiled version of the code.

5.5.3 SWAT MUSLE IMPLEMENTATION

The MUSLE equation for sediment yield is given by Williams (1995) as

$$sed = 11.8 \cdot (Q_{surf} \cdot q_{peak} \cdot A)^{0.56} \, K \cdot LS \cdot C \cdot P \cdot CFRG \, ,$$

in which *sed* is the sediment yield on a given day (metric tons), Q_{surf} is the surface runoff depth (mm/ha), q_{peak} is the peak runoff rate (m^3/s), A is the area (ha), K, LS, C, and P are the USLE erodibility, length-slope, cover, and practice factors, respectively, and $CFRG$ is a correction factor for coarse fragments. The two parameters (11.8 and 0.56) were derived based on a number of study watersheds in Texas and Oklahoma and may need to be adjusted for other parts of the country – although no provision for modification is made in the SWAT input deck.

A common procedure in SWAT sediment calibration is to adjust the practice (*P*) factor until agreement with observations is obtained. This seems inadvisable; it would be better to adjust the MUSLE parameters and use *P* values as recommended in NRCS handbooks.

Within the SWAT2000 code, the product of the MUSLE multiplicative factor (11.8), the USLE *K*, *LS*, *C*, and *P* factors, and the coarse fragment factor are calculated in subroutine soil_phys.f. Subroutine ysed.f adjusts this factor for the current residue cover, storing the result as *cklsp*. MUSLE sediment yield is then calculated as *sedyld* = (*surfq* * *peakr* * 1000 * *hru_km(j)*)** .56 * *cklsp(j)*, where *sedyld* is the sediment yield in metric tons, *surfq* is surface runoff for the day (mm), *peakr* is the estimated peak runoff (m^3/s), and *hru_km* is the area of the hydrologic response unit in square kilometers. The factor 1000 is apparently intended to convert the area to hectares. This, however, is incorrect, as the factor to convert square kilometers to hectares is actually 100. As a result, it appears that SWAT2000 will tend to over-estimate MUSLE sediment yield by a factor of $10^{0.56} = 3.63$.

To remedy this problem, a revised version of SWAT2000 (swatTt.exe) was created, containing the correct units conversion. Use of this modified form of the model can then be selected by setting ISPROJ = 5 in the .cod input file.

5.5.4 INITIAL RESIDUE COVER

Rainfall detachment of soil particles is mitigated by the presence of organic residue on the surface. The initial residue cover (RSDIN in the .hru file) is set by the SWAT interface to 0, which is inappropriate for untilled natural vegetation. Further, in arid climates, residue buildup may be slow, affecting many years of simulation and potentially depleting the soil by erosion before a realistic biomass growth simulation is obtained. To remedy this issue, it is necessary to specify an initial value of RSDIN. Few data were located on this parameter; however, an estimate of 150 kg/ha appeared to provide reasonably stable estimates of sediment yield.

5.5.5 SLOPE LENGTH

The MUSLE erosion calculation includes a Length-Slope (LS) factor, which is calculated from slope length. The SWAT GIS interface attempts to calculate an appropriate slope length from the DEM. However, this calculation does not always succeed when slopes are steep. When a slope length is not calculated, the interface defaults to a slope length of 50 m. The default slope length of 50 m is appropriate for relatively flat watersheds, but in watersheds with steep average slopes (> 25 percent), SWAT will simulate excessive sheet erosion. The average slope length was revised from the default 50 m to 5 m for watersheds with steep average slopes. This revision notably reduced estimated sediment loading in five steeply sloping watersheds, as shown in Figure 5-5, with correspond reductions in estimated nutrient loads. For the more gently sloping watershed 455, BASINS calculated a slope length of 15 m during model set-up and a revision to the slope length was not necessary.

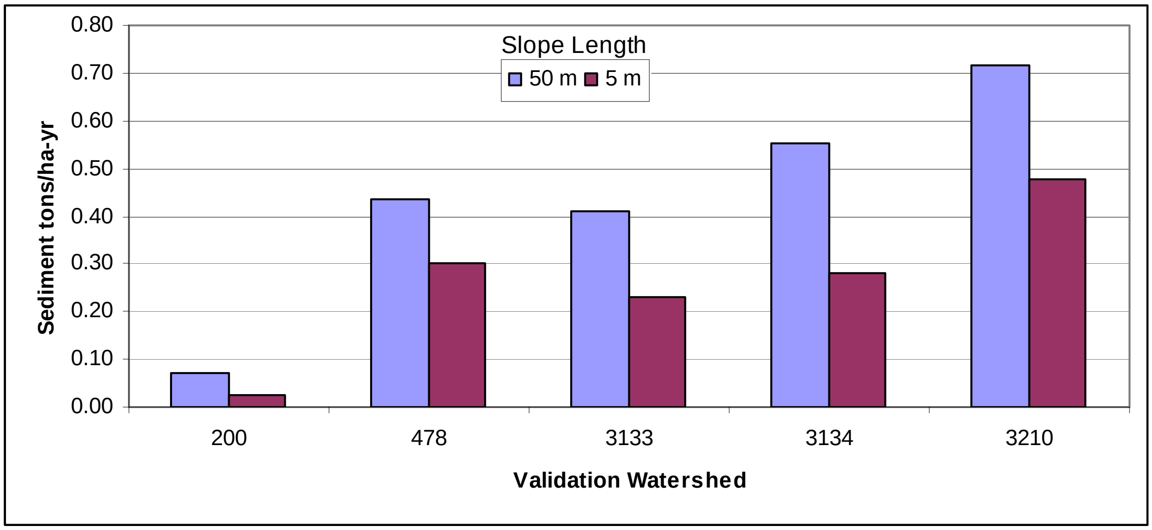


Figure 5-5 Comparison of Sediment loads with 50 m and 5 m slope lengths for 5 validation watersheds (prior to code modification for MUSLE)

5.5.6 Notes on Sediment Simulation for Urban Land

While not relevant to the simulation of natural background conditions, it should also be noted that the default SWAT algorithm may yield unrealistic results from HRUs that contain a mix of urban pervious and impervious land cover because MUSLE is calculated with the peak flow from the entire HRU, using a weighted curve number, and not from the flow from the pervious section. This is equivalent to assuming that all impervious area runoff proceeds as sheet flow across the pervious sections, rather than being piped or channelized, and can result in a significant over-estimation of sediment load from developed areas.

5.5.7 Revised Sediment Results

Making the changes noted above results in a large reduction in predicted sediment yields. The final average annual yield rates range from less than 1 to 4.92 metric tons/ha/yr, which appears more in line with expectations for natural loading rates than the extremely high yields initially predicted by SWAT (Table 5-7).

Table 5-7
Revised SWAT Predictions of Sediment Yield

Watershed	Runoff (in/yr)	Sediment Yield (MT/ha/yr)
200	9.0	0.006
455	12.2	0.061
478	19.7	0.906
3132	7.0	0.179
3133	20.9	4.921
3134	23.4	4.618
3210	11.4	2.325
3279	3.4	0.756
w8	5.8	2.577

5.5.8 BIOMASS SIMULATION

Significant problems were also noted with the SWAT biomass simulation for native vegetation in the validation watersheds, which in turn introduced errors into the erosion and nutrient simulations. SWAT develops the biomass simulation primarily on the basis of information in the CROP database, which was modified to reflect appropriate parameters for regional cover types. However, these parameters also interact with information in the .MGT file for each HRU. Default information entered into this file by the SWAT interface creates problems for a realistic simulation.

SWAT simulates plant growth using a heat unit approach that is most appropriate for annuals, but can be used for perennials (only the new growth biomass, not woody biomass, is simulated by the program). Plant heat units are defined as the difference between plant base temperature and the daily temperature, summed for all days when temperature is above the base temperature, while base zero heat units are calculated relative to freezing. By default, SWAT specifies a "planting" (growth start) operation and a "kill" (stop growth and convert to residue) operation for each cover. Planting time defaults to 15 percent of the base-zero heat units, and the kill operation corresponds to 120 percent of the plant heat units required for maturity. The program will also calculate potential heat units (PHU) required to reach maturity.

Each of these entries is problematic, for different reasons. The PHU calculation in the SWAT interface should be accurate, given an appropriate estimate of growing season length, even though the user would have to enter this value manually rather than retrieving it from the CROP database. However, the calculation is carried out in a dynamic link library (PHU.DLL) that appears to have an undocumented, internal limit of 100 different crop/vegetation types. When more than 100 crop types are added to the database (as in the California application), the PHU for all cover types with index greater than 100 is set to zero. This in turn causes the SWAT program to assume a default PHU of 700. In Southern California, reversion to the default value of PHU means that the plants mature and reach senescence in only a few months, resulting in an underestimate of biomass and leaf area.

The default number of heat units (HUSC) that control start and stop of growth are appropriate for an annual crop that is planted after danger of frost and allowed to dry-down before harvest, but not for many annuals. Setting the initiation of growth at the default of 15% of the base zero heat units means that trees do not start leafing out until the beginning of April in many of the test watersheds, which is clearly too late. Finally, setting the heat units for the "kill" operation to the default of 1.20 is probably too large for woody plants in arid areas, as it would mean that leaves are likely to be retained for a long period after the completion of the annual growth cycle.

It is also important to recognize that SWAT has a built-in, hardwired dormancy period for perennials that is based on day length as determined by latitude. This cannot be modified by the user, and forces growth to stop during the shorter days of December and January. In fact, many Southern California plants are adapted to grow during this period, when water is available. When dormancy is activated, the model sets the leaf area index (LAI) to 0.75, while reducing biomass to a fraction of the pre-dormancy biomass. This can result in some odd interactions with the heat unit scheduling. For instance, if the PHU value is sufficiently high, a perennial crop may not experience a "kill" operation (conversion to biomass). Further, if the HUSC value for the kill is such that the growth cycle is completed, at which point leaf area declines, but the heat units for the kill have not been reached before dormancy is triggered, the simulated leaf area may suddenly increase from near zero to the dormancy default of 0.75.

In sum, SWAT experienced problems in simulating biomass production for native California land cover even when all parameters in the CROP database were correctly specified. In addition, cover-specific modifications of the .MGT files, for each HRU, were required to achieve what appeared to be reasonable biomass simulations. Efforts to improve the simulation by generalized cover type are summarized below.

5.5.9 Biomass Simulation for Deciduous Oaks

Modification of model behavior for deciduous oaks was tested using information for Blue Oak (*Quercus douglasii*), which is simulated with a base temperature for growth of 5°C. According to reported ecological characteristics of Blue Oak at http://www.fs.fed.us/database/feis/plants/tree/quedou/botanical_and_ecological_characteristics.html, leaf out typically occurs in early March, with acorn maturation and leaf fall in August to November. These characteristics can be reproduced with the following changes:

*.MGT file – Planting Operation (1):

HUSC (Grow): 0.10

HEATUNITS: 2250

*.MGT file – Kill Operation (8)

HUSC (Kill): 1.1

With these changes, the biomass and leaf area simulations become more realistic, with leaf development in March and decline of leafy biomass in late September to mid-October.

A remaining concern regards the leaf area simulation (which in turn determines the biomass accretion rate). For the example given above, the maximum leaf area (BLAI) was specified as 3.61, but this is never reached in the simulation. Instead, summer leaf area is limited by the accumulation of water and nutrient stress. To achieve a greater LAI prior to stress limitation it would be necessary to reduce the values of FRGRW1 and FRGRW2 in the crop database. These are the fractions of the growing season HEATUNITS corresponding to the defined points on the leaf area development curve, with FRGRW2 determining when the leaf area approaches the maximum. Lower values would cause greater LAI accumulation during the spring, wet season.

Ideally, the values of FRGRW1 and FRGRW2 should be set by comparing the time of year at which maximum leaf area is expected to the accumulated HEATUNITS at that time. Experimentation suggests that values of 0.10 and 0.16 are reasonable approximations.

5.5.10 Evergreen Species

To simulate evergreen species, it seems preferable to make use of the built in dormancy option in SWAT. For these species, the simulation can eliminate the kill operation and assign a large value of HEATUNITS that is close to the number of growing degree days for the area. Growth will continue to be limited by water, nutrient, and temperature stress, and will go dormant during the short-day period of the winter; however, LAI will not go to zero as it does with deciduous plants. Experiments with watershed 3210 showed that a value of HEATUNITS=3200 was sufficient to prevent LAI drop off from senescence.

Templates for evergreens considered the ecological characteristics of the California Live Oak (*Q. agrifolia*) and the Canyon Live Oak (*Q. chrysolepis*). The former species has active biomass accumulation from December to April, with acorns ripening in September-October. The Canyon Live Oak appears to have its active growth a little later in the season.

Assigning an artificially high value of HEATUNITS to allow leaves to be held until dormancy also requires a correction to FRGRW1 and FRGRW2 to have sufficient leaf area develop before water stress takes over.

For evergreen trees, HUSC (Grow) is not an important parameter if a kill is not simulated. This is because no new growth-initiation will be simulated after the start up year. The simulation of the first year would be improved if HUSC(Grow) is set to a nominal small value (say 0.01) in the **.mgt* files. Note that even if a kill is not simulated, the value of HEATUNITS read at the first year start of growth is retained for subsequent years of the simulation.

The general behavior shown by the model for evergreen simulations is attractive, but not an exact replication of expectations. In particular, the hardwired dormancy period may not be appropriate for California Live Oak, which has maximum biomass accumulation in December through April. However, this may be a result of the particular latitude of the test watershed , which is at 35.6° N, apparently a little further north than the range of this species. SWAT calculates dormancy based on a period when daylight hours are within a calculated distance of the annual minimum daylight hours. Between 20 and 40° latitude, this threshold (in hours) is calculated as $(\theta-20)/20$, where θ is the latitude in degrees. Thus, the length of the dormancy period decreases as we move south. It may, however, be advisable to consider modifying the dormancy assumptions in the program executable (in subroutine *dormant.f*).

Evergreen conifers can likely be simulated in the same way as outlined above for evergreen oaks – but with different growth characteristics as defined in *crop.dbf* and appropriate values of HEATUNITS.

5.5.11 REVISED BIOMASS SIMULATIONS

SWAT simulated more realistic biomass production following the above revisions – but further work will evidently be necessary. Typical results of the current simulations are shown below. LAI for evergreen land covers decreased to and remained at 0.75 through the winter (Figure 5-6, Figure 5-9, and Figure 5-10). Biomass for deciduous trees and annual herbaceous plants decreased to zero in the fall, simulating leaf fall (Figure 5-7, Figure 5-8, and Figure 5-11). Growth slowed during the summer due to water and temperature stress, and in some years, growth increased in the early fall with more rain and lower temperatures.

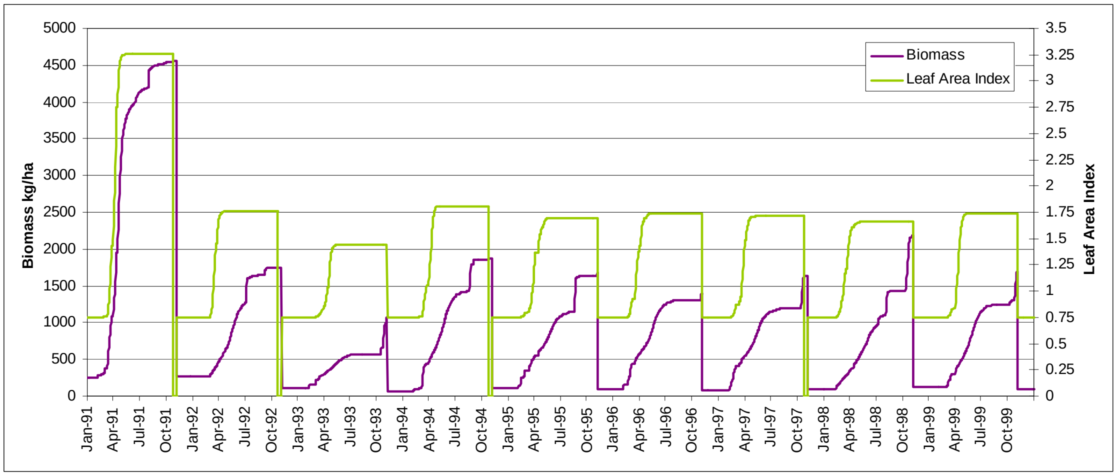


Figure 5-6 Watershed 3210: HRU 5; Evergreen, Coast Live Oak

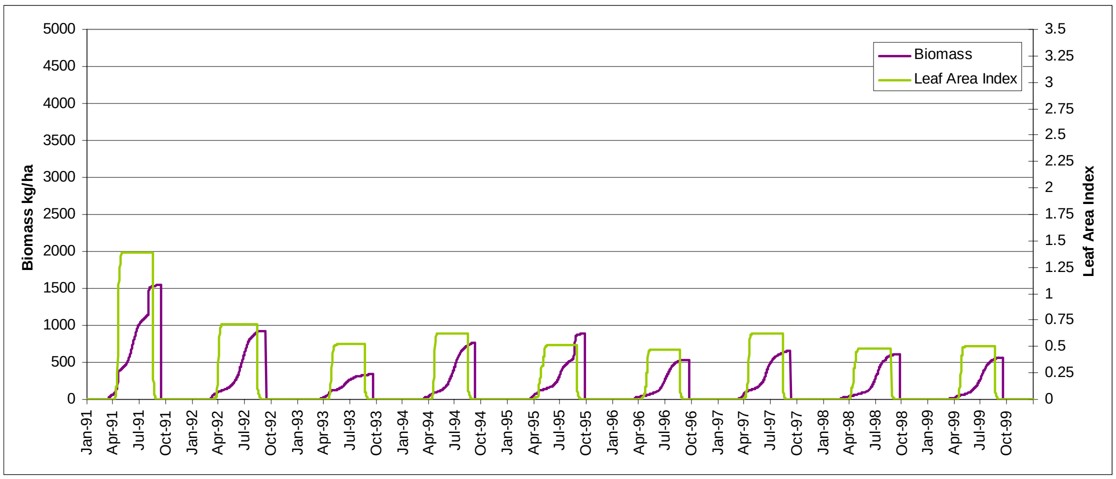


Figure 5-7 Watershed 3210: HRU11; Deciduous, Blue Oak and Annual Grassland

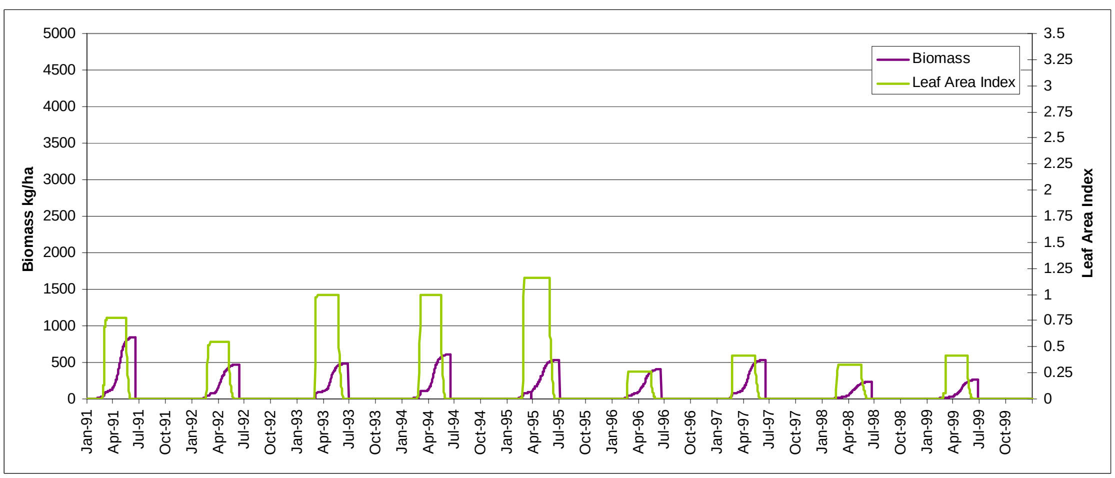


Figure 5-8 Watershed 3210: HRU16; Annual Grassland

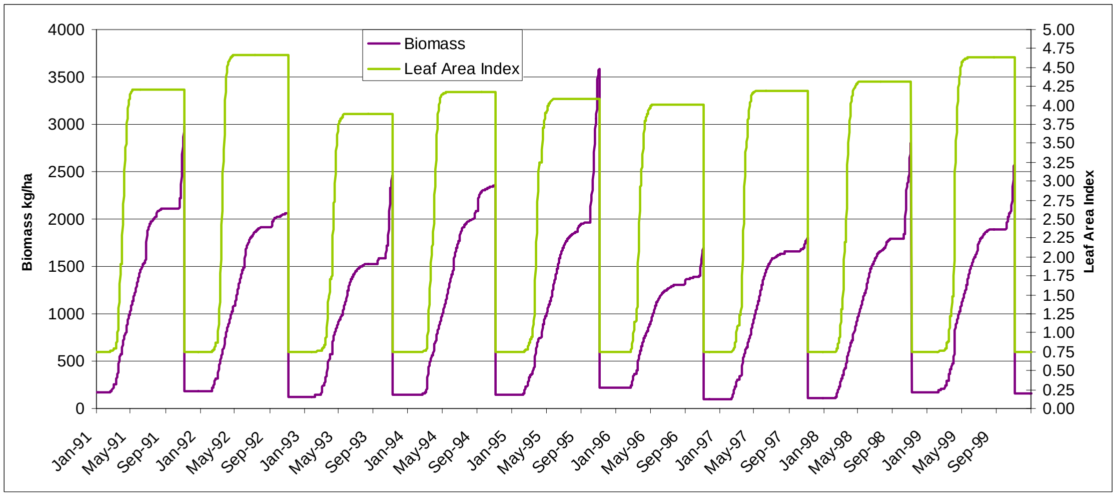


Figure 5-9 Watershed 200: HRU5; Evergreen, Coast Redwood

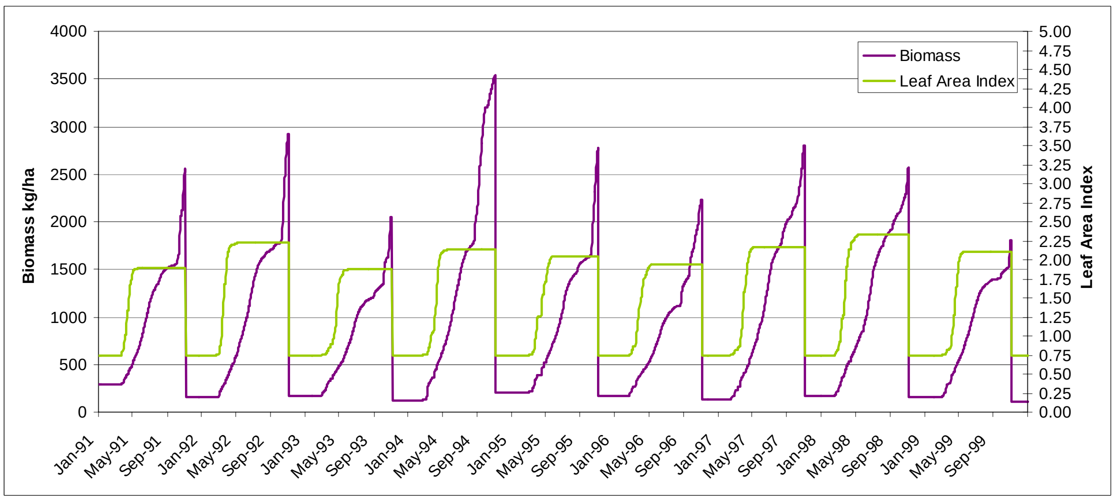


Figure 5-10 Watershed 478: HRU4; Evergreen, Conifer and Coast Live Oak

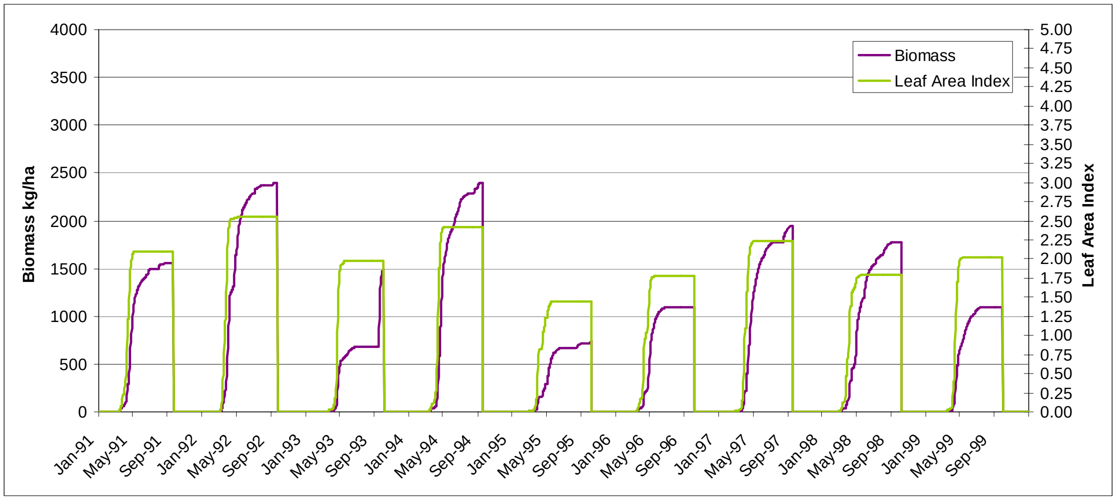


Figure 5-11 Watershed 3134: HRU18; Deciduous, Chaparral, Ceanothus

The problems with the biomass simulation reveal that biomass should be reviewed and adjusted after each new watershed model is developed. At a minimum, a simulation of leaf fall for deciduous trees and dormancy for evergreen trees should be verified for each watershed, possibly for each major HRU. To prevent absence of growth in Year 1, all watersheds could be set with land cover growing at the beginning of the simulation. Beyond these measures, further research will be needed to determine if model inputs can be selected that produce reliable biomass simulations for the major plant communities and climates of Ecoregion 6.

The performance of SWAT's plant simulation could be greatly improved by refining the growth parameters for the major plant species. Ideally, further research would obtain the following information for each major land cover and a location typical of its distribution:

- Evergreen or deciduous (for woody plants)
- Typical date of leaf out/start of growth
- Typical date for leaf area to approach maximum
- Date of conclusion of annual growth cycle (including leaf and/or seed drop)
- Base temperature for growth
- Number of heat units available above the base temperature at the corresponding location
- Information on amount of leaf area retained during dormancy (for evergreen trees/shrubs and perennial grasses)

With this information it would be possible to fine-tune the growth representation to replicate typical behavior of the species. This could be done by adjusting growth parameters so that the life cycle events occur with the correct seasonal timing and biomass production for the representative location.

5.6 REVISED SWAT VALIDATION RESULTS

As noted above, sufficient monitoring data have not been identified within Ecoregion 6 for a true validation test of model performance on unimpaired watersheds. Checks for consistency can, however, be made against other published estimates of nutrient yield and limited observational data.

To implement the validation tests, the model, modified as described in the previous sections, was run for a simulation time of 50 years using the SWAT stochastic weather generator. Note that the runs do not use observed meteorology; thus a direct comparison cannot be made to point-in-time observations.

General validation results are summarized in Table 5-8, and discussed further below.

For estimation of annual load, the SWAT results can be compared to the estimates of Smith *et al*. (1997, 2003), based primarily on correlations to land use and flow, respectively. Data are not sufficient to estimate loads directly from monitoring in any of the validation watersheds. The two Smith *et al*. methods tend to provide rather different results in Ecoregion 6, and the estimates by the land use method (Smith *et al*., 1997) are generally higher than those by the flow method (Smith *et al*., 2003).

Both of the Smith *et al*. methods contain information that allows calculation of uncertainty bounds. For the 1997 method, 90 percent confidence limits may be calculated directly. For the 2003 method, approximate confidence limits are obtained from the published information on the standard error of the model parameters. Uncertainty bounds spanning the two methods were then calculated from the minimum and maximum of these two estimates. SWAT load estimates for the validation watersheds (along with the maximum and minimum annual loads over the 50-year simulation period) are compared to the Smith *et al*. estimates in Figure 5-12 and Figure 5-13.

In every case, the SWAT median loads fall within the range estimated for the Smith *et al*. methods, and in most cases the distributions overlap well. In the watersheds with greater average runoff, the SWAT median load estimates do tend to be greater than those from the Smith *et al*. consensus, particularly for phosphorus. This could reflect the transport of organic debris in bedload that would not be captured in the NASQAN monitoring used to develop Smith *et al*.'s load estimates. Note that the model predicts that much of the total load is transported to the stream in organic form.

Table 5-8
SWAT Validation Results for Ecoregion 6

Watershed	200	455	478	3132	3133	3134	3210	3279	w8
Average Runoff (in)	9.0	12.2	19.7	7.0	20.9	23.4	11.4	3.4	5.8
Nitrogen: Loading									
SWAT Total Nitrogen Median Annual Load (kg/ha*year)	0.258	0.511	1.305	0.905	2.893	2.372	2.933	1.291	1.738
Smith *et al* 1997 Total Nitrogen Annual Load(kg/ha*year)	3.344	3.328	2.493	3.856	1.180	2.368	3.878	3.887	2.654
Smith *et al* 2003 Total Nitrogen Annual Load(kg/ha*year)	0.258	0.375	0.618	0.188	0.585	0.626	0.474	0.139	0.200
SWAT Inorganic Nitrogen Median Load (kg/ha*year)	0.216	0.438	0.540	0.221	0.877	0.697	0.394	0.184	0.382
Nitrogen: Concentration									
SWAT Total Nitrogen Flow Weighted Average Conc. (mg/L)	0.132	0.176	0.274	0.588	0.514	0.502	1.130	0.881	1.556
SWAT Total Nitrogen Median Conc. (mg/L)	0.050	0.083	0.090	0.071	0.044	0.052	0.095	0.068	0.100
Observed Average Total Nitrogen (mg/L)	NA	0.603	0.504	0.413	0.154	0.105	4.090	0.860	1.460
Observed Median Total Nitrogen (mg/L)	NA	0.250	0.250	0.441	0.151	0.075	3.351	0.408	NA
SWAT Nitrate Flow Weighted Average Conc. (mg/L)	0.102	0.145	0.108	0.138	0.164	0.149	0.134	0.225	0.302
SWAT Nitrate Median Conc. (mg/L)	0.042	0.061	0.056	0.053	0.042	0.041	0.070	0.045	0.046
Observed Average Nitrate (mg/L)	NA	0.181	0.160	0.062	0.049	0.013	3.757	0.044	NA
Observed Median Nitrate (mg/L)	NA	0.050	0.050	0.019	0.043	0.004	2.898	0.008	NA
Phosphorus: Loading									
SWAT Total Phosphorus Median Annual Load (kg/ha*year)	0.019	0.038	0.208	0.100	0.442	0.414	0.568	0.191	0.257
Smith *et al* 1997 Total Phosphorus Annual Load(kg/ha*year)	0.211	0.202	0.115	0.303	0.072	0.134	0.300	0.283	0.204
Smith *et al* 2003 Total Phosphorus Annual Load(kg/ha*year)	0.048	0.090	0.120	0.044	0.102	0.132	0.064	0.024	0.040
SWAT Inorganic Phosphorus Median Load (kg/ha*year)	0.013	0.023	0.043	0.007	0.065	0.044	0.049	0.015	0.024
Phosphorus: Concentration									
SWAT Total Phosphorus Flow Weighted Average Conc. (mg/L)	0.010	0.013	0.044	0.069	0.078	0.092	0.205	0.134	0.256
SWAT Total Phosphorus Median Conc. (mg/L)	0.016	0.014	0.013	0.017	0.017	0.018	0.016	0.018	0.028
Observed Average Total Phosphate (mg/L)	0.154	0.033	0.024	0.204	0.028	0.025	0.483	0.117	0.039
Observed Median Total Phosphate (mg/L)	0.069	0.016	0.016	0.200	0.015	0.010	0.185	0.060	NA
SWAT Mineral Phosphorus Flow Weighted Average Conc. (mg/L)	0.006	0.008	0.017	0.018	0.026	0.029	0.059	0.113	0.080
SWAT Mineral Phosphorus Median Conc. (mg/L)	0.015	0.013	0.012	0.016	0.016	0.017	0.014	0.017	0.023
Observed Average Ortho Phosphate (mg/L)	NA	NA	NA	0.185	0.009	0.007	0.322	0.039	NA
Observed Median Ortho Phosphate (mg/L)	NA	NA	NA	0.185	0.003	0.003	0.300	0.022	NA

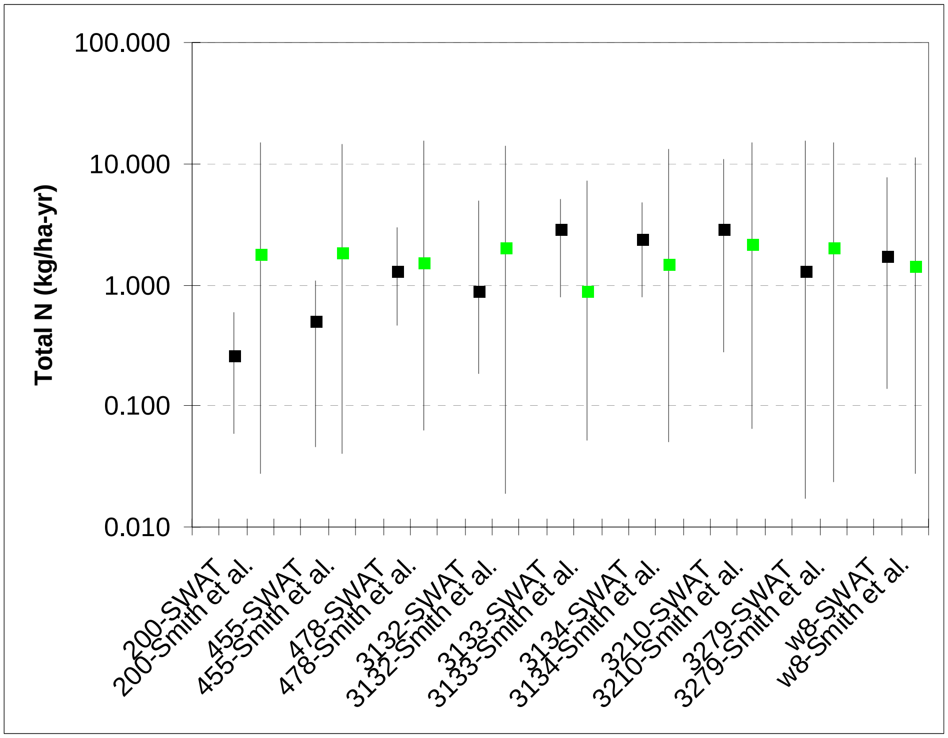


Figure 5-12 SWAT Validation Results for Total Nitrogen Load

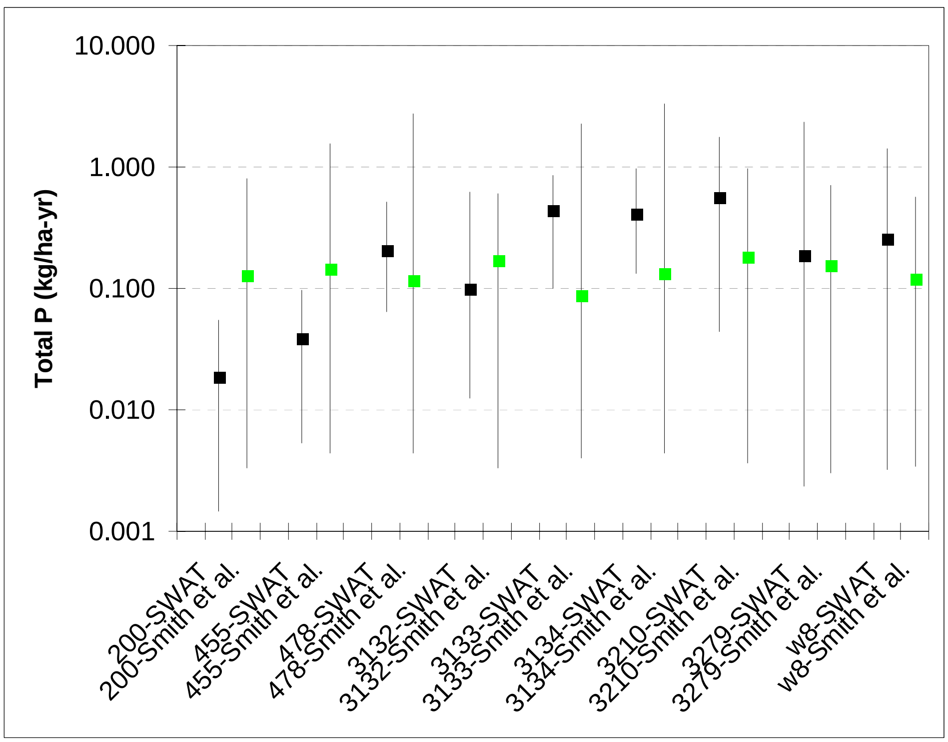


Figure 5-13 SWAT Validation Results for Total Phosphorus Load

Comparison to monitored concentration data is more problematic. First, the available sample size is quite small, raising questions as to the representativeness of the data. In addition, it is not entirely certain that the study watersheds are truly unimpacted. For example, a relatively small number of improperly functioning septic systems in a watershed could have a significant effect on nutrient concentrations.

Basic statistics for total nitrogen and total phosphorus are compared for the SWAT results and observed data in Figure 5-14 and Figure 5-15 Note that observed total nitrogen data are not available for watershed 200, and only summaries of the data were available for watershed w8. In most cases, the SWAT results are in general qualitative agreement with observations, but there are also notable exceptions. In particular, observed total nitrogen concentrations in watersheds 455 and 3210, and observed total phosphorus concentrations in watersheds 200, 3132, and 3210 appear to be considerably greater than those predicted by SWAT. In the case of watershed 3210, the elevated total nitrogen concentrations (present primarily as nitrate and consistently high during baseflow) suggest the presence of an anthropogenic impact. This could also be the case for other discrepancies; however, there may also be some watersheds in which the natural soil concentrations of phosphate is higher than the default estimates produced by SWAT.

Further insights are gained by comparing the cumulative frequency distributions for the model output and observations, as shown in Figure 5-16 and Figure 5-17. When one cumulative frequency line lies consistently to the left of the other this indicates a systematic underestimation. In general, the model cumulative frequency distribution either lies near to the observed distribution or to its left. This suggests that SWAT is performing adequately, but that some watersheds have additional nutrient sources (whether anthropogenic, atmospheric, or geologic) that have not been accounted for. In no case does SWAT consistently over-estimate the concentration distribution, which supports use of SWAT (as modified here) to provide a conservative unimpaired baseline for watersheds in Ecoregion 6.

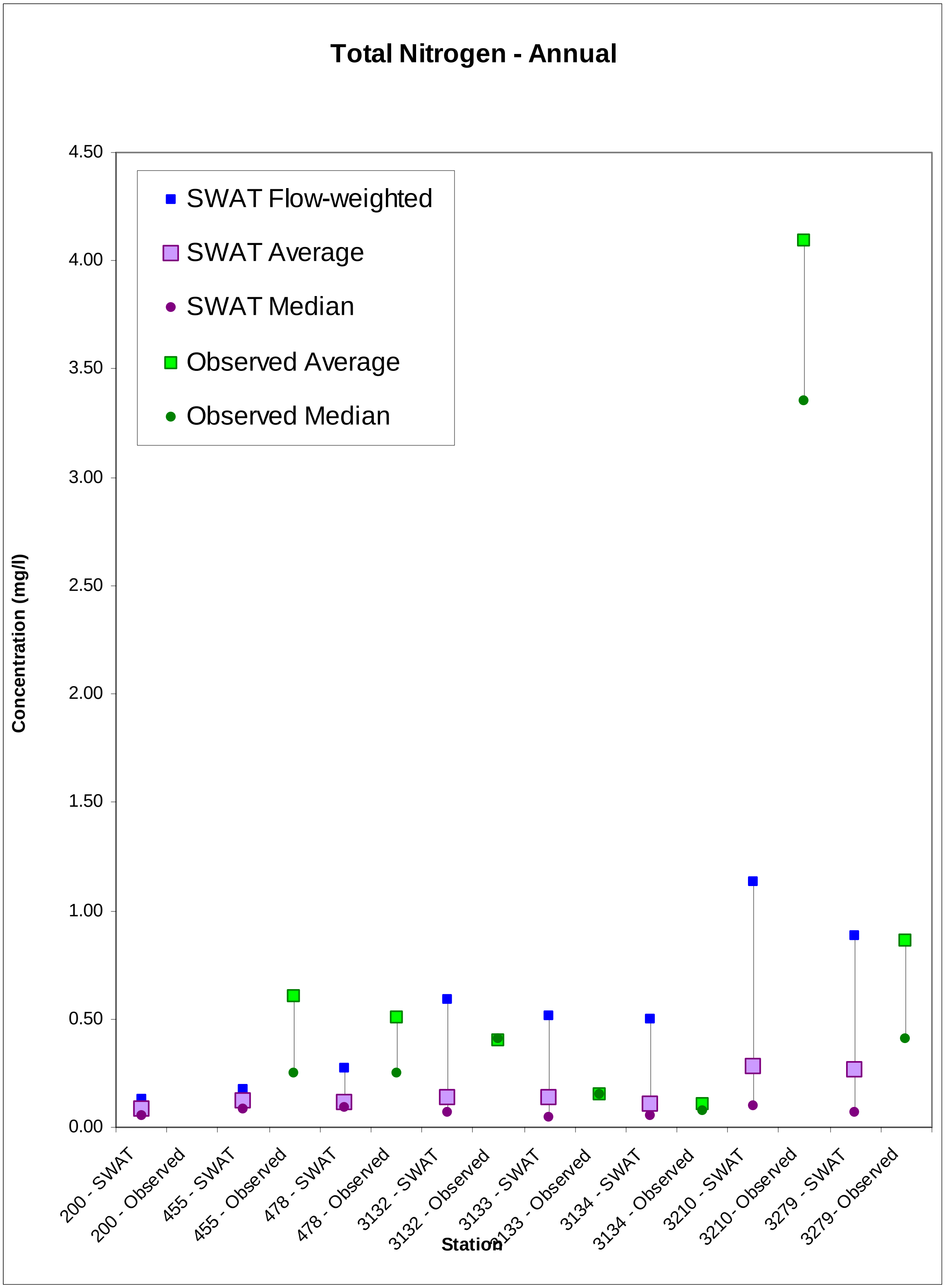


Figure 5-14 SWAT Validation, Total Nitrogen Concentration

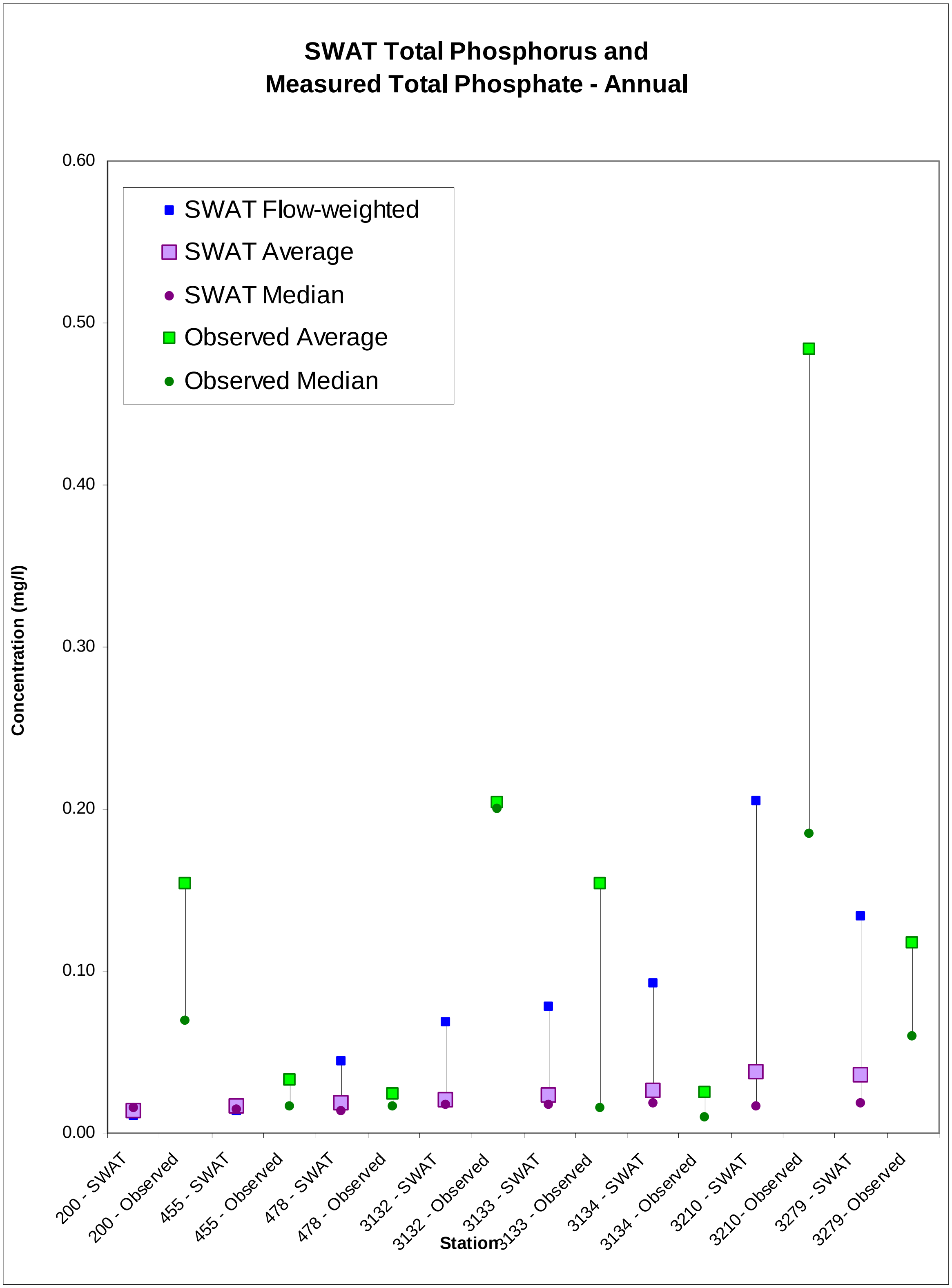


Figure 5-15 SWAT Validation, Total Phosphorus Concentration

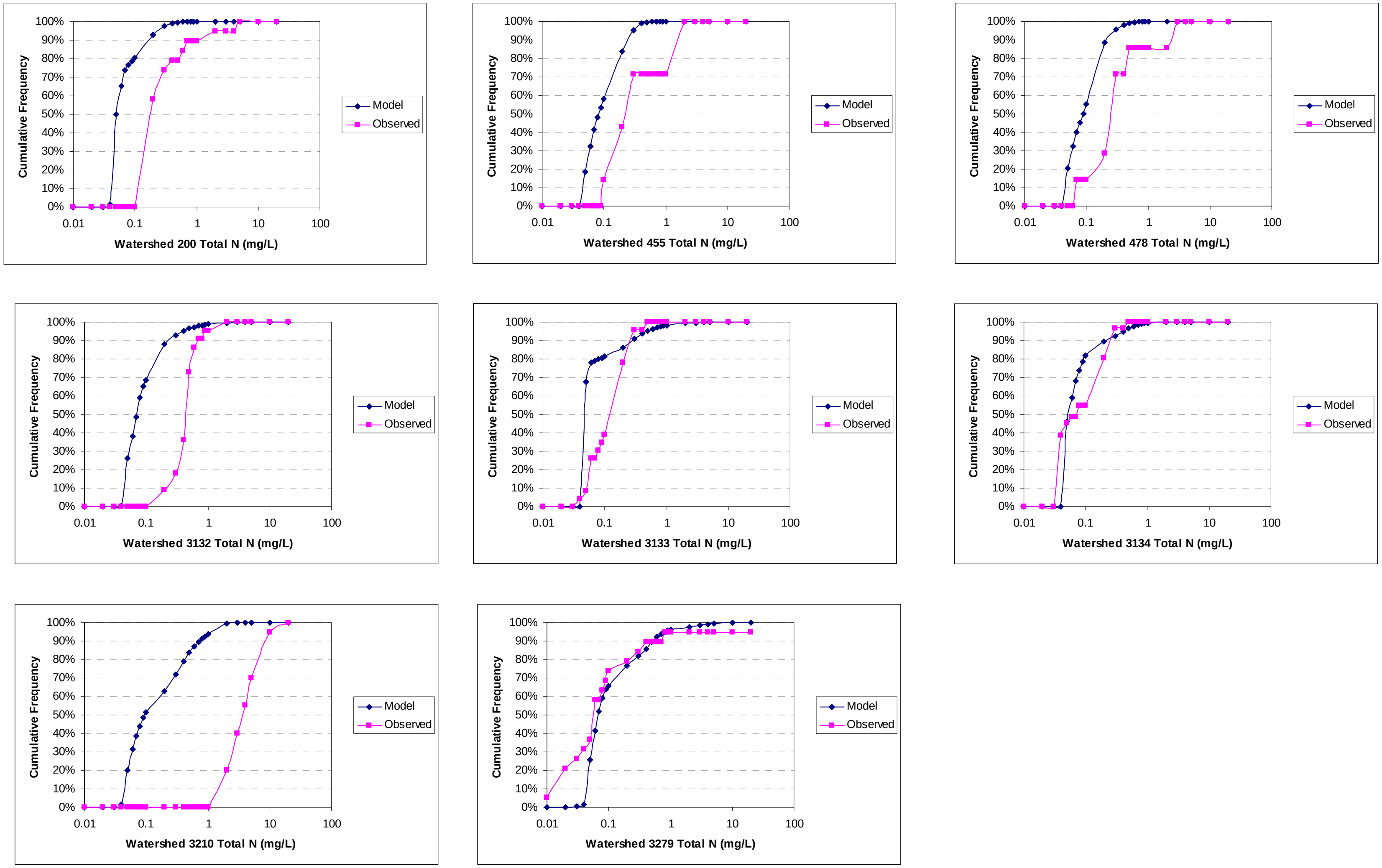


Figure 5-16 Comparison of Simulated and Observed Cumulative Frequency Distributions for Total Nitrogen

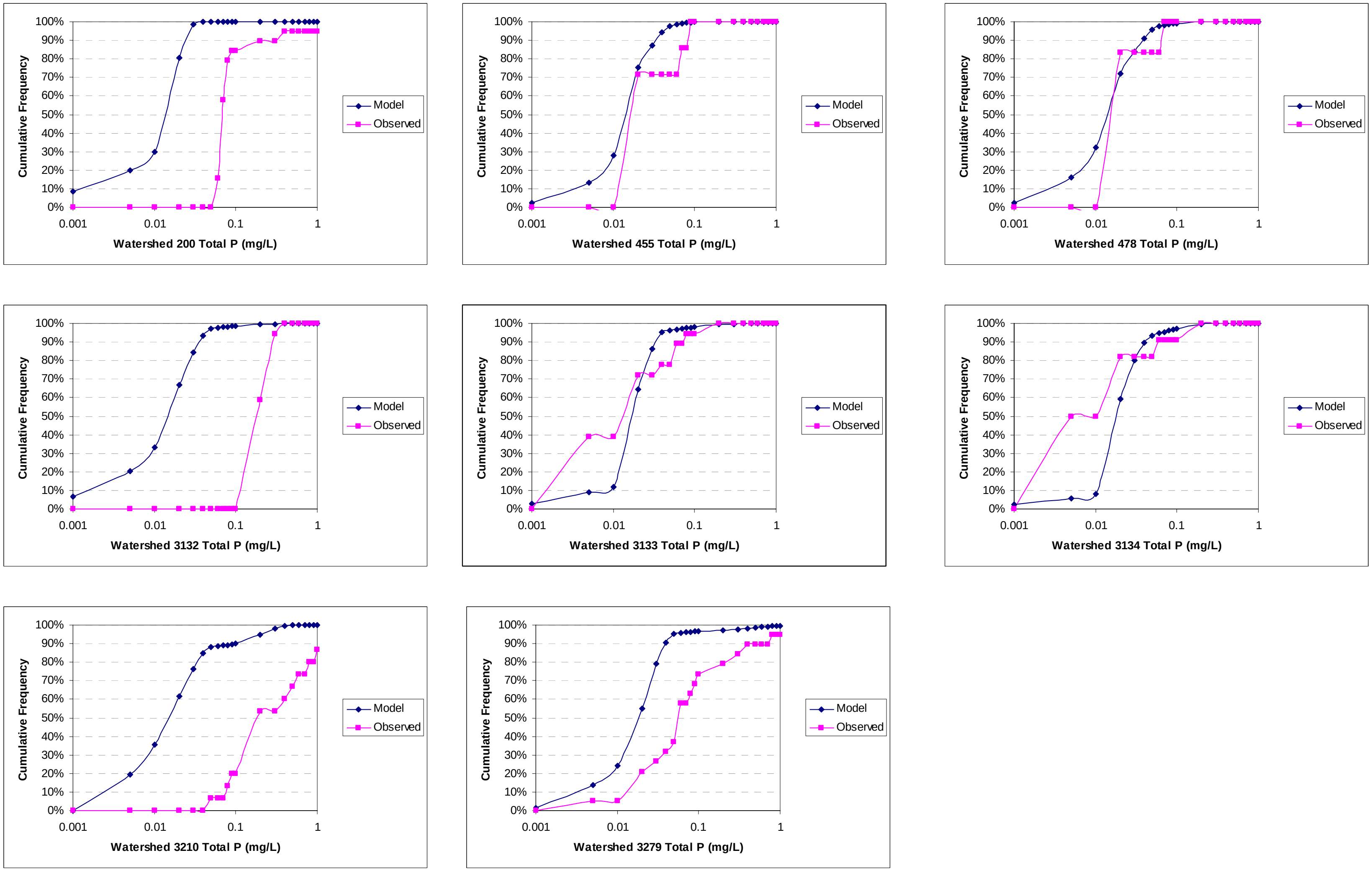


Figure 5-17 Comparison of Simulated and Observed Cumulative Frequency Distributions for Total Phosphorus

5.7 SWAT: Summary and Next Steps

Use of the SWAT model and GIS interface in default mode, as distributed, does not appear to yield reasonable results for Ecoregion 6, and likely over-estimates total nutrient loads. A series of modifications to the model, described above, bring the predictions into general agreement with both the USGS load estimates and observed concentration data. Comparison of cumulative distribution functions for concentration data indicates that the current SWAT setup may generally provide a reasonable lower bound on observed concentrations, perhaps due to the presence of some anthropogenic inputs, which is appropriate for use of the model to help establish when observations are likely consistent with natural background levels.

The extensive revisions and improvements to model set up described above means that the analysis of significant cofactors associated with nutrient load presented in the 2003 Progress Report are no longer valid. New work to evaluate appropriate stratification factors will need to be conducted during the next phase of the project. Based on results to date, it may be appropriate to undertake this analysis through a parameter perturbation approach on individual HRUs (e.g., examining response of loads to changes in precipitation, erodibility, etc.) rather than through the more laborious approach of simulating multiple individual watersheds. The end result of such an analysis could be one or more response surfaces that could be used to estimate the expected range of natural background nutrient loads and concentrations, based on appropriate exogenous factors, in Ecoregion 6 without rerunning the full SWAT model.

Finally, the experience with SWAT to date emphasizes the considerable amount of uncertainty that is present in the use of this method. Therefore, SWAT should be used as only one among several lines of evidence in developing nutrient criteria for a watershed.

5.8 SPARROW Transport Modeling

The SWAT application provides an estimate of nutrient concentrations and loads at the scale of local, low-order wastersheds. These concentrations can be expected to generally decline as flow accumulates to higher-order streams. This occurs due to a variety of trapping and removal processes during transport. As pollutant mass is removed, the exerted concentration of upstream watersheds declines, and the concentration at a downstream point represents a mixture of full-strength contributions from local watersheds and reduced concentrations from upstream watersheds. If all sub-watersheds generated the same pollutant concentration and flow response, concentration would necessarily decline with movement downstream.

5.8.1 Transport Representation

The processes that result in trapping of nutrients during transport are varied. Among them are uptake by rooted plants, sequestration in the sediment, export to the flood plain during high flow events, loss to ground water, and, for nitrogen, conversion to gaseous forms and volatilization to the atmosphere. Simulation models handle these processes explicitly or implicitly at varying levels of accuracy.

SWAT provides routines (based on QUAL2E kinetics) that describe transport through streams and cover some of the potential trapping processes. These routines, in our experience, do not provide very reliable results without site-specific calibration, and are thus of limited use for generic analysis. An alternative is to take an empirical approach. For this we utilize the stream transport component of the USGS SPARROW model (Smith *et al*., 1997). SPARROW refers to spatially referenced regressions of contaminant transport on watershed attributes, and was developed based on nationwide USGS NASQAN monitoring of 414 stations. The model empirically estimates the origin and fate of contaminants in streams, and quantifies uncertainties in these estimates based on model coefficient error and unexplained variability in the observed data.

The SPARROW tool actually contains two portions, one to generate upland loads and one to account for mass transport through stream reaches. Our approach is to use SWAT to generate the upland loads, and then apply the portion of SPARROW that estimates instream transport losses.

In SPARROW, nutrient mass reduction during transport is calculated using first order decay equations that are a function of time-of-travel:

$$C_t = C_o \cdot e^{-\delta t}$$

where:

C_o = pollutant mass present at the upstream end of a reach

C_t = pollutant mass present at the downstream end of a reach following travel time *t*

δ = decay rate (1/day)

t = time of travel (days)

5.8.2 SPARROW Transport Model Setup

The key parameters for SPARROW transport are the decay coefficient or rate of loss (δ, day^{-1}) and time of travel.

Decay coefficients are based on the national SPARROW model. The values initially developed and reported in Smith *et al*. (1997) have subsequently been updated and reported in Smith *et al*. (2003). These values are summarized in Table 5-9. It should be noted that these national values are not specifically calibrated to Ecoregion 6, and could well be biased relative to typical geochemical processes for this ecoregion. However, they should be sufficient to provide a relative representation of net transport processes.

Table 5-9
SPARROW Decay Coefficients (from Smith *et al.*, 2003)

Mean Flow Regime	Total Nitrogen	Total Phosphorus
< 1000 cfs (<28.3 m3/s)	0.455	0.258
1000 – 10000 cfs (28.3 – 283 m3/s)	0.118	0.096
10000 – 30,000 cfs (283 - 850 m3/s)	0.051	0
> 30,000 cfs (> 850 m3/s)	0.005	0

The SWAT model provides nutrient load/concentration estimates at the sub-watershed scale. SPARROW transport is applied above this scale – that is, to flow leaving the pour point of each sub-watershed. For this calculation time of travel is based on path length and mean flow velocity within the major stream network. Both estimates are assembled from data included in EPA's Reach File 1 (RF1; U.S. EPA 1996). Accuracy of these data is often low, but, again, the information is sufficient to produce relative estimates.

5.8.3 Application to 2003 Reference Watersheds

In the 2003 Progress Report, SPARROW analysis results were presented for six reference watersheds. These results were dependent on the preliminary SWAT model results, which are no longer valid. Thus, results for specific reference watersheds are omitted from this report.

5.8.4 Generalized Transport Analysis

The results for example stream systems depend on the spatial arrangement of contributing watersheds, along with travel path length and flow velocity. The change in load with stream order is thus a function of watershed shape, or the rate at which a stream gains contributing area relative to path lengths. For narrow, elongated watersheds, the rate of increase in path length is high relative to the accumulation of contributing area. This type of watershed should show a faster decline in nutrient concentration with increasing stream order than broader networks, given equivalent flow velocities.

These issues can be addressed in a more generalized context by consideration of some of the general features of catchment topology, as summarized in Eagleson (1970). First, it is well known that catchments tend to increase in length relative to area as total area increases – where catchment length is defined as the length of the mainstem from its outlet to its (projected) intersection with the upstream catchment boundary. Surveying data from throughout the world, Grey (1961) found that the correlation between catchment area and catchment length was well-described by the relationship

$$L = 1.40\ A^{0.568}$$

with a 25 percent standard error of estimation, where L is catchment length in miles and A is catchment area in square miles. This relationship can be used to estimate the typical change in length with increase in watershed area.

Another important topological concept is the bifurcation ratio, R_b, which describes the increase in number of stream segments with increasing catchment order. This is defined as

$$R_b = I_u \big/ I_{u+1}$$

where I_u is the total number of stream segments of order u. Strahler (1952) found that R_b was uncorrelated with relief, had a range from 3 to 5 in natural catchments, and was remarkably stable about an average value of 4.

If a catchment is assumed to be composed of a set of individual sub-catchment blocks in which the area of first order watersheds is a constant, A_0, then the total area present at catchment order Ω will be given by

$$A_\Omega = \sum_{i=1}^{\Omega} A_0 \cdot {R_b}^{i-1}$$

This equation may be combined with Grey's (1961) relationship for catchment length to determine the increase in length in going from order Ω-1 to order Ω.

The generic analysis first assumes that flow is accumulated at a constant depth from all contributing areas in the watershed; thus, catchment area may be converted to an average flow, Q. Analysis of loss during transport requires travel time, which depends on both path length and velocity. Velocity in open channels is in turn a function of the square root of channel slope multiplied by the hydraulic radius (the Chezy formula). In general, channel slope declines with catchment order, with the ratio of slopes at catchment order Ω to slope at order Ω-1 in the range of 0.55 to 0.57 (Eagleson, 1970). However, this is counteracted by increasing hydraulic radius. The work of Leopold and Mattock (1953) shows that velocity changes as a power function of discharge, with an exponent for river systems in semi-arid regions of about 0.1. This relationship may then be used to examine the change in velocity as a function of catchment area.

The generic analysis next assumes that nutrient load is generated in the local watersheds at some constant areal rate. For purposes of illustration, the loading rate was set at the average annual loading rate obtained from the 2003 SWAT model output for the reference watersheds (excluding watershed 9). Use of the average means that the load includes the impacts of infrequent, high-load events. This approach is appropriate for evaluating long-term loading impacts in a terminal reservoir or estuary that has a sufficiently long residence time so that summer growing season conditions reflect loading over the preceding wet season. The exerted load downstream is, however, subject to exponential decay based on travel time, using the SPARROW relationships.

Combining these assumptions, graphical relationships can be developed between catchment area and flow, delivered load, and concentration of nutrients. This is done, for example, using the average rates developed in the 2003 analysis:

First-order watershed size, A_0 (mi^2)	4
Water yield (mm/yr)	176.39
Total nitrogen yield (kg/ha/yr)	7.98
Total phosphorus yield (kg/ha/yr)	1.42
Low-order stream velocity (m/s)	0.3
Decay rates (day^{-1})	As specified by SPARROW.

The relationship of flow to area is, by assumption, linear, and can be described by

$$Q = 0.0145\,A$$

for flow in m^3/s and area in mi^2.

As a result of removal processes (which vary by flow regime), the load present at a given catchment area is not linear (Figure 5-19 and Figure 5-20). It is, however, nearly linear on a log-log plot. With the rates specified above, the annual loading rates for nitrogen and phosphorus can be approximated as a function of area (for natural cover within ecoregion 6) as follows:

$$Total\ \ Nitrogen\ (kg / yr) = 5119.2\ A^{0.6526}$$

$$Total\ \ Phosphorus\ (kg / yr) = 558.11\ A^{0.8247}$$

where area (A) is again expressed in square miles. The coefficients in these equations are expected to change with the modifications to the SWAT modeling documented in Section 5.6.

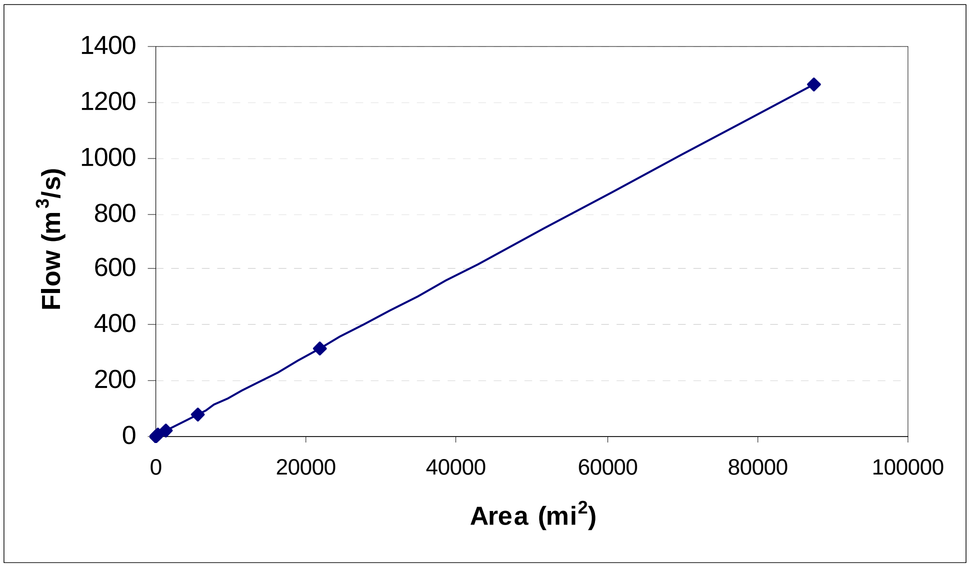


Figure 5-18 Estimated Theoretical Relationship of Average Annual Flow to Catchment Area for Ecoregion 6

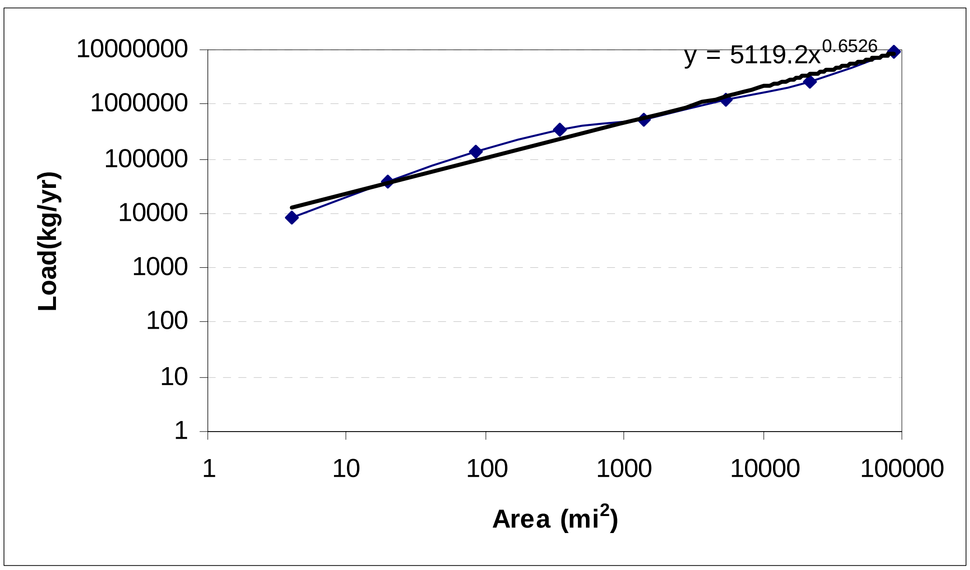


Figure 5-19 Estimated Theoretical Relationship of Annual Nitrogen Load to Catchment Area for Ecoregion 6

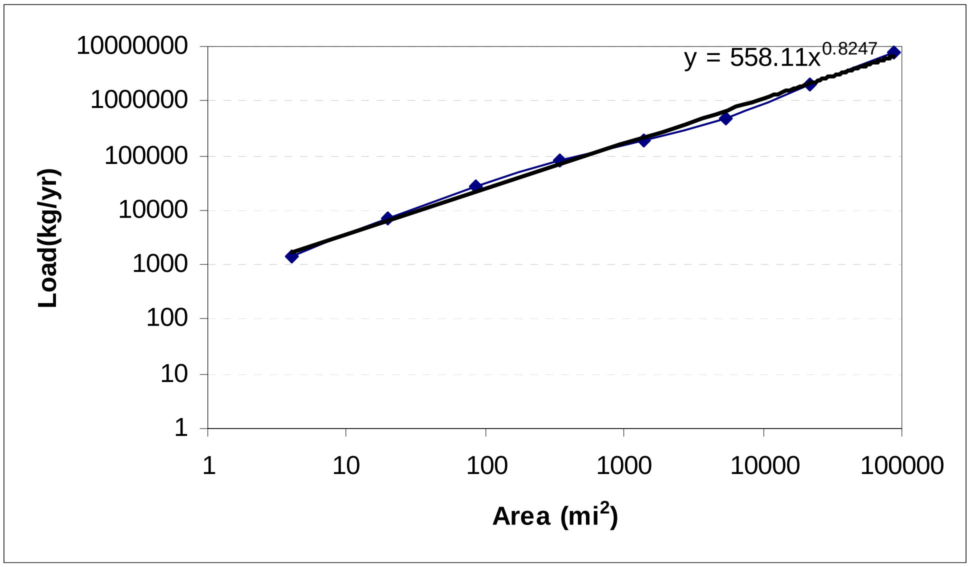


Figure 5-20 Estimated Theoretical Relationship of Annual Phosphorus Load to Catchment Area for Ecoregion 6

The average flow-weighted concentration is the load divided by the flow. Predicted concentrations as a function of area are shown in Figure 5-21 and Figure 5-22. The relationships are not linear, and exhibit kinks that are due to transitions among different SPARROW loss regimes. An approximate fit to the predicted concentration may, however, be obtained through a power function representation (shown as the magenta line on the plots):

$$Total\ Nitrogen\ (mg/L) = 11.19\ A^{-0.3474}$$

$$Total\ Phosphorus\ (mg/L) = 1.22\ A^{-0.1753}$$

Once again, these coefficients are expected to change as a result of revisions in SWAT modeling. Note that the phosphorus concentration converges to a constant value at large catchment size because the SPARROW loss coefficients are zero at flows greater than 283 m^3/s. Nitrogen concentrations are predicted to continue to decline with increased watershed area for conditions of constant unit areal loading of nitrogen load and flow.

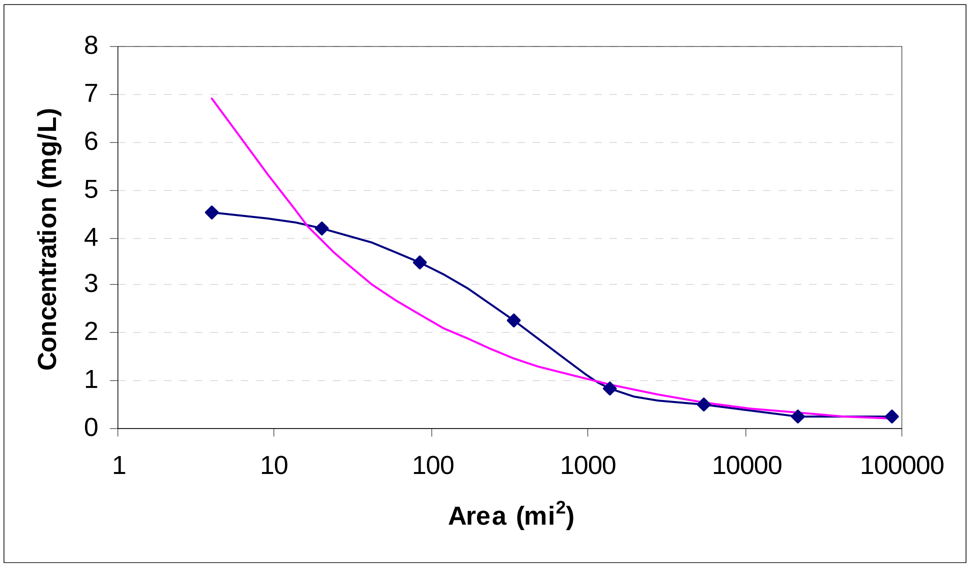


Figure 5-21 Estimated Theoretical Relationship of Flow-Weighted Average Annual Total Nitrogen Concentration to Catchment Area for Ecoregion 6

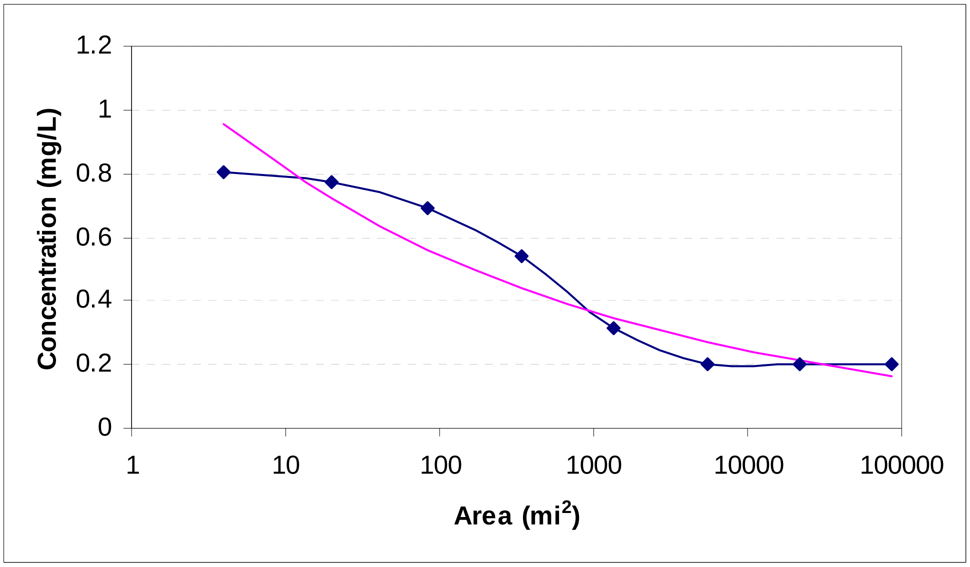


Figure 5-22 Estimated Theoretical Relationship of Flow-Weighted Average Annual Total Phosphorus Concentration to Catchment Area for Ecoregion 6

5.9 BATHTUB MODEL-RECEIVING WATER ENDPOINT ANALYSIS

The objective of the BATHTUB model application is to establish acceptable nutrient loading into lakes and reservoirs by estimating algal response as a function of hydraulic residence time and other key variables. Effects of variations in other parameters can then be analyzed as secondary variables in a sensitivity analysis. The first objective is to establish a three-dimensional allowable loading response surface in which the boundary of predicted acceptable and unacceptable conditions is plotted as a function of residence time, nitrogen load, and phosphorus load. Acceptable and unacceptable conditions will be defined based upon whether the receiving waters exceed certain threshold criteria for chlorophyll-*a* defined as a function of the end use designation for the receiving water body.

5.9.1 MODEL DESCRIPTION

The Army Corps of Engineers' BATHTUB model (Walker, 1996) was used to analyze the water quality response in a typical Ecoregion 6 watershed lake to different nutrient loading scenarios. BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll-*a* concentrations (or algal densities), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a steady-state mass balance model approach that estimates the distribution of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day or seasonal variations in water quality. Algal concentrations are predicted for the summer growing season when water quality problems are most severe. Annual differences in water quality, or differences resulting from different loading or hydrologic conditions (e.g., wet vs. dry years), can be evaluated by running the model separately for each scenario.

BATHTUB first calculates steady-state phosphorus and nitrogen balances based on nutrient loads, nutrient sedimentation, and transport processes (lake flushing, transport between segments). Several options are provided to allow first-order, second-order, and other loss rate formulations for nutrient sedimentation that have been proposed from various nutrient loading models in the literature. The resulting nutrient levels are then used in a series of empirical relationships to calculate chlorophyll-*a*, oxygen depletion, and turbidity. Phytoplankton concentrations are estimated from mechanistically based steady-state relationships that include processes such as photosynthesis, settling, respiration, grazing mortality, and flushing. Both nitrogen and phosphorus can be considered as limiting nutrients, at the option of the user. Several options are also provided to account for variations in nutrient availability for phytoplankton growth based on the nutrient speciation in the inflows. The empirical relationships used in BATHTUB were derived from field data from many different lakes, including those in EPA's National Eutrophication Survey and lakes operated by the Army Corps of Engineers. Default values are provided for most of the model parameters based on extensive statistical analyses of these data.

Spatial variability in water quality can be simulated with BATHTUB by dividing the lake horizontally into segments, and calculating transport processes such as advection and dispersion between the segments. This is appropriate for large lakes, particularly lakes with multiple sidearms and tributary inflows, that have substantially different water quality in different portions of the lake. However, this was not necessary for the Ecoregion 6 lakes due to their generally small to moderate sizes, and the lack of detailed data demonstrating significant spatial variations in Ecoregion 6 lake characteristics and water quality. Therefore, BATHTUB was applied as a whole lake model to these lakes.

5.9.2 PRIOR ECOREGION 6 BATHTUB MODEL APPLICATIONS

Four urban lakes (Lindero, Westlake, Sherwood, and Malibu) within the Malibu watershed in Ecoregion 6 were simulated using the BATHTUB model as part of a TMDL investigation for Malibu Creek (Tetra Tech, 2002). The TMDL investigation identified the amounts of nitrogen and phosphorus that can be discharged to the water bodies in the Malibu Creek watershed without causing violations of applicable water quality standards, and allocated allowable nutrient loads among different discharges.

The BATHTUB model was used to develop the linkage between loadings to the four Malibu Creek lakes, and resulting lake nutrient concentrations and lake algal biomass. The Malibu Lake models were calibrated using BATHTUB default parameters and site specific flow estimates, with total load estimates and tributary inflow concentrations determined during calibration. External loads and tributary concentrations were adjusted until the predicted nutrient concentrations in the lakes matched the observed data. The model predictions represent average algal concentrations during the growing season based on the observed nutrient levels, flushing rates, and lake geometry, as estimated from default parameters derived from many other lakes during the development of BATHTUB.

The BATHTUB model was calibrated to these four Malibu Creek Watershed Lakes using the second order available phosphorus model (Phosphorus model 1) for phosphorus, the second order available nitrogen model (Nitrogen model 1) for nitrogen, and the phosphorus, nitrogen, light, and flushing model (mean chlorophyll-*a* model 1) for chlorophyll-*a*. These BATHTUB model applications used a vertical (Z) mixing layer of 2 to 2.3 meters and had average lake depths in the range of 2 to 4.9 meters. Residence times for these lakes ranged from 0.015 to 0.34 years. The non-algal turbidity parameter in the four lakes varied from 0.6 to 2.1 m^{-1}.

A comparison of the BATHTUB model results with observed data for these four Malibu Creek Watershed lakes is given for lake TP, TN, and chlorophyll-*a* concentrations in Figures 5-23, 5-24, and 5-25, respectively. An excellent fit was obtained for nutrient concentrations. The predicted chlorophyll-*a* concentrations are within the same general range as the measured concentrations in all four lakes (about 20-45 ug/l chlorophyll-*a*), but trends were not well captured, with the higher measured chlorophyll-*a* concentrations observed in the lakes with the lower measured nitrogen and phosphorus concentrations (Lindero, Malibu). The apparently poor fit to observed chlorophyll-*a* concentration may reflect uncertainty in the estimation of the true growing season concentration as well as control by other factors, such as light availability and mixing patterns.

Comparison of the lake monitoring data and model results with the available stream monitoring data at various locations in the watershed demonstrates that these lakes behave as nutrient sinks under current loading conditions. Nitrate and phosphate concentrations in the lakes and in the stream reaches immediately downstream of the lakes are typically lower than in the upstream tributaries feeding the lakes. Algal growth in the lakes removes nutrients from the water and deposits them in the sediments as the algae settle. Organic sediment decomposition releases some of the nutrients back to the water, and could become a significant source if existing loads from the watershed are greatly reduced in the future. Nutrient removal in the lake waters is generally highest during the summer growing season.

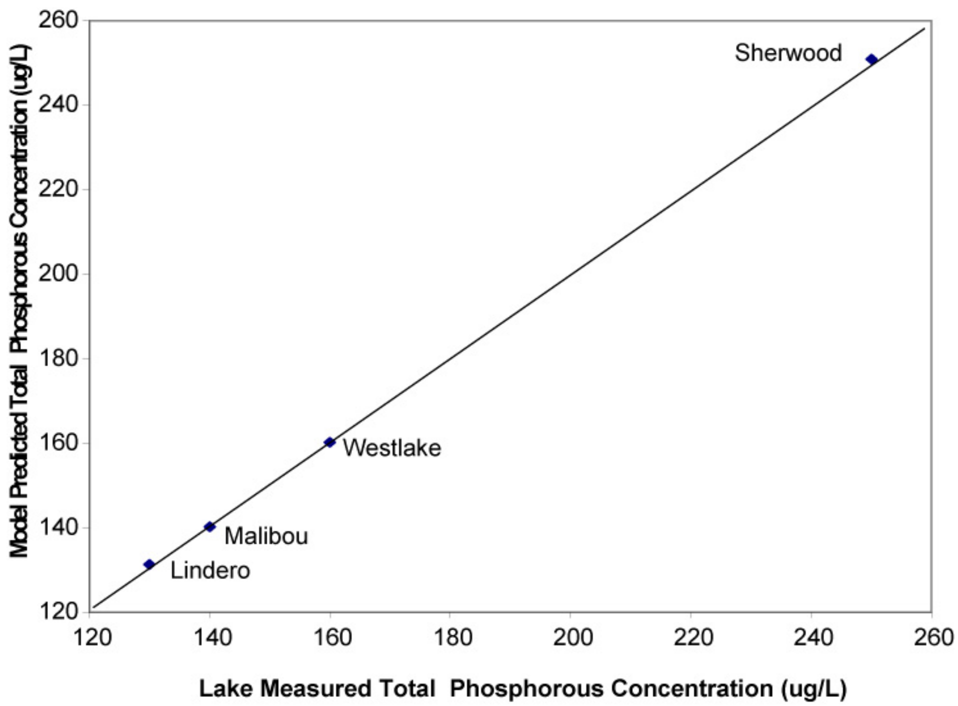


Figure 5-23 Comparison of BATHTUB Model and Measured Total Phosphorus Concentrations, Malibu Creek Lakes TMDL

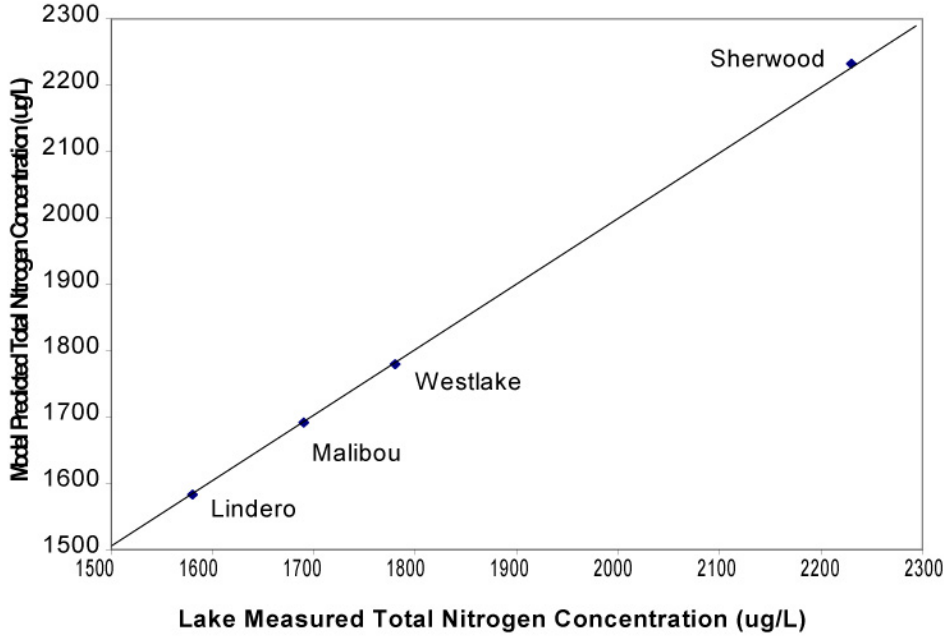


Figure 5-24 Comparison of BATHTUB Model and Measured Total Nitrogen Concentrations, Malibu Creek Lakes TMDL.

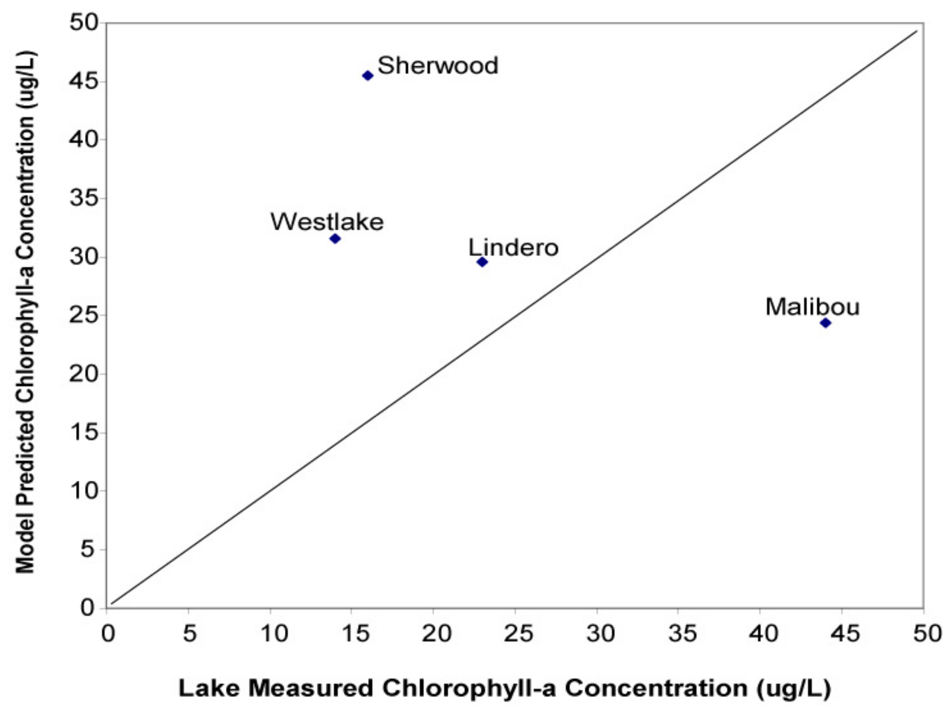


Figure 5-25 Comparison of BATHTUB Model and Measured Chlorophyll-a Concentrations, Malibu Creek Lakes

5.9.3 LINKAGE WITH SWAT/SPARROW MODEL RESULTS

For a generic application of the BATHTUB Model to a typical Ecoregion 6 lake or reservoir, it is desirable to normalize both the SWAT/SPARROW and the BATHTUB Model to the watershed area or lake size The SWAT/SPARROW Model generates output loadings in terms of nutrient mass per unit time in the inflow discharged to potential receiving water bodies such as lakes or reservoirs. In contrast, the BATHTUB Model equations use nutrient concentrations in the lake influent as input to calculate lake nutrient and algal concentrations. Thus, the SWAT/SPARROW Model output, which is in units of mass per unit time, requires some modification for use in the BATHTUB Model input, which is in units of mass per unit volume. The SWAT/SPARROW Model output and BATHTUB Model input then requires some adjustment to develop a common normalizing criteria based upon the watershed area or lake size.

Although the BATHTUB Model equations are based upon nutrient concentrations in the lake influent rather than mass flux into the lake, the BATHTUB Model lake influent nutrient concentrations can also be viewed as mass per time normalized by inflow rate since this results in final units of concentration.

Setting the following definitions:

Input loadings in BATHTUB Model (C) are mass per time normalized by inflow rate as follows:

C = Mass Inflow Rate, R (mg/year) /Water Inflow Rate, W (m^3 /year)

C = Mass (mg) /Inflow Water Volume (m^3)

Output loadings from SWAT/SPARROW (R) are mass per time based as follows:

R = Mass (mg) / Time (year)

Residence Time in lakes (T) is defined as follows:

T (Year) = Lake volume, V (m^3) / Water Inflow, W (m^3/year)

The influent concentration loadings in the BATHTUB Model can be related to SWAT/SPARROW mass per time loadings using influent concentration loadings, mass flux loadings, lake volume, and lake residence times. The loadings from SWAT/SPARROW (R) are normalized by the lake volume (V), resulting in mass loadings per unit lake volume per unit time (mg / m^3 -year). The SWAT SPARROW normalized loadings (R/V) then correlate with the ratio of the BATHTUB Model concentration based loadings divided by the lake residence time as indicated below:

C / T = (R/W) /(V/W) = R/V mg/year - m^3 lake volume

Thus, lake algal concentrations are expressed as functions of the nutrient mass flux into the lake divided by the lake volume, which is identically equal to the concentration of nutrients in the lake inflow water divided by the water residence time in the lake. The model results presented in the following section are therefore developed as functions of this normalized variable.

Initial iterations of the SWAT/SPARROW Model output for mass flux loadings (R) were interpreted as functions of watershed drainage area (with slightly different coefficients from the final model given in Section 5.8.4) to set the approximate range for the analysis:

N load (Kg/Yr) = $4408.4\ x^{0.6526}$

P load (Kg/Yr) = $378.88\ x^{0.8247}$

where "x" refers to the watershed drainage area in square miles. A plot of lake volume versus watershed drainage area in square miles from the Ecoregion 6 database (Figure 5-26) illustrates that lake volume can be related to drainage area as follows:

Lake Volume/v (Acre-feet) = $619.43\ x^{0.6751}$

Thus, dividing the N and P loading equations by this lake volume correlation results in the following N and P loadings normalized by lake volume:

N/v (Kg/Acre-feet-Year) = $7.117\ x^{-0.0225}$

P/v (Kg/Acre-feet-Year) = $0.6117\ x^{0.1496}$

Note: gm/cu meter -Year = 0.8107 Kg/Acre-feet-Year

The above relations (given in Figure 5-27) for N and P loadings normalized to lake volume show very little variation with watershed drainage area, which was expected based upon the normalization process. The normalized nitrogen loads range from 4.45 to 5.56 gm/cu meter-year and average 5.01 gm/cu meter-year over watershed drainage areas from 10 to 100,000 square miles. The normalized phosphorus loads range from 0.63 to 2.78 gm/cu meter-year and average 1.44 gm/cu meter-yea over watershed drainage areas from 10 to 100,000 square miles.

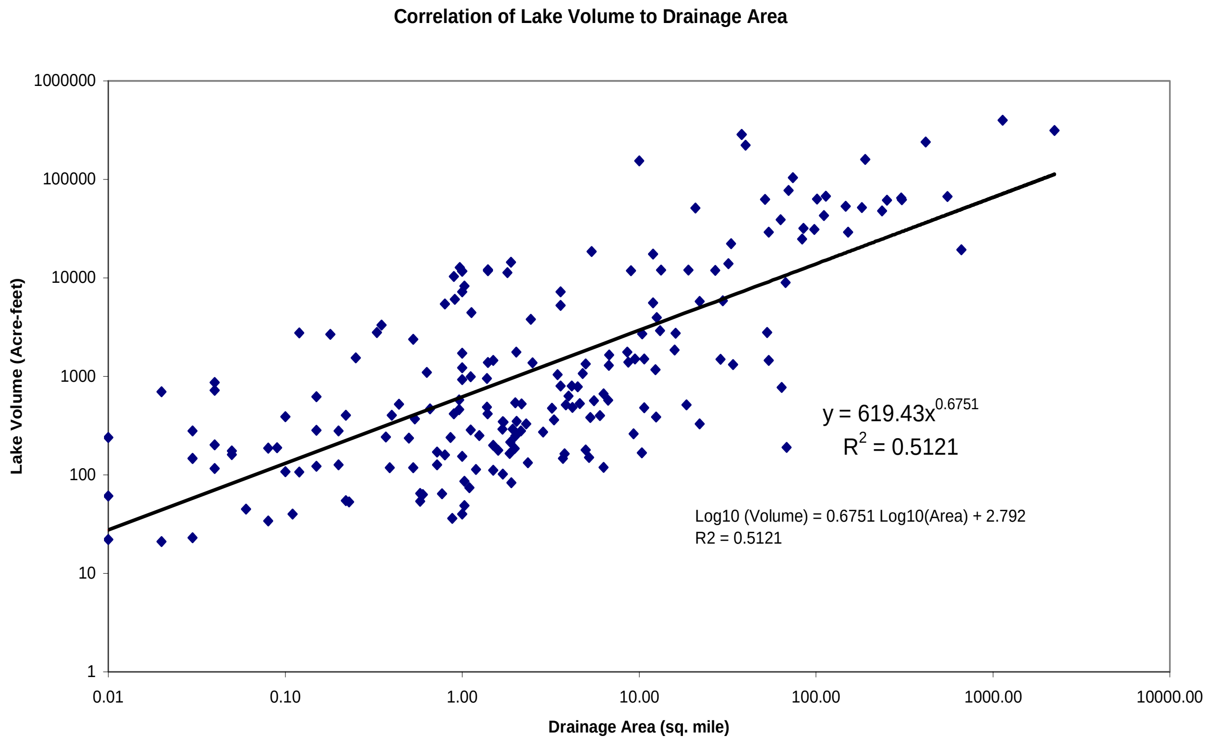


Figure 5-26 Correlation between lake or reservoir volume and watershed drainage area for Ecoregion 6.

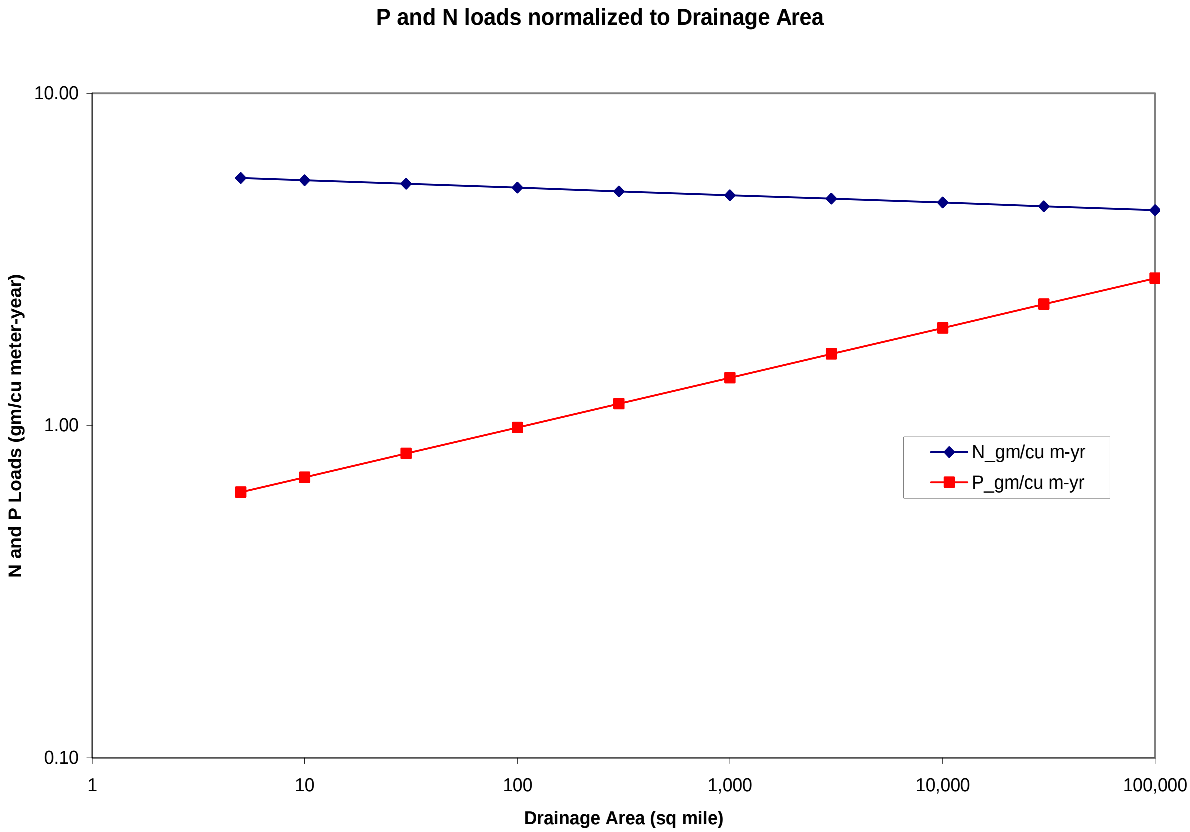


Figure 5-27 Figure 4-27. Correlation between normalized Total Nitrogen and Total Phosphorus loads and watershed drainage area for Ecoregion 6.

5.9.4 Model Parameter Selection for the Ecoregion 6 Evaluation

Chlorophyll-*a* targets associated with specific waterbody uses for Ecoregion 6 lakes and reservoirs have not yet been finalized. Initial targets for the analysis were defined by reviewing the database for lakes in Ecoregion 6 and the results of the recent TMDL for the four lakes in the Malibu Creek Watershed. Figure 28 shows a plot of the Cumulative Distribution Frequency (CDF) for Chlorophyll-*a* data in Ecoregion 6. The 80th percentile of the data is approximately 10 ug/L. The Malibu Lakes TMDL set 10 ug/L as the target concentrations for the four lakes based upon a 30 percent algae cover. For this analysis, 10 ug/L, 25 ug/L, and 40 ug/L were chosen as chlorophyll-*a* targets, which relate to different potential designated end uses for the lake.

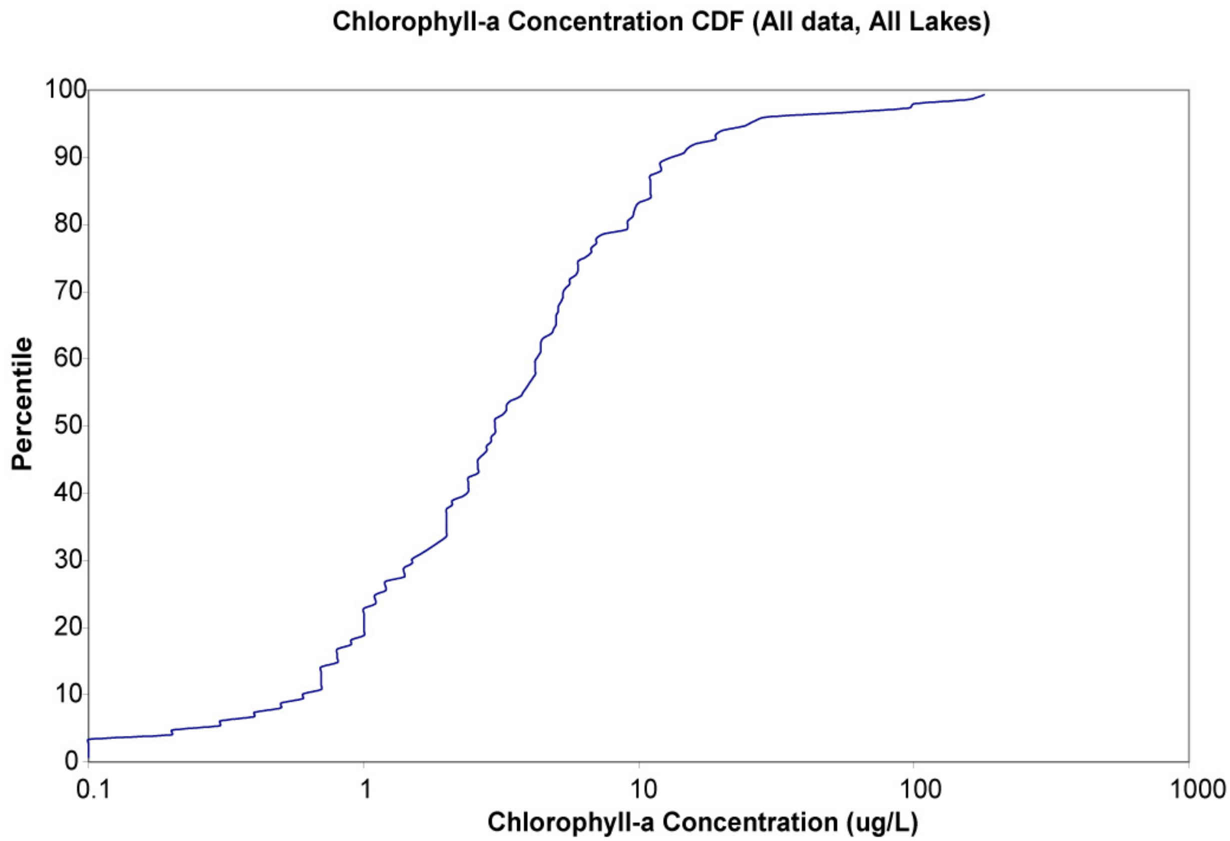


Figure 5-28 Lake Chlorophyll-a Concentration in Ecoregion 6 Lakes database.

The applications of the BATHTUB model to the four lakes in the Malibu Creek Watershed were used along with Ecoregion 6 specific data to define BATHTUB Model parameters. The Ecoregion 6 database was used to create a database of the lakes in the target area Ecoregion 6 with information on residence time, phosphorus concentration, nitrogen concentration, chlorophyll-*a* concentration, and Secchi depth as given in Figures 5-29 through 5-34. Hydraulic data were only available for lakes with a dam. Residence times for these reservoirs were defined using the reservoir storage capacity divided by the catchment flowrate (Figure 5-35), where the catchment flowrate (Q, m^3/s) is estimated based upon the drainage area (A, square miles) using the relation given in SWAT SPARROW analyses (Q=0.0145A). The median residence time was 0.51 years. The mean reservoir depth was defined by the average reservoir storage divided by the surface area (Figure 5-36). The median depth of all reservoirs was 5.7 meters.

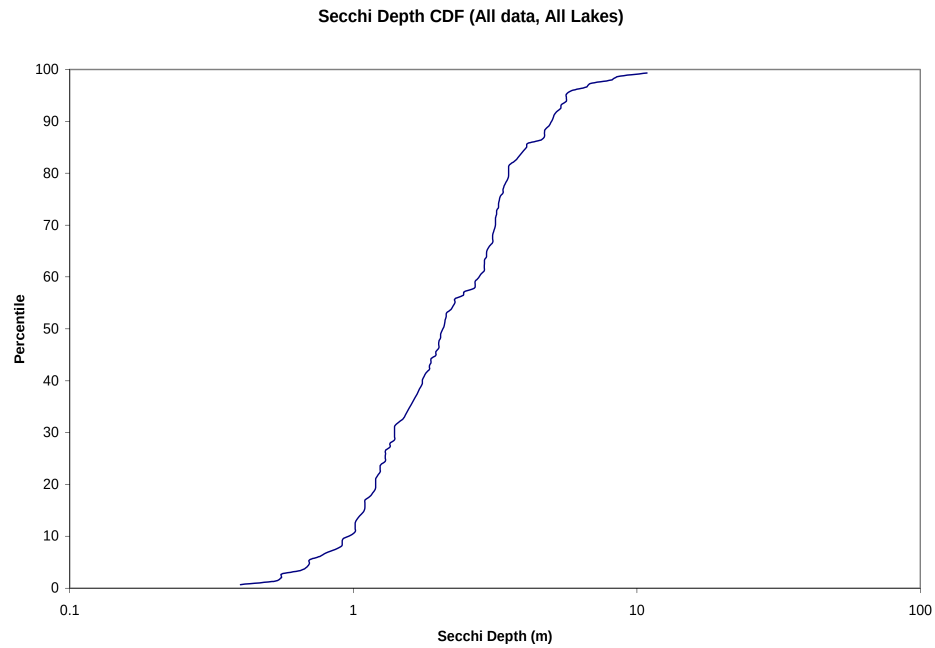


Figure 5-29 Secchi Depth measurements for lakes in Ecoregion 6.

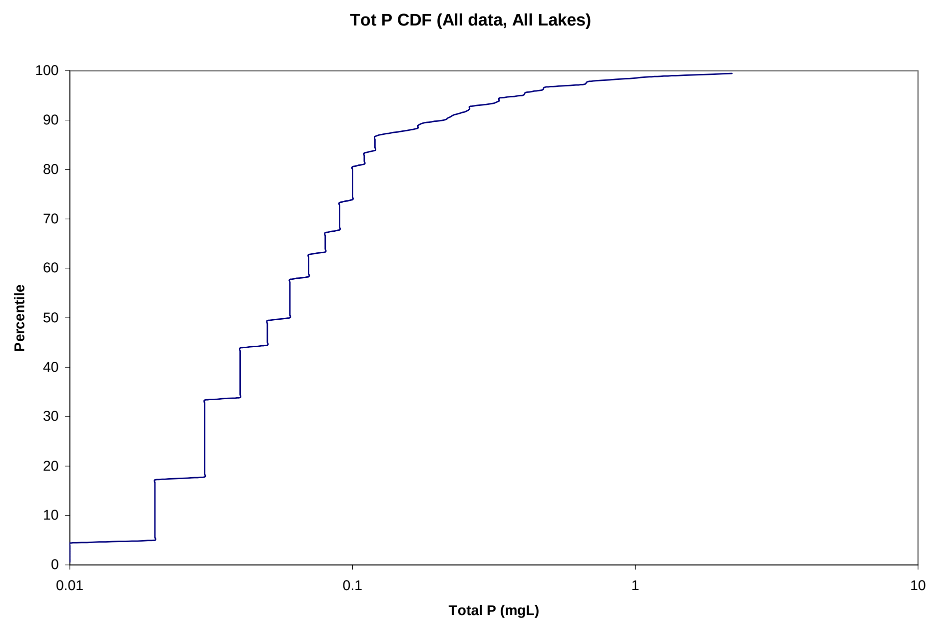


Figure 5-30 Total Phosphorus concentration measurements for lakes in Ecoregion 6.

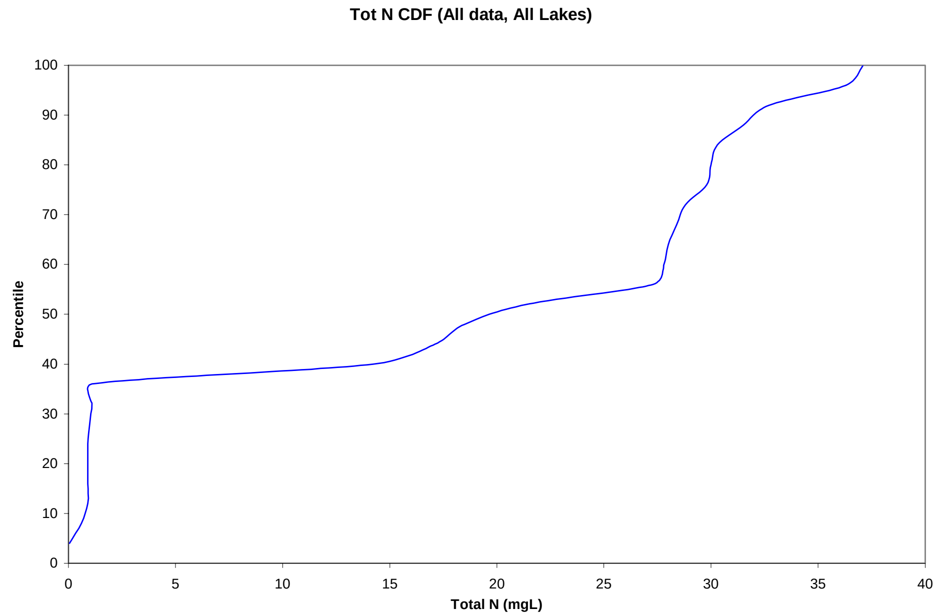


Figure 5-31 Total Nitrogen concentration measurements for lakes in Ecoregion 6.

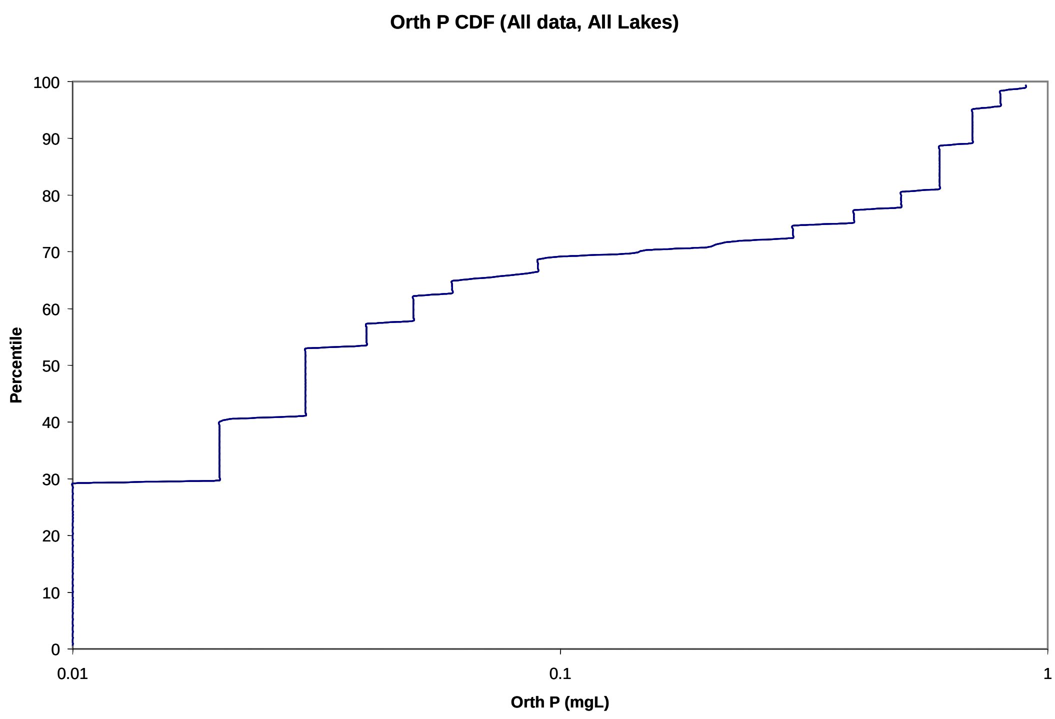


Figure 5-32 Orthophosphate (P) concentration measurements for lakes in Ecoregion 6.

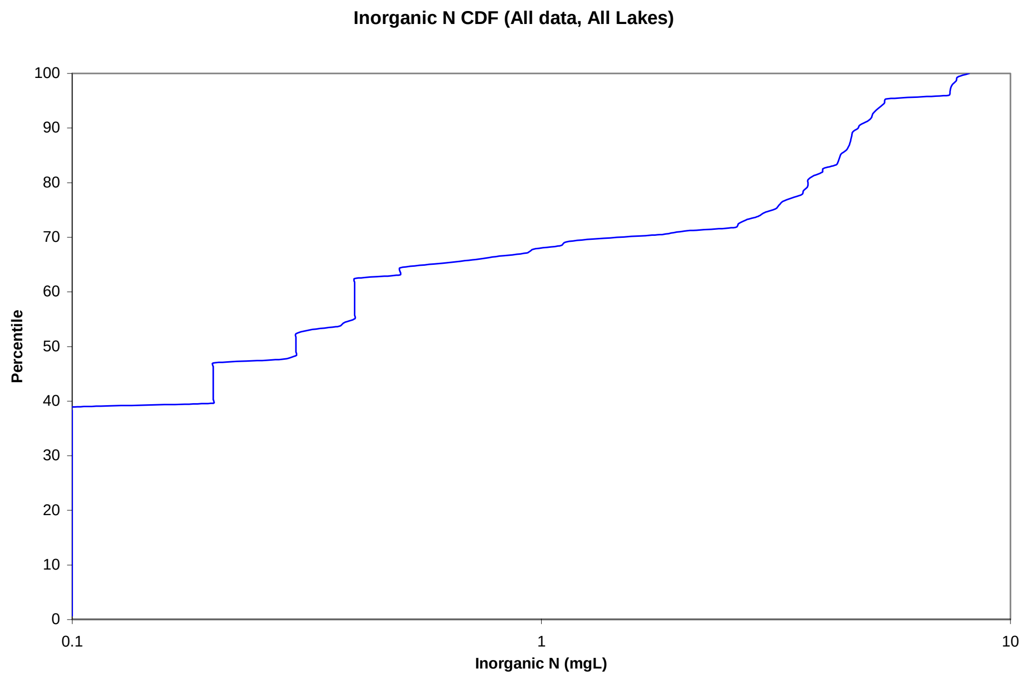


Figure 5-33 Inorganic Nitrogen concentration measurements for lakes in Ecoregion 6.

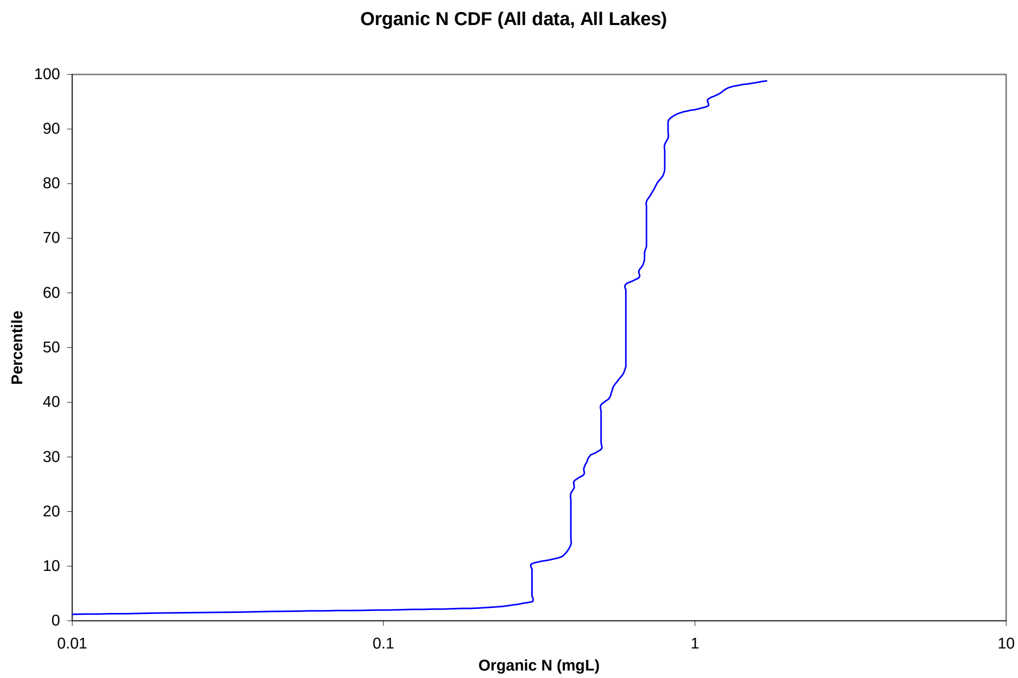


Figure 5-34 Organic Nitrogen concentration measurements for lakes in Ecoregion 6.

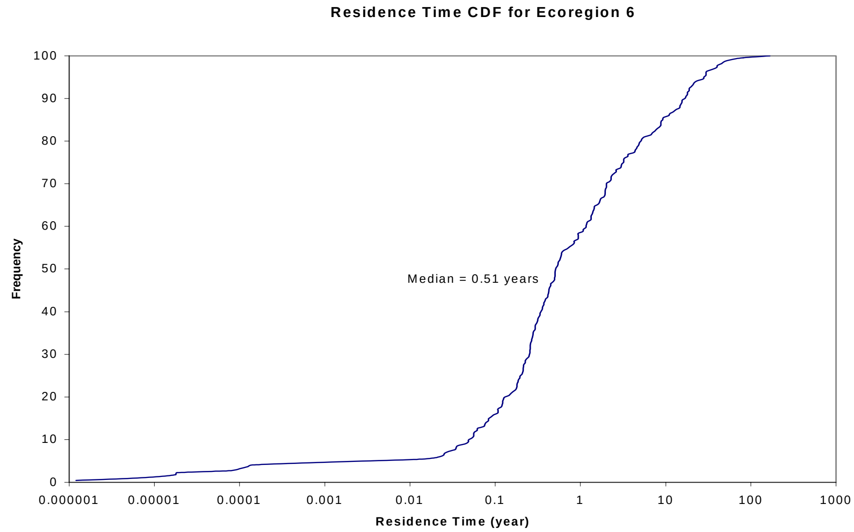


Figure 5-35 Residence time (years) for lakes (with a dam) in Ecoregion 6.

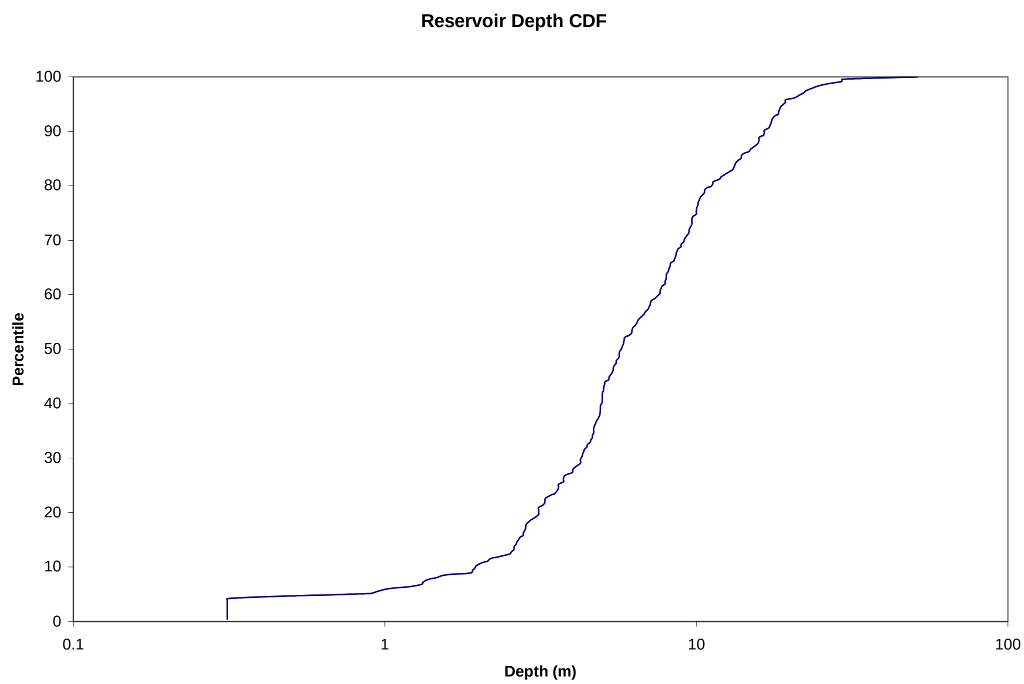


Figure 5-36 Reservoir depth (meters) for dammed lakes in Ecoregion 6.

The non-algal turbidity was calculated from Secchi depth and chlorophyll using two methods. One method was the inverse of the Secchi depth minus 2.5 percent of the chlorophyll-*a* concentration as given in the BATHTUB model documentation, and the other was a more complex relationship as given in Equation 7.30 of Thomann and Mueller (1987), with the results given in Figure 5-37. The correlation from Thomann and Mueller was used in the model applications, which had non-algal turbidity values ranging from 0.4 to 4 1/m, with a median value of 1.25 1/m. A correlation between chlorophyll-*a* and Total Phosphorus concentration is given in Figure 5-38. This indicates that phosphorus alone appears to explain only a small portion of the observed variability in chlorophyll-*a* concentrations.

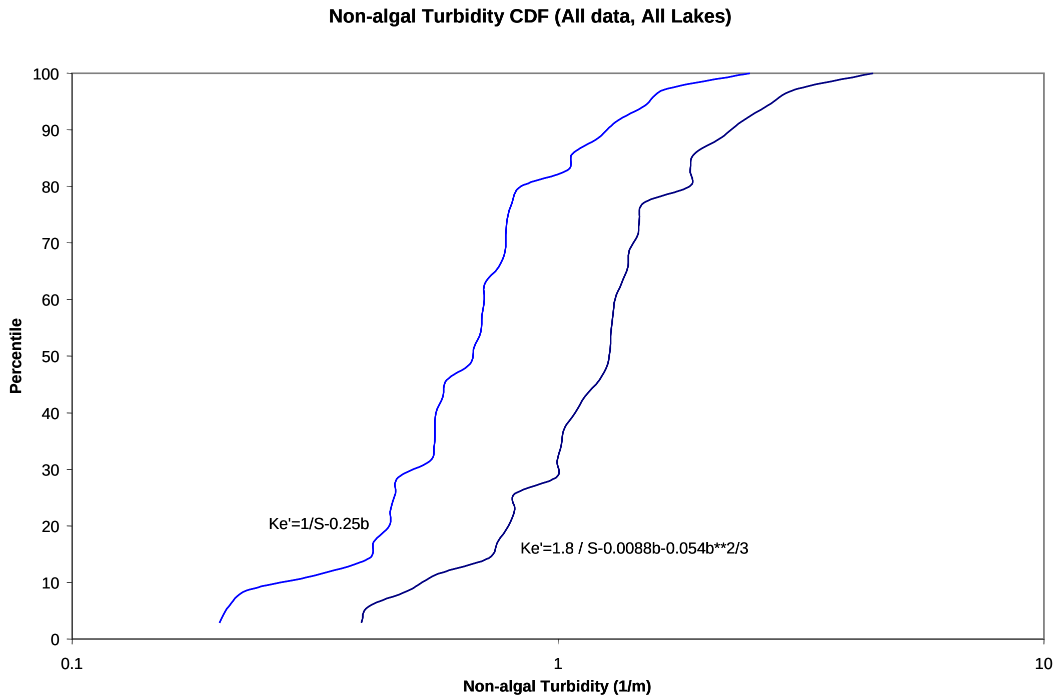


Figure 5-37 Non-algal turbidity calculated from Secchi Depth and Chlorophyll-a measurements for lakes in Ecoregion 6.

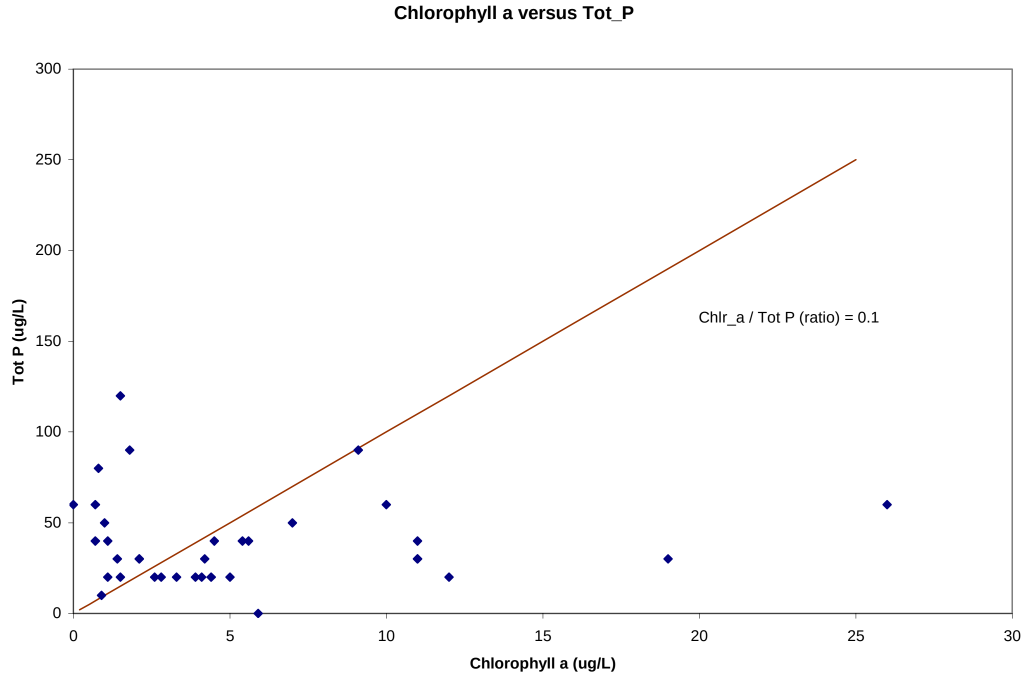


Figure 5-38 Correlation of Chlorophyll-a with Total Phosphorus for lakes in Ecoregion 6.

The normalized P and N loads given by the initial runs of the SWAT/SPARROW Models for Ecoregion 6 for were used to define the anticipated ranges of nitrogen and phosphorus loadings into Ecoregion 6 lakes. The normalized nitrogen loads averaged 5.01 gm/cu meter-year while the normalized phosphorus loads averaged 1.44 gm/cu meter-year over watershed drainage areas from 10 to 100,000 square miles (Figure 5-27). But, for the BATHTUB analysis, some estimate of the variance or variability in P and N loads is also needed. The variability about these average values was determined from the 1.4 coefficient of variation reported in the SWAT and SPARROW Model calculated P and N loads for all of Ecoregion 6. Using a 1.4 coefficient of variation results in a standard deviation for nitrogen of 7 gm/cu meter-year about a mean value of 5.01 gm/cu meter-year, and a standard deviation for phosphorus of 2 gm/cu meter-year about a mean value of 1.44 gm/cu meter-year. The upper end of the nutrient range was defined as the 95th percentile, which is roughly equal to the mean plus twice the standard deviation. For nitrogen, the 95th percentile is 19 gm/cu meter-year and for phosphorus the 95th percentile is 5.44 gm/cu meter-year. Similarly, the lower end of the range (5th percentile) is 1.3 gm/cu meter-year for nitrogen and 0.38 gm/cu meter-year for phosphorus. Based upon these results, the BATHTUB Model used nitrogen values ranging from 1 to 20 gm/cu meter-year and phosphorus values ranging from 0.1 to 5 gm/cu meter-year.

5.9.5 MODEL RESULTS: PREDICTED CHLOROPHYLL-*A* CONCENTRATIONS

The BATHTUB model as developed in the Malibu Creek TMDL was applied to a generic set of lakes. The ranges of parameter values were defined from the Ecoregion

6 data in Figures 5-28 through 5-38, and the SWAT SPARROW Model results for nitrogen loading (1 to 20 gm/cu meter-year) and phosphorus loading (0.1 to 5 gm/cu meter-year). Residence times varied from 0.05 to 16.9 years, which spans from the 10^{th} to 90^{th} percentile of the data in Figure 5-35. Lake depth was held constant for this analysis at the median value for Ecoregion 6 (5.7 meters in Figure 5-36). Non-algal turbidity was varied from the 5^{th} to 95^{th} percentile (0.4 to 4 1/m), which spans the values defined using the Thomann and Mueller correlation in Figure 5-37.

Chlorophyll-*a* concentrations were predicted for all these cases. The results are given in Figures 5-39 through 5-41 for residence times of 0.25 (30^{th} percentile), 0.51 (50^{th} percentile), and 2.03 (70^{th} percentile) years and a non-algal turbidity of 1.4 1/m. Figure 5-42 overlays the results for all three residence times (0.25, 0.51, and 2.03 years) on a single plot. The model results typically display phosphorus limited algal growth when nitrogen loadings exceed 5,000 ug/L-year and phosphorus loadings are less than 200 ug/L-year. Conversely, the model results display nitrogen limited algal growth when phosphorus loadings exceed 500 ug/L-year and nitrogen loadings are less than 2,000 ug/L-year. Maximum predicted chlorophyll-*a* concentrations are 50 to 60 ug/L at the upper end of the nutrient loadings and residence times. The minimum predicted chlorophyll-*a* concentrations are 2 ug/L for a residence time of 0.25 years, approximately 6 ug/L for a residence time of 0.51 years, and approximately 15 ug/L for a residence time of 2.03 years. The model predicted chlorophyll-*a* concentrations appear quite sensitive to residence time for the three residence time values shown, with higher concentrations at larger residence times. The model predicted chlorophyll-*a* concentrations are also sensitive to nutrient concentrations along the diagonal of the plots, with higher concentrations at larger nutrient concentrations. Off the diagonal, however, the model predicted chlorophyll-*a* concentrations are only sensitive to the limiting nutrient concentrations.

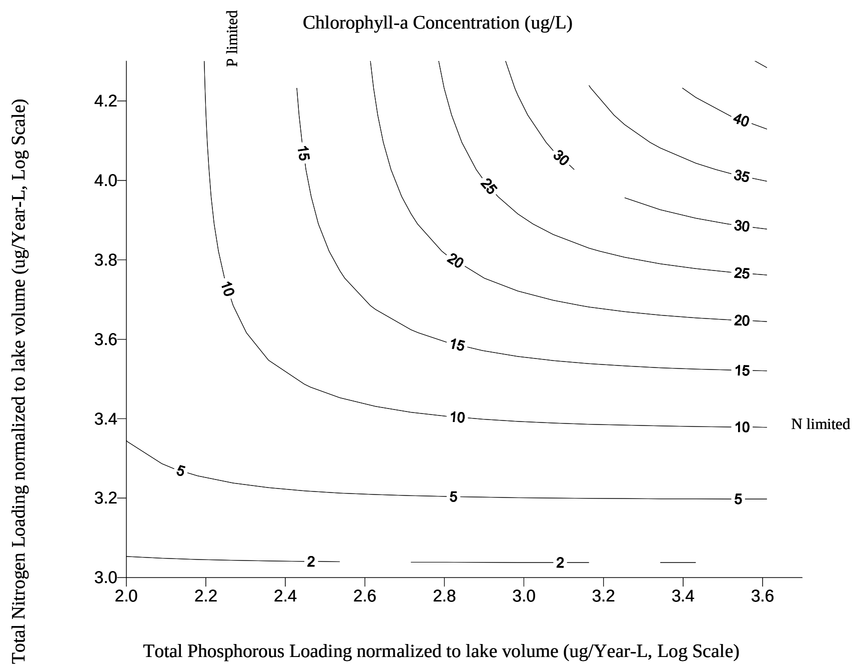


Figure 5-39 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 0.25 Years and Non-algal turbidity of 1.25 1/m.

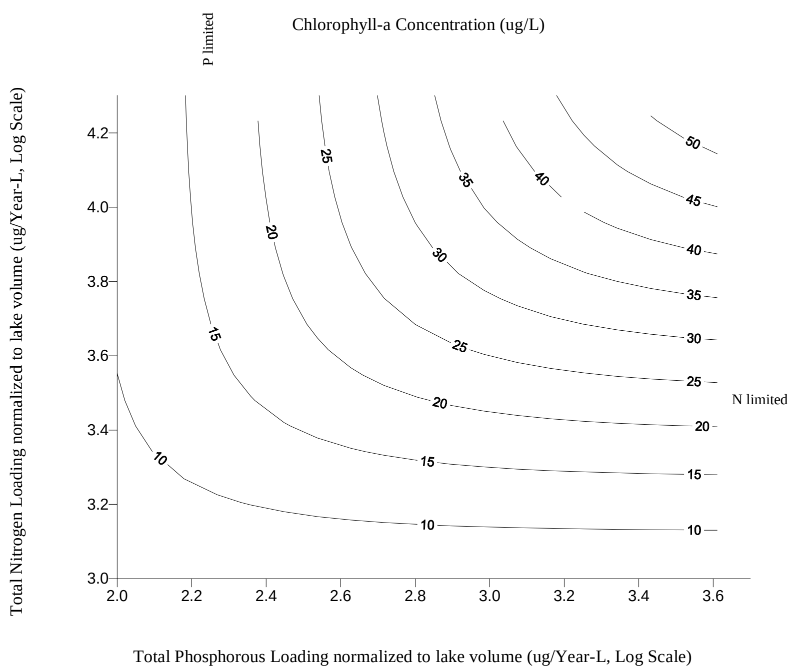


Figure 5-40 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 0.51 Years and Non-algal turbidity of 1.25 1/m

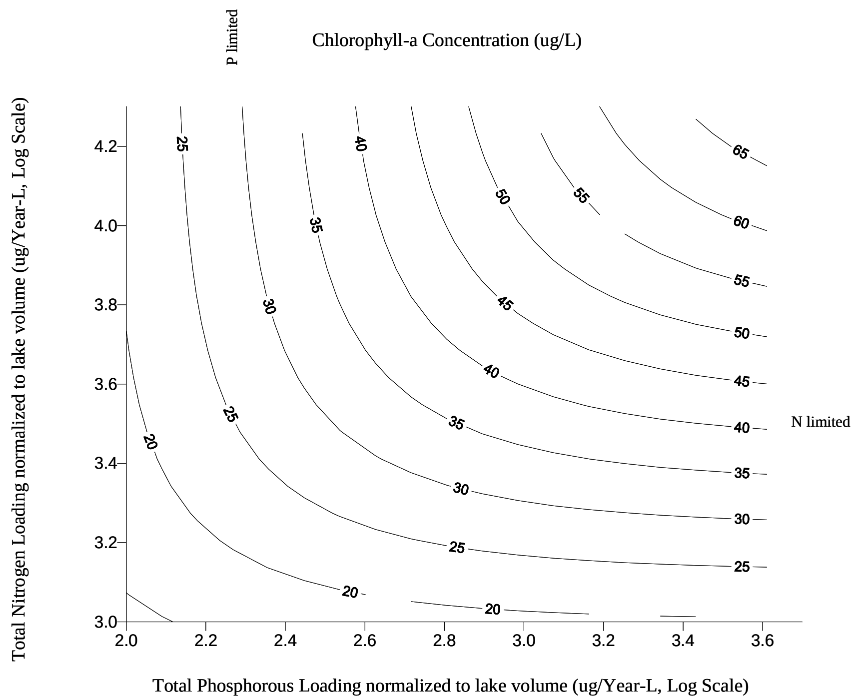


Figure 5-41 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 2.03 Years and Non-algal turbidity of 1.25 1/m

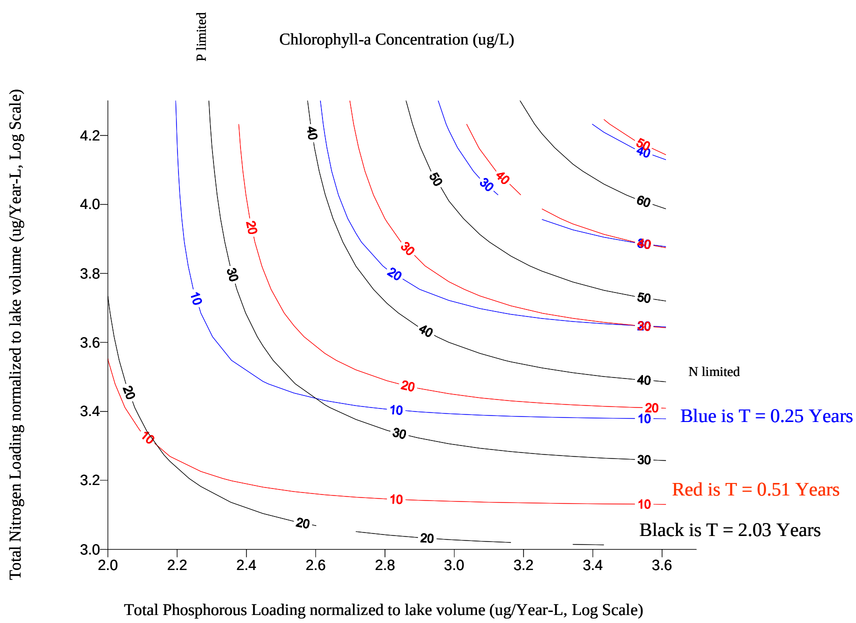


Figure 5-42 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times of 0.25, 0.51, and 2.03 Years and Non-algal turbidity of 1.25 1/m

Similarly, predicted chlorophyll-*a* concentrations are given in Figures 5-43 through 5-46 for residence times of 0.25, 0.51, and 2.03 years and a non-algal turbidity of 4.0 1/m, and in Figures 5-47 through 5-50 for residence times of 0.25, 0.51, and 2.03 years and a non-algal turbidity of 0.4 1/m Figures 5-46 and 5-50 overlays the results for all three residence times on a single plot for non-algal turbidity values of 4.0 and 0.4 1/m, respectively. The model results also display phosphorus and nitrogen limited algal growth in the lower-right and upper-left quadrants. Maximum predicted chlorophyll-*a* concentrations are only 35 ug/L for a non-algal turbidity value of 4.0 1/m and 70 ug/L for a non-algal turbidity value of 0.4 1/m, which contrasts with maximum predicted chlorophyll-*a* concentrations of 60 ug/L for a non-algal turbidity value of 1.4 1/m. The maximum predicted chlorophyll-*a* concentrations are sensitive to non-algal turbidity values, and appear more sensitive to highest non-algal turbidity values (the 90th percentile) than to the lowest non-algal turbidity values (the 10th percentile). The minimum predicted chlorophyll-*a* concentrations for a non-algal turbidity value of 4.0 1/m are less than 1 ug/L for a residence time of 0.25 years, approximately 2 ug/L for a residence time of 0.51 years, and approximately 9 ug/L for a residence time of 2.03 years. The minimum predicted chlorophyll-*a* concentrations for a non-algal turbidity value of 0.4 1/m are less than 2 ug/L for a residence time of 0.25 years, approximately 4 ug/L for a residence time of 0.51 years, and approximately 14 ug/L for a residence time of 2.03 years. The model predicted chlorophyll-*a* concentrations appear quite sensitive to residence time over the range of residence time values shown, with higher concentrations at larger residence times.

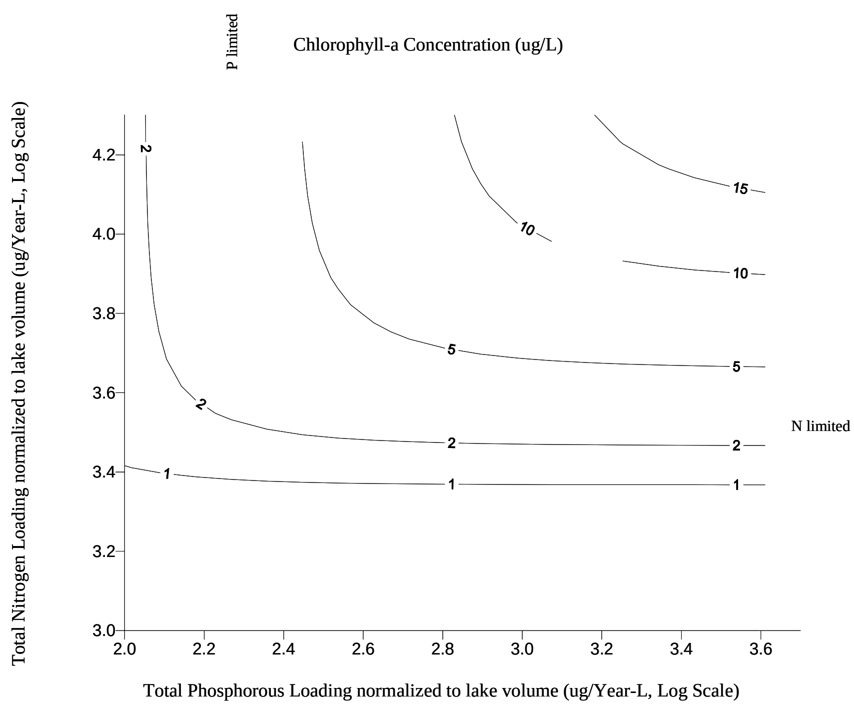


Figure 5-43 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 0.25 Years and Non-algal turbidity of 4.0 1/m

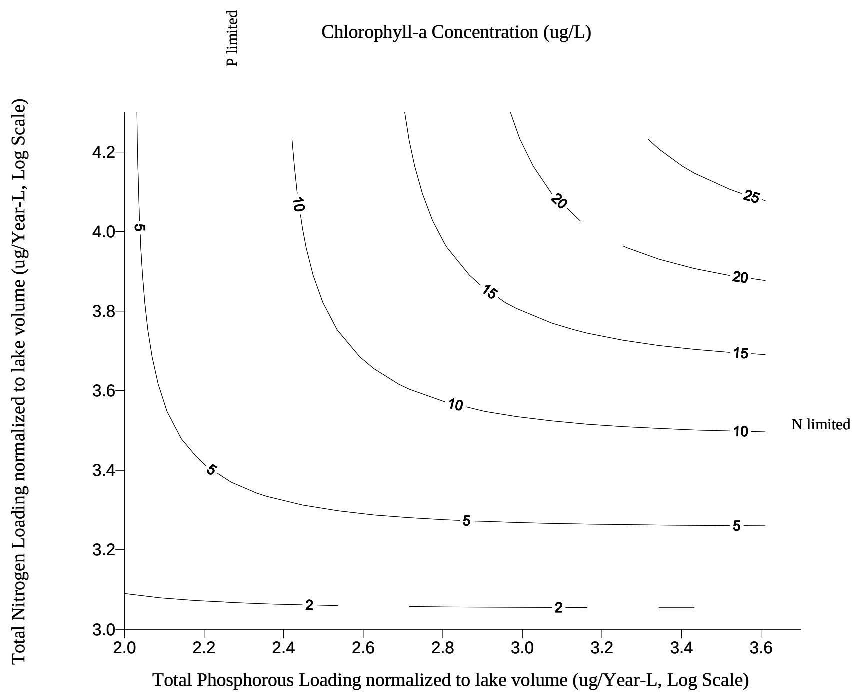


Figure 5-44 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 0.51 Years and Non-algal turbidity of 4.0 1/m

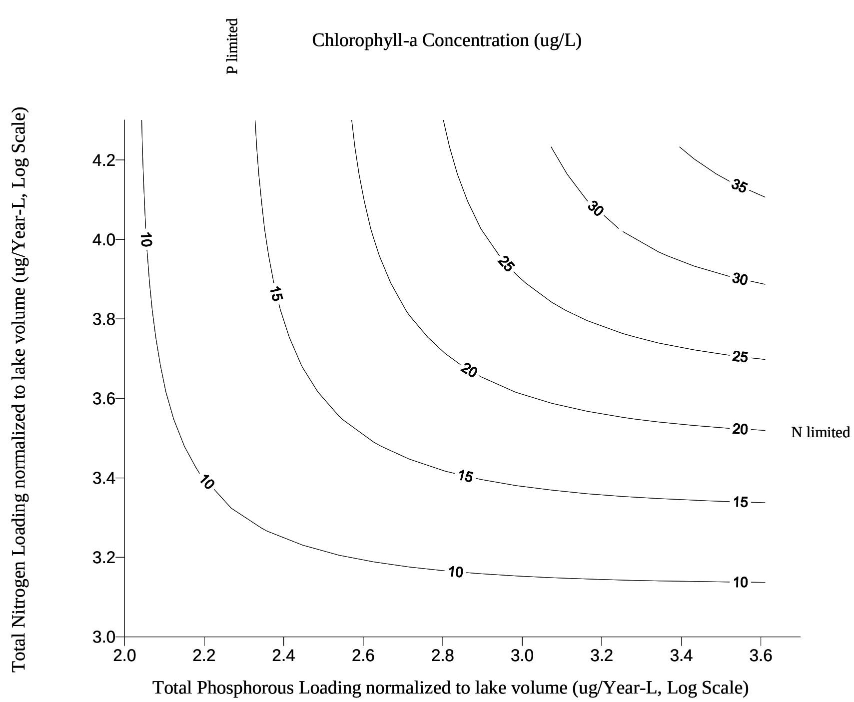


Figure 5-45 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 2.03 Years and Non-algal turbidity of 4.0 1/m

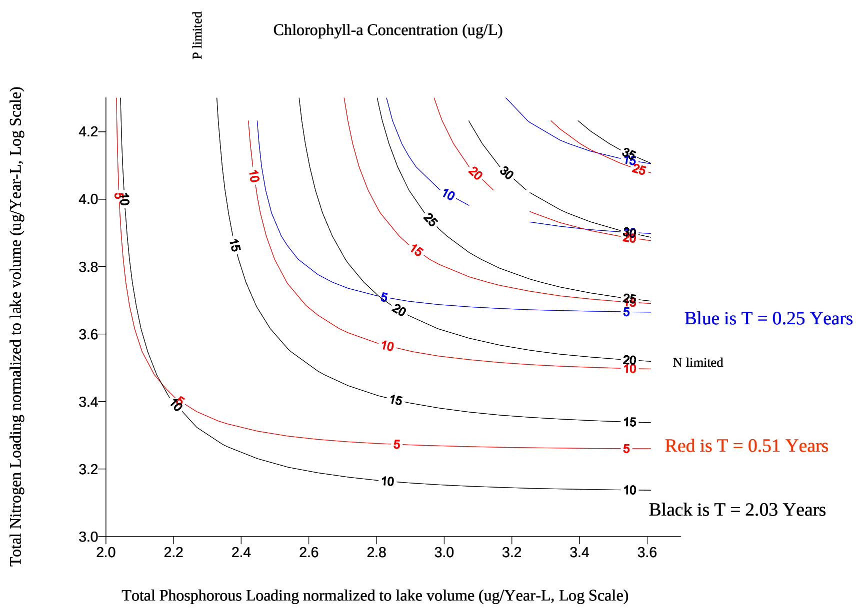


Figure 5-46 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times of 0.25, 0.51, and 2.03 Years and Non-algal turbidity of 4.0 1/m

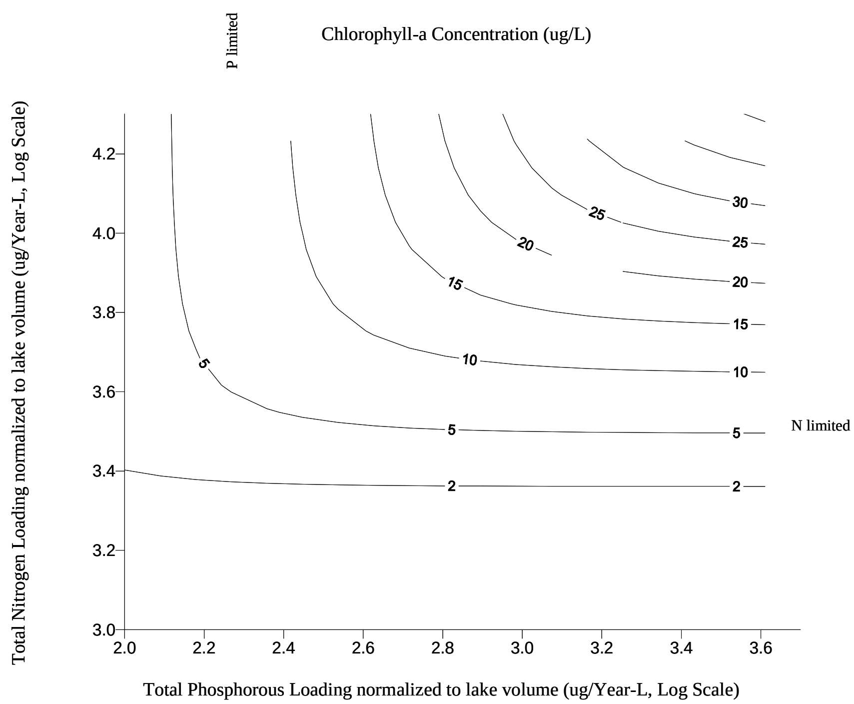


Figure 5-47 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 0.25 Years and Non-algal turbidity of 0.4 1/m.

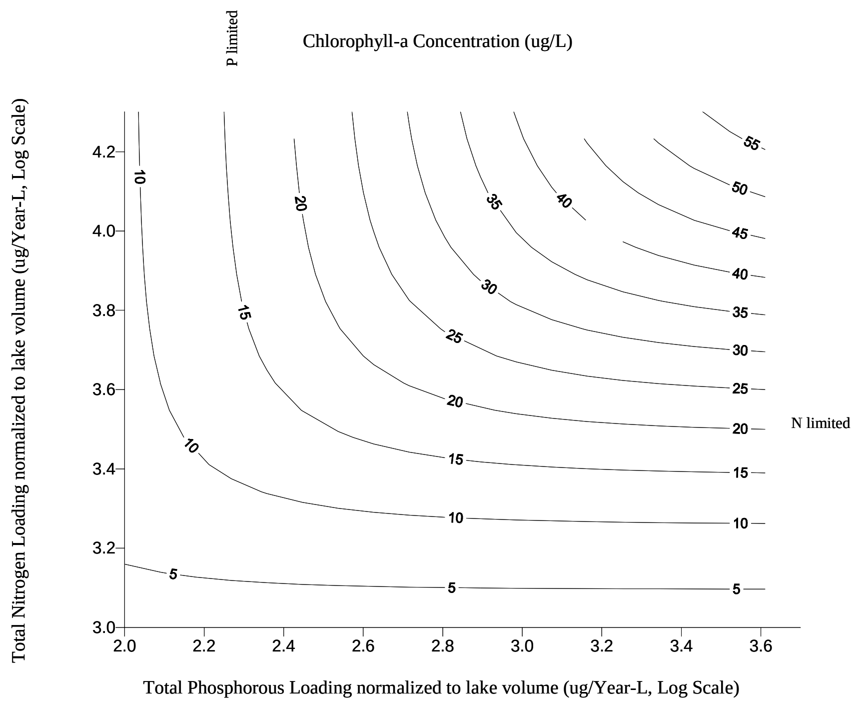


Figure 5-48 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 0.51 Years and Non-algal turbidity of 0.4 1/m

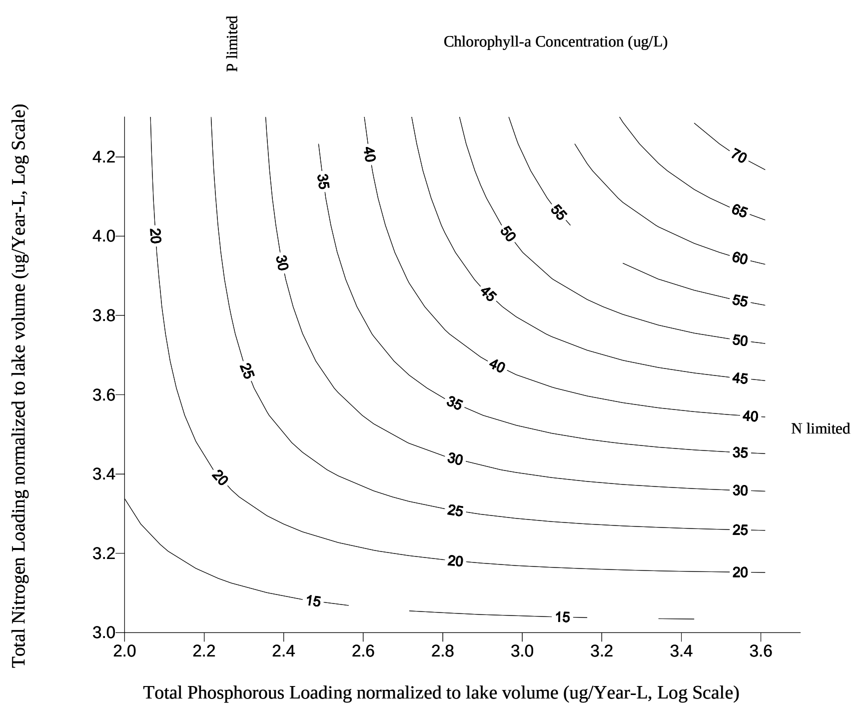


Figure 5-49 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for a Residence Time of 2.03 Years and Non-algal turbidity of 0.4 1/m

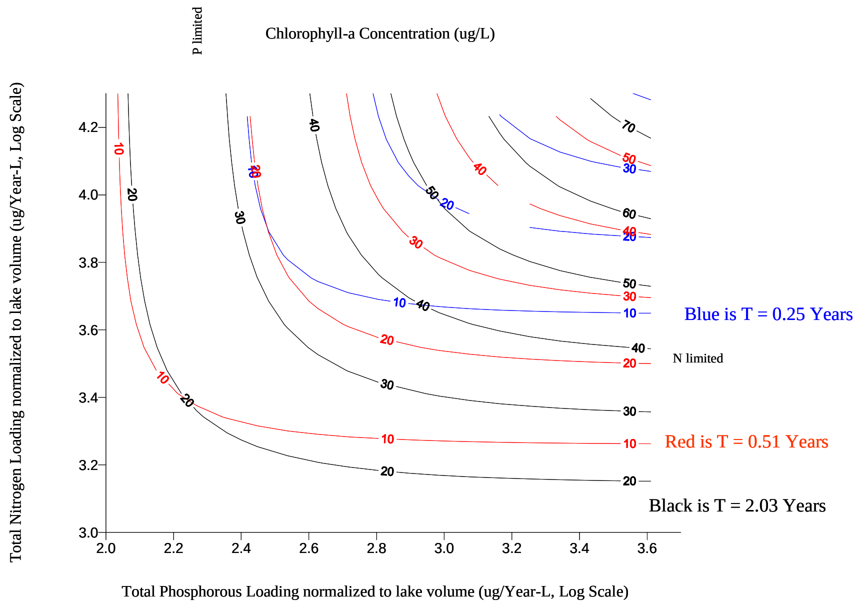


Figure 5-50 Lake Chlorophyll-a Concentration versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times of 0.25, 0.51, and 2.03 Years and Non-algal turbidity of 0.4 1/m

5.9.6 CHLOROPHYLL-*A* TARGET CRITERIA RESPONSE SURFACES

The model results in Figures 5-39 through 5-50 were used to produce generic response surfaces for the chlorophyll-*a* target criteria relative to various values of non-algal turbidity. These results are given for a turbidity value of 1.4 1/m in Figures 5-51, 5-52, and 5-53, where chlorophyll-*a* targets of 10 ug/L, 25 ug/L and 40 ug/L, respectively, are plotted as a function of normalized nitrogen loading, normalized phosphorus loading, and residence time. Similarly, results are given for a turbidity value of 4.0 1/m in Figures 5-54, 5-55, and 5-56, and for a turbidity value of 0.4 1/m in Figures 5-57, 5-58, and 5-59.

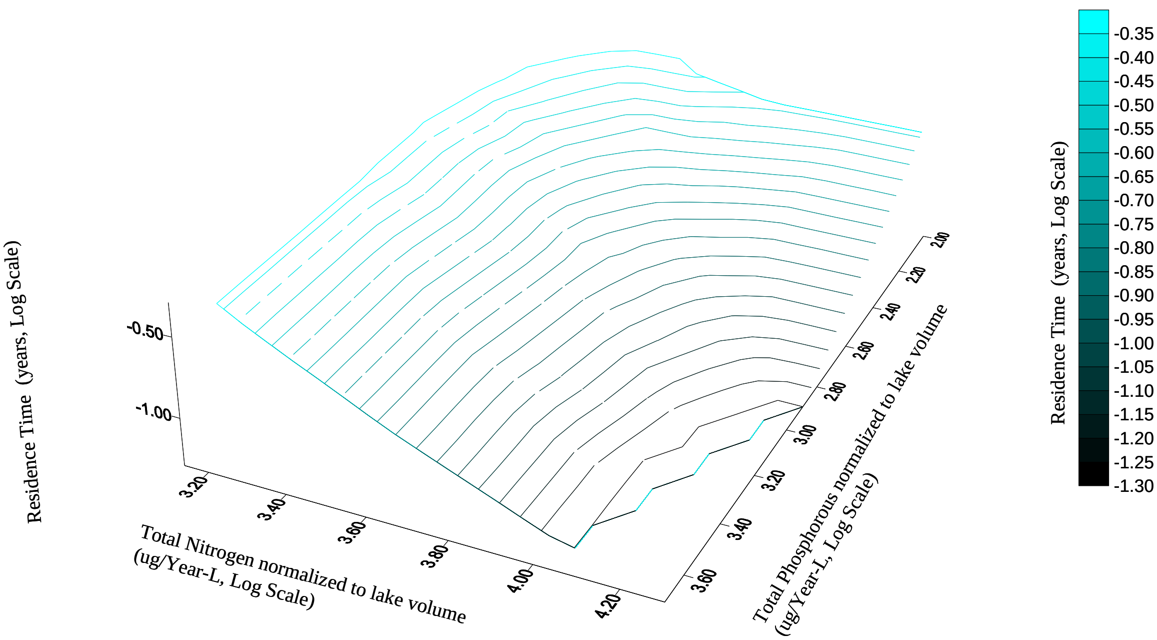


Figure 5-51 Chlorophyll-a Target Concentrations (10 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 1.25 1/m

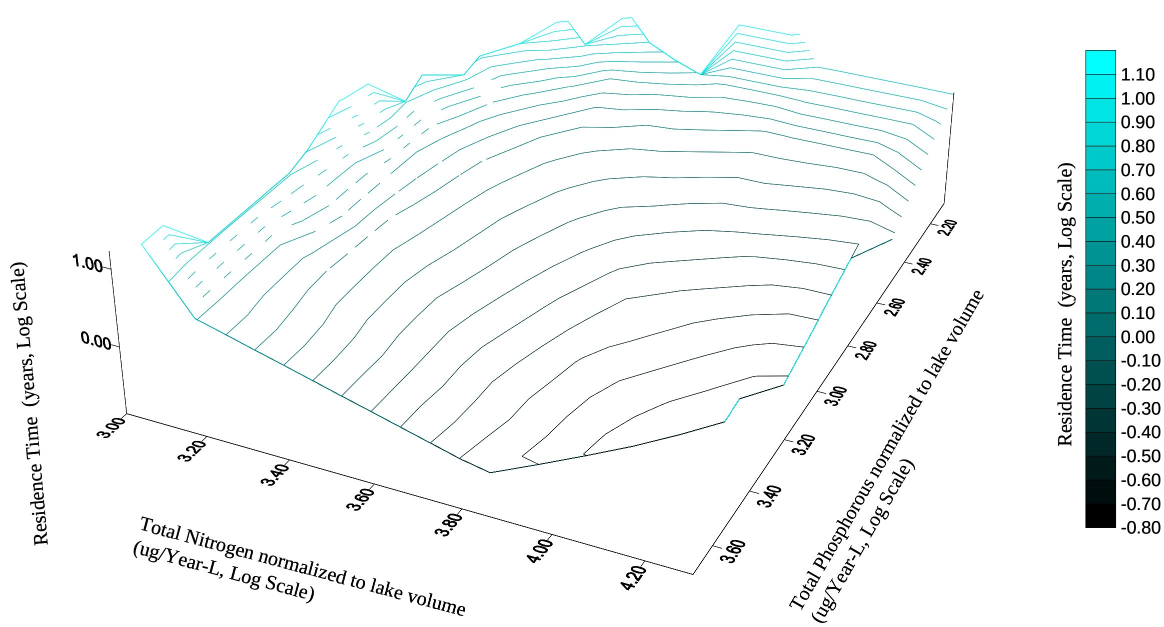


Figure 5-52 Chlorophyll-a Target Concentrations (25 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 1.25 1/m

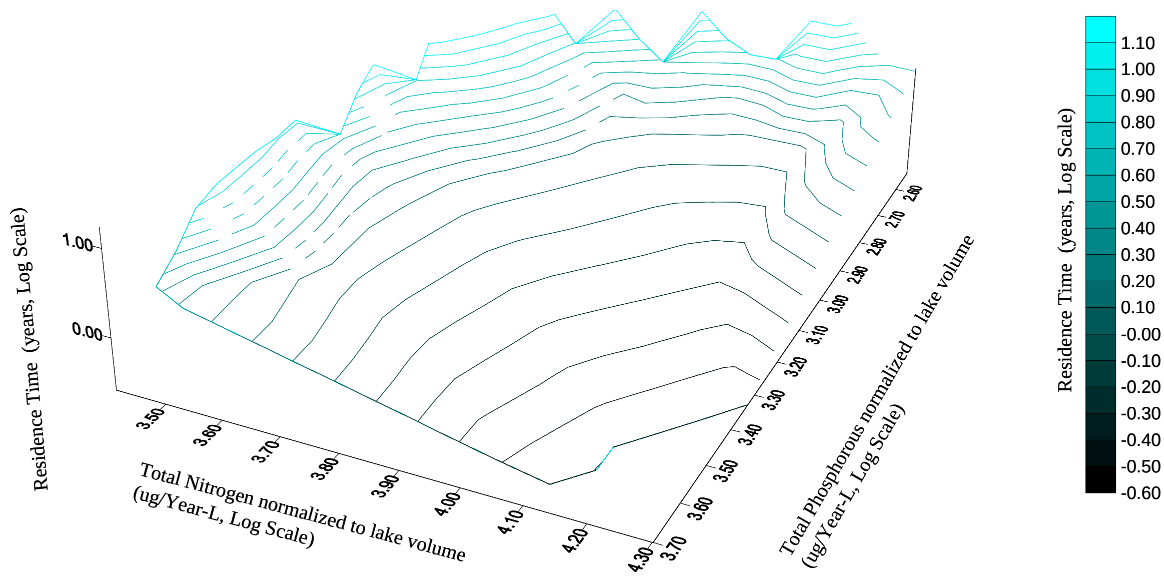


Figure 5-53 Chlorophyll-a Target Concentrations (40 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 1.25 1/m

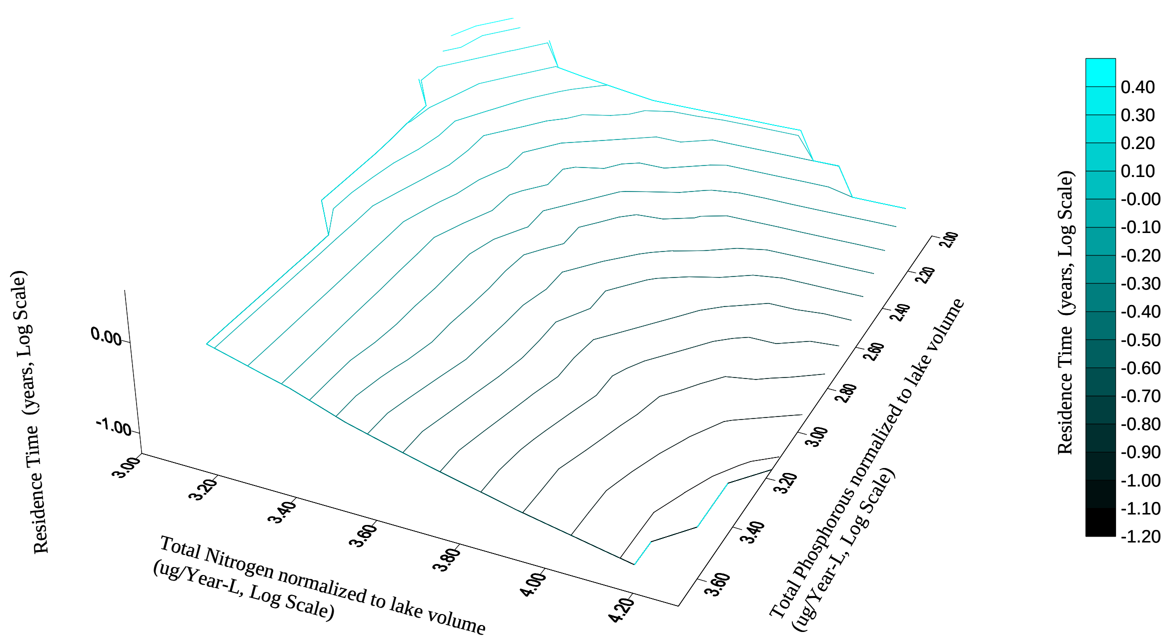


Figure 5-54 Chlorophyll-a Target Concentrations (10 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 4.0 1/m

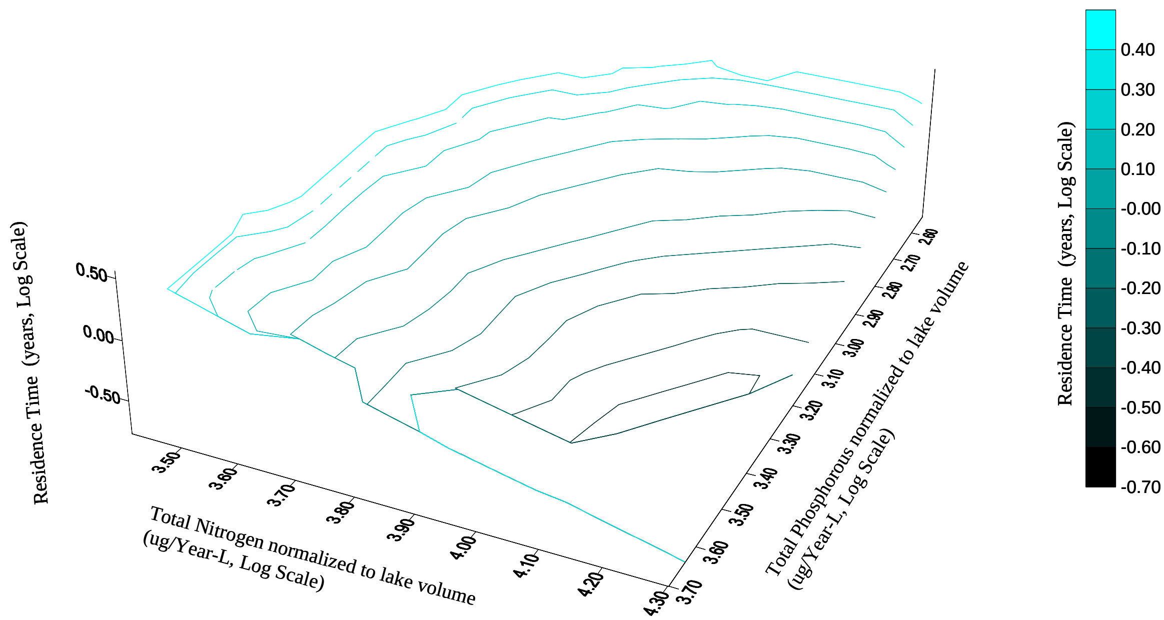


Figure 5-55 Chlorophyll-a Target Concentrations (25 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 4.0 1/m

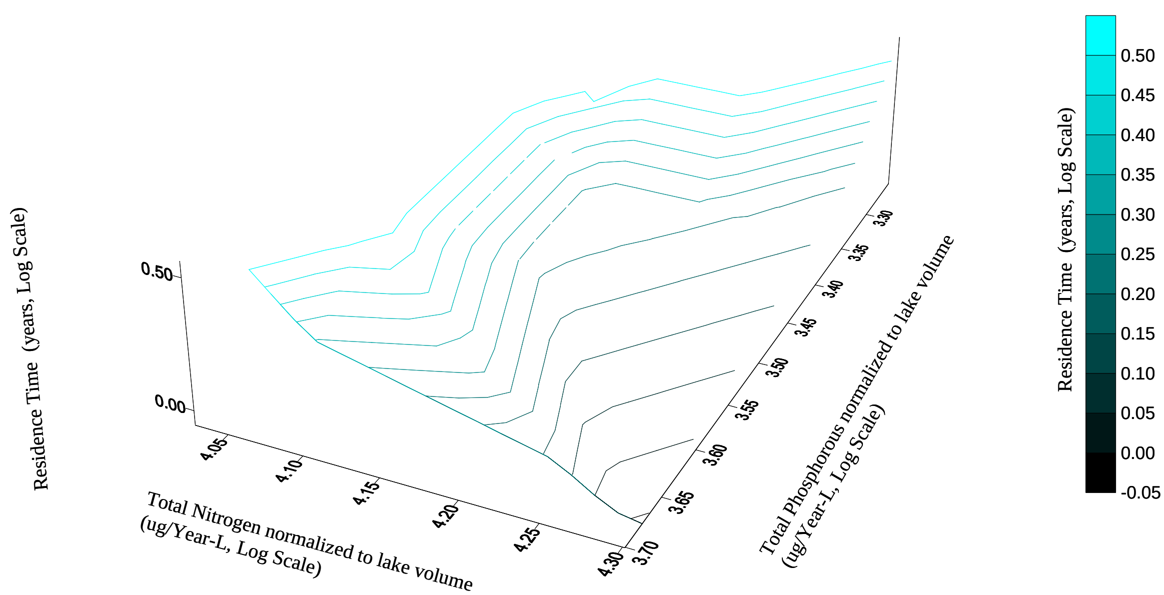


Figure 5-56 Chlorophyll-a Target Concentrations (40 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 4.0 1/m

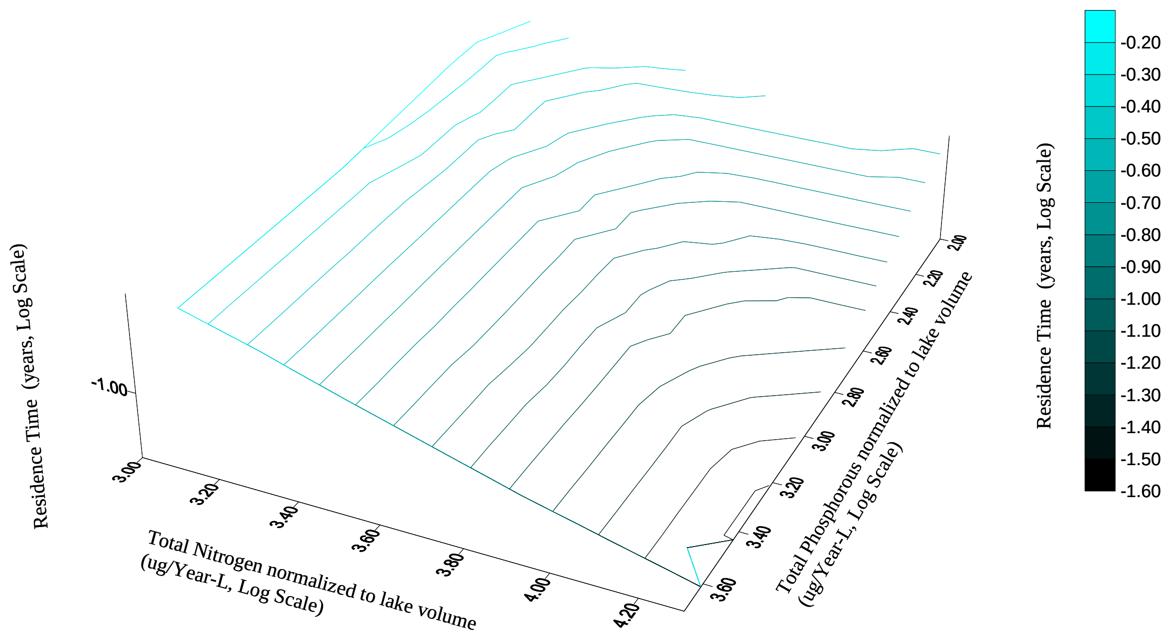


Figure 5-57 Chlorophyll-a Target Concentrations (10 ug/l) versus Phosphorus and NitrogenLoading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 0.4 1/m

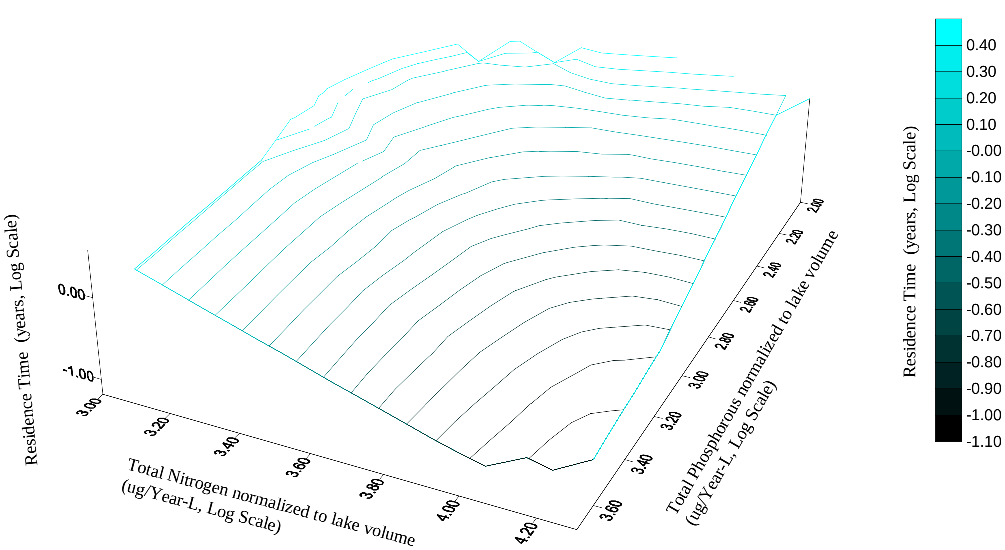


Figure 5-58 Chlorophyll-a Target Concentrations (25 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 0.4 1/m

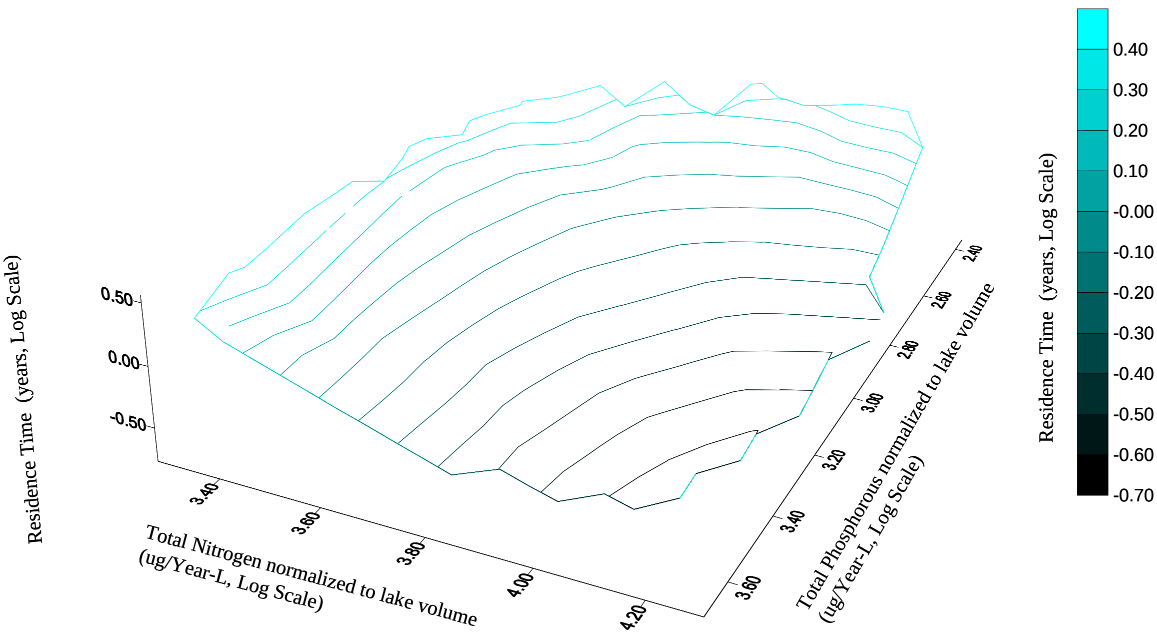


Figure 5-59 Chlorophyll-a Target Concentrations (40 ug/l) versus Phosphorus and Nitrogen Loading (lake volume normalized) for Residence Times from 0.05 to 17 Years and non-algal turbidity of 0.4 1/m

These results show that there is an approximately log-linear inverse relationship between allowable normalized nutrient concentrations and residence times, with allowable normalized nutrient concentrations increasing with decreasing residence time. Allowable normalized nutrient concentrations also increase with increasing turbidity values. A much larger range of nutrient and residence time parameter values exceeds the 10 ug/L target than the 25 or 40 ug/L target values. For example, essentially all residence times less than 0.25 years resulted in chlorophyll-*a* concentrations less than a 40 ug/L target, while residence times had to be less than 0.05 years to result in all chlorophyll-*a* concentrations less than a 10 ug/L target. All normalized nitrogen concentrations less than 3,200 ug/L resulted in chlorophyll-a concentrations less than a 40 ug/L target, while normalized nitrogen concentrations had to be less than 1,500 ug/L to result in all chlorophyll-*a* concentrations less than a 10 ug/L target. Similarly, all normalized phosphorus concentrations less than 400 ug/L resulted in chlorophyll-*a* concentrations less than a 40 ug/L target, while normalized phosphorus concentrations had to be less than 100 ug/L to result in all chlorophyll-*a* concentrations less than a 10 ug/L target.

5.9.7 Model Parameter Sensitivity Analyses

Figures 5-60, 5-61, 5-62, and 5-63 show generalized sensitivity plots for normalized total phosphorus, normalized total nitrogen, residence time, and non-algal turbidity for model results grouped into those runs exceeding 25 ug/L chlorophyll-*a*, and those runs with less than 25 ug/L chlorophyll-*a*. Residence time appears to be the most sensitive parameter (defined by the spread between the two group CDFs), and non-algal turbidity the least. Non-algal turbidity was inversely correlated with chlorophyll-*a*, with lower chlorophyll-*a* associated with higher non-algal turbidity. Normalized total phosphorus, normalized total nitrogen, and residence time are directly correlated with chlorophyll-*a*, with higher chlorophyll-*a* associated with higher normalized total phosphorus, higher normalized total nitrogen, and higher residence times.

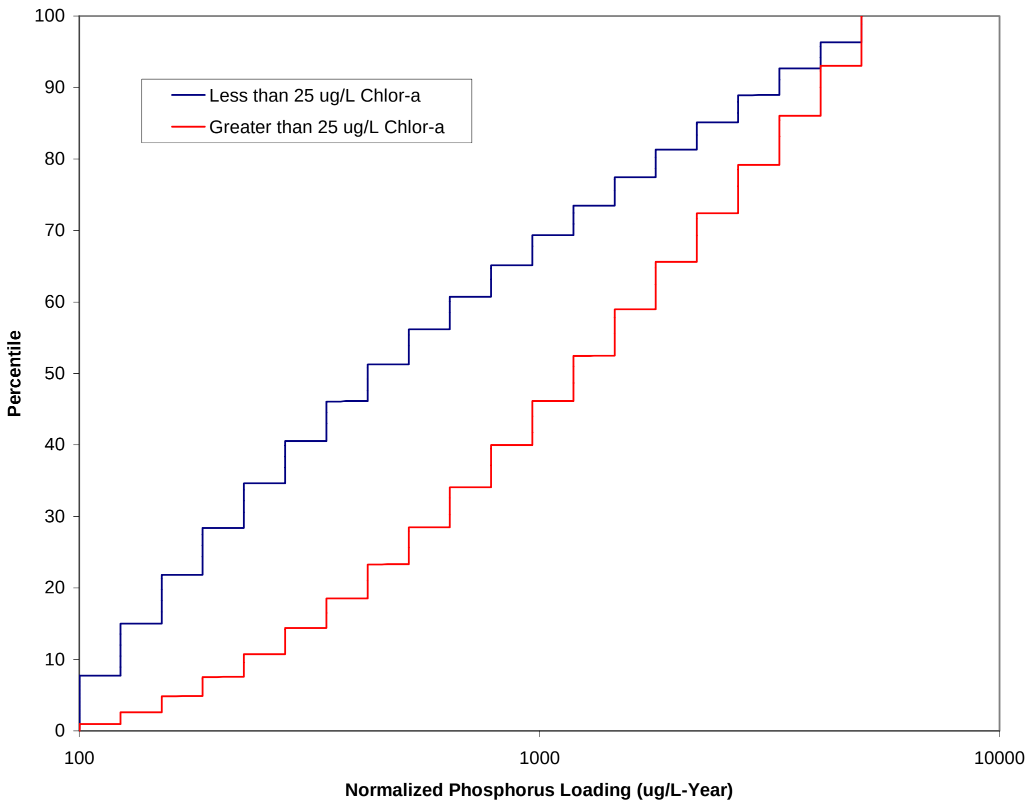


Figure 5-60 Generalized sensitivity plot for phosphorus variable, all parameters varying (P, N, Res Time, and Turbidity)

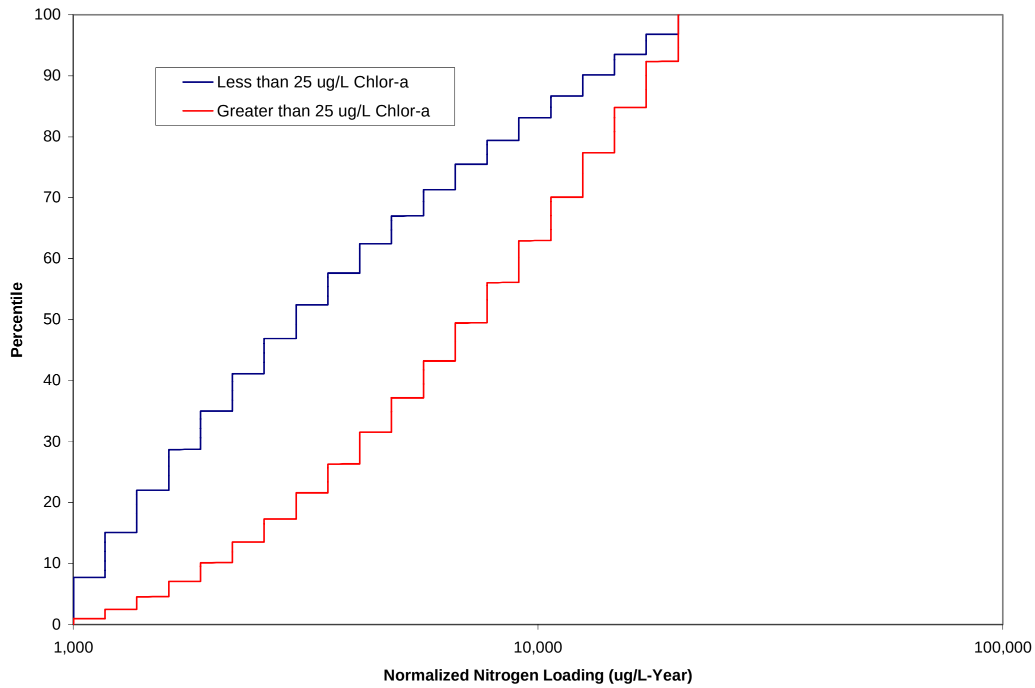


Figure 5-61 Generalized sensitivity plot for nitrogen variable, all parameters varying (P, N, Res Time, and Turbidity)

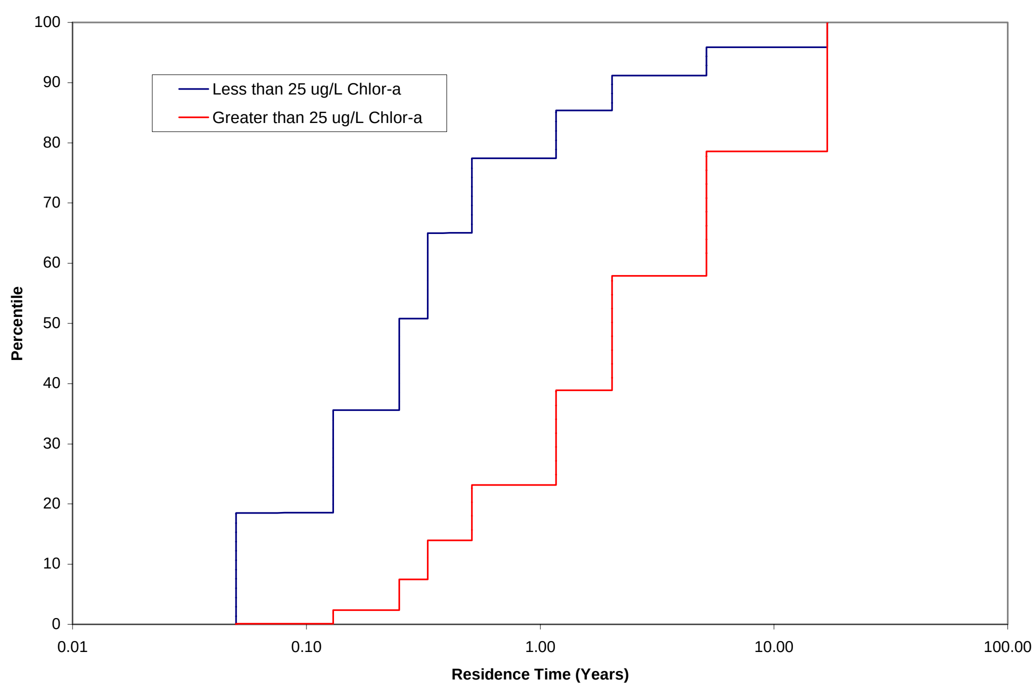


Figure 5-62 Generalized sensitivity plot for residence time variable, all parameters varying (P, N, Res Time, and Turbidity)

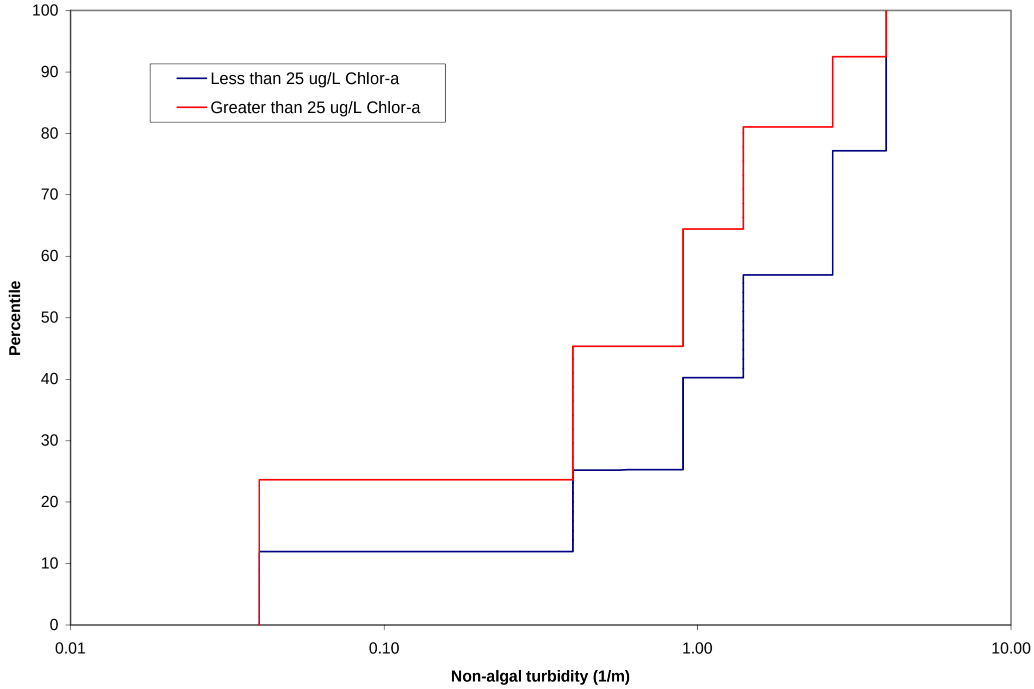


Figure 5-63 Generalized sensitivity plot for turbidity variable, all parameters varying (P, N, Res Time, and Turbidity)

5.9.8 FINDINGS OF BATHTUB MODELING

The BATHTUB Model was used in this analysis for Ecoregion 6 to calculate the algal response to nutrient loadings as a function of lake residence time and non-algal turbidity for three target chlorophyll-*a* concentrations. The BATHTUB Model parameters were defined based upon prior model applications in Ecoregion 6, an Ecoregion 6 database of lake water quality and hydraulic parameters, and Ecoregion 6 nitrogen and phosphorus loadings estimated using the SWAT SPARROW model. The model results show that regardless of lake residence time or turbidity, all normalized phosphorus concentrations less than 400 ug/L resulted in predicted growing-season average chlorophyll-*a* concentrations less than a 40 ug/L target, while normalized phosphorus concentrations had to be less than 100 ug/L to result in all chlorophyll-*a* concentrations less than a 10 ug/L target. Similarly, regardless of lake residence time or turbidity, all normalized nitrogen concentrations less than 3,200 ug/L resulted in growing-season average chlorophyll-*a* concentrations less than a 40 ug/L target, while normalized nitrogen concentrations had to be less than 1,500 ug/L to result in all chlorophyll-*a* concentrations less than a 10 ug/L target. The model results were very sensitive to residence time and moderately sensitive to turbidity over the range of Ecoregion 6 parameter values.

5.10 STREAM PERIPHYTON ANALYSIS

5.10.1 SIMULATION MODELING OF PERIPHYTON IN STREAMS

Nutrient loading can lead to undesirable densities of benthic algae or periphyton in flowing streams. However, in many cases light availability and losses to scour are important controls on periphyton density. Estimating nutrient load or concentration targets for flowing streams can thus present a difficult challenge.

One line of evidence for the estimation of benthic algal or periphyton growth potential in streams is provided by simulation modeling. Recently, a benthic algal component has been incorporated into a revised version of the QUAL2E water quality model, known as QUAL2K (Chapra and Pelletier, 2003). This parametric representation can be adapted to investigate benthic algal responses to availability of light and nutrients. The following sections lay out the details and initial results of this approach.

5.10.2 PARAMETRIC REPRESENTATION OF BENTHIC ALGAL GROWTH POTENTIAL

Following the QUAL2K development, if B is the concentration of benthic algal biomass (mass per area), then

$$\frac{dB}{dt} = K_p - K_r - K_d .$$

Where, K_p is the rate of photosynthesis, K_r is the rate of algal respiration, and K_d is the rate of algal death (all in dry weight mass per time per unit area).

5.10.2.1 Photosynthesis

K_p is defined by the product of the maximum photosynthesis rate ($K_{p\max}$, g DW/m^2/d), the benthic algae nutrient attenuation factor (ϕ_{Nb}), and the benthic algae light attenuation factor (ϕ_{Lb}). ϕ_{Nb} and ϕ_{Lb} are dimensionless factors ranging from 0 to 1.

$$K_p = K_{p\max} * \phi_{Nb} * \phi_{Lb} .$$

Note that in QUAL2K K_{pmax} is defined as a fixed number, rather than as a rate (per day) multiplied times concentration. This helps reflect self-limiting factors on periphyton growth. The maximum photosynthetic rate is temperature dependent, and temperature effects on rate are specified with the Arrhenius relationship to the maximum photosynthetic rate at a reference temperature of 20°C ($K_{p\max,20}$):

$$K_{p\max} = K_{p\max,20}\theta^{(T-20)} .$$

The benthic algae nutrient attenuation factor (ϕ_{Nb}) is represented by the Michaelis-Menten nutrient limitation equation for inorganic nitrogen and phosphorus:

$$\phi_{Nb} = \min(\frac{n_a + n_n}{k_{sNb} + n_a + n_n}, \frac{p_i}{k_{sPb} + p_i}) .$$

Where n_a is the ammonia concentration in the above water (mg-N/L), n_n is the nitrate plus nitrite concentration in the overlying water (mg-N/L), k_{sNb} is the nitrogen half-saturation constant (mg-N/L), p_i is the inorganic phosphorus in the overlying water, and k_{sPb} is the phosphorus half-saturation constant (mg-P/L).

For benthic algae, light limitation depends on the amount of photosynthetically active radiation (I) reaching the bottom of the water column, which is defined by the Beer-Lambert law:

$$I = I_o e^{-k_e H} .$$

Where I_o is the solar radiation at the water surface (cal/cm^2/d), k_e is the light extinction coefficient (m^{-1}), and H is the water depth (m).

The half-saturation light model defines the light limitation factor with a benthic algae light parameter (K_{Lb}) so that

$$\phi_{Lb} = \frac{I_o e^{-k_e H}}{K_{Lb} + I_o e^{-k_e H}} .$$

5.10.2.2 Respiration and Death

Benthic algae respiration is a first-order rate defined as

$$K_r = B * k_{rb}$$

Where B is benthic algae (g/m^2) and k_{rb} is the temperature-dependent bottom algae respiration rate (d^{-1}).

Benthic algae death is also a first-order rate defined as

$$K_d = B * k_{db}$$

Where B is benthic algae (g/m^2) and k_{db} is the temperature-dependent bottom algae respiration rate (d^{-1}).

5.10.2.3 Steady State Benthic Algal Biomass

Under steady-state conditions, dB/dt goes to zero, and the steady-state maximum benthic algal biomass (g/m^2) may be calculated as:

$$B = \frac{K_{p\max} * \phi_{Nb} * \phi_{Lb}}{k_{rb} + k_{db}}.$$

5.10.2.4 Kinetic Parameter Values

The steady-state estimate of biomass is very sensitive to the sum of the loss rates, k_{rb} + k_{db}. These values are highly variable and not well documented for benthic algae. Respiration rates for benthic algae used in modeling are typically around 0.1 per day (Bowie *et al*., 1985), while the recommended value for planktonic algae is 0.125 per day (Wool *et al*., 2003), which seems a reasonable first cut estimate for simulation. The death rate used for simulation of benthic algae is often inflated above a natural death rate to account for grazing pressure and scour. We consider scour separately, but must account for grazing within the “natural” death rate. It is assumed that the typical death rate for planktonic algae (0.02 per day; Wool *et al*., 2003) is a reasonable representation of the benthic algal death rate in the absence of scour, which will be considered later. Estimates of $K_{p\ max}$, K_{sNb}, K_{sPb}, and K_{Lb} were set to QUAL2K defaults for the initial analysis. The initial kinetic parameter values are summarized in Table 5-10.

Table 5-10
QUAL2K Default Kinetic Parameters for Benthic Algae

Parameter	Default Value	Units
$K_{p\max}$ (maximum photosynthetic rate)	60	g/m^2/d
θ (Arrhenius temperature constant)	1.07	Unitless
k_{rb} (respiration rate)	0.125	1/d
k_{db} (natural death rate)	0.02	1/d
k_{sNb} (nitrogen half-saturation constant)	0.015	mg-inorganic N/L
k_{sPb} (phosphorus half-saturation constant)	0.002	mg-inorganic P/L
K_{Lb} (light half-saturation constant)	50	cal/cm^2/d

5.10.2.5 *Relationship to Biomass as Chlorophyll* a

The QUAL2K model predicts benthic algal biomass as grams of ash-free dry weight (AFDW) per square meter. The bulk of the literature values on benthic algal biomass and targets are reported as milligrams of chlorophyll *a* per square meter. Unfortunately, the chlorophyll content of benthic algae is highly variable, depending on species, number of heterotrophs present, and light conditions, rendering a conversion difficult, and many different values are reported in the literature. Rather than rely on a literature conversion, which may not be appropriate to California streams, we relied on empirical evidence. EMAP data for California reports both ash-free dry weight and chlorophyll *a* for periphyton, along with their ratio. The ratio of chlorophyll *a* density (mg/m^2) to ash-free dry weight (g/m^2) on 173 California EMAP samples ranged from 0.06 to 6.73, with an average of 1.673 and a median of 1.14. Most of the values lie between 0.6 and 2.1 (interquartile range).

5.10.3 MAXIMUM BENTHIC ALGAL GROWTH POTENTIAL

The maximum benthic algal growth potential is assessed under typical summer conditions with no shading and no additional algal loss due to scour. This yields a theoretical upper bound on expected average biomass as a function of nutrient concentration.

5.10.3.1 *Temperature and Light Conditions*

Algal growth is affected by both temperature and light. For this theoretical simulation, temperature is assumed to be 20°C – thus requiring no Arrhenius correction.

For light, insolation at the water surface, I_0, depends on latitude, time of year, and sky conditions. For initial comparisons regarding maximum potential benthic algal growth it is appropriate to use cloudless summer insolation for summer conditions. At the summer maximum (June), variation with latitude in the northern hemisphere is small and the insolation at the outer edge of the atmosphere over California is approximately 975 cal/cm^2/d. Even without clouds, this amount is reduced by transmission through the air mass. Without shading (or urban smog), the atmospheric transmission factor is about 0.71 (Black *et al*., 1954), yielding a value of insolation at the land surface for cloudless summer skies of approximately 690 cal/cm^2/d. This is further reduced by reflection at the water surface (albedo), which, for high solar altitude, is approximately 0.05, yielding a value of I_0 of 658 cal/cm^2/d. This value is used to establish a baseline of potential benthic algal biomass.

Canopy cover on a stream will further reduce the value of I_0. Estimates in USACE (1956) show that a forest canopy density of 20 percent (percentage of surface area covered by a horizontal projection of the vegetation canopy) reduces surface insolation to 45 percent of that on an unshaded stream, a 40 percent canopy density reduces insolation to about 25 percent, and an 80 percent canopy density reduces it to about 10 percent. Streams in steep topography will have further reductions due to topographic shading.

The simulation also depends on light penetration to the stream bottom, which varies with the product k_eH, the extinction coefficient times average depth. For the initial simulations, this product is assumed to have a value of 0.5 (e.g., a depth of 1 m with an extinction coefficient of 0.5 m^{-1}, typical of a low turbidity stream.)

The California Ecoregion 6 Nutrient Criteria Pilot Study (Tetra Tech, 2003) lists ranges of nutrient concentrations found in California streams ranging from minimally impacted to impaired. Table 5-11 summarizes these concentrations, which can be used to assess the sensitivity of maximum benthic algal density to a range of inorganic nitrogen and phosphorus concentrations in the water column.

Table 5-11
Ranges of Nutrient Concentrations Reported in Ecoregion 6 Streams

Nutrient	Minimum 1st Quartile Reported (mg/L)	Maximum 4th Quartile Reported (mg/L)	Average Concentration for Impaired Stream (mg/L)
Ammonia	0.00	16.30	0.36
Nitrite	0.00	2.50	0.15
Nitrate	0.03	42.45	5.05
Total Kjeldhal Nitrogen	0.08	8.60	0.81
Phosphate	0.01	12.55	0.42
Total Phosphorus	0.10	4.42	0.29

Figure 5-64 and Figure 5-65 plot the predicted potential biomass of benthic algae as $g/m^2/d$ under nitrogen and phosphorus limitation, respectively, with the other nutrient and light at maximum (non-limiting) levels. Inorganic nitrogen and phosphorus concentrations range from zero to maximum reported in Table 5-11.

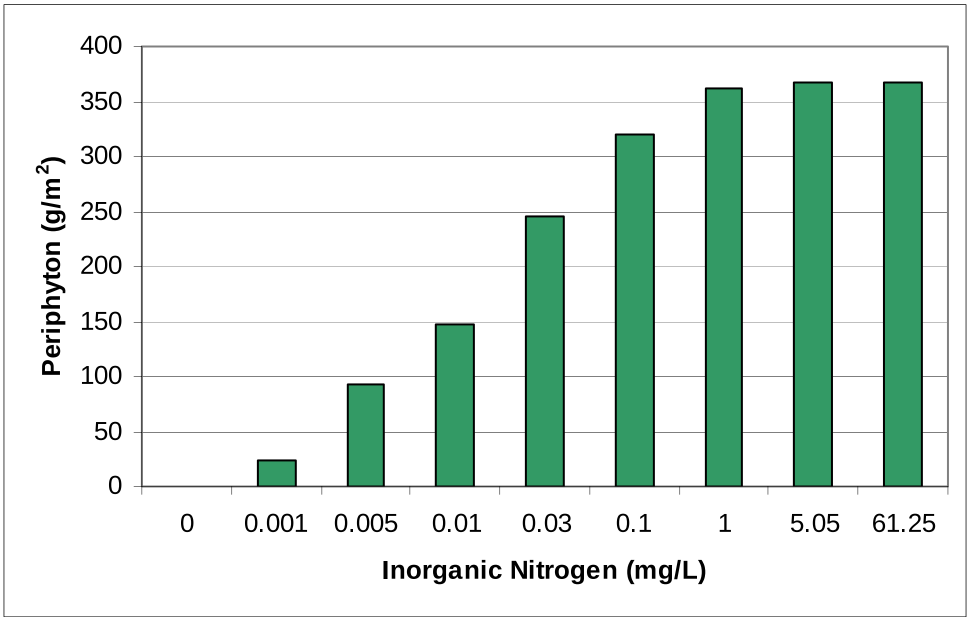


Figure 5-64 Predicted Steady-State Maximum Benthic Algal Biomass under Nitrogen Limitation with QUAL2K Default Parameters

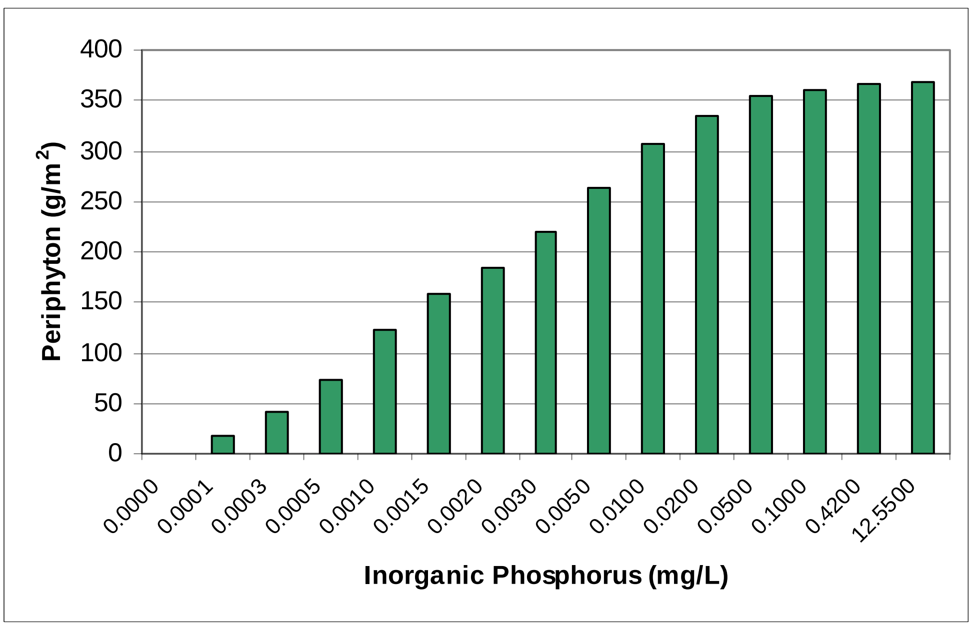


Figure 5-65 Predicted Steady-State Maximum Benthic Algal Biomass under Phosphorus Limitation with Default QUAL2K Parameters.

The maximum predicted periphyton biomass is about 360 g/m^2 AFDW, which would be equivalent to about 410 mg/m^2 chlorophyll *a* using the median ratio from the EMAP dataset. This is a concentration with no nutrient limitation, little light limitation (June clear sky insolation with relatively high water column transmission) and no losses to scour.

A response surface of the steady-state periphyton biomass versus inorganic nitrogen and inorganic phosphorus concentrations predicted by the model using default parameters is shown in Figure 5-66. Two targets often cited for benthic chlorophyll *a* are 100 and 200 mg/m^2, which translate approximately to AFDW biomass of 87.7 and 175 g/m^2 (using the median ratio from the EMAP dataset). The results with initial default parameters shown in Figure 5-66 suggest that to achieve the 100 mg/m^2 chlorophyll *a* target under conditions of no shading and no scour it would be necessary to hold inorganic phosphorus below 0.7 µg/L or inorganic nitrogen less than 5 µg/L, while to achieve the 200 mg/m^2 benthic chlorophyll *a* target it would be necessary to hold inorganic phosphorus below 2 µg/L or inorganic nitrogen below 20 µg/L. These numbers are based on poorly constrained default parameter values and are not very meaningful for criteria in real streams. As will be shown below, significantly higher criteria look to be appropriate for Ecoregion 6. Note that these results are for full light and no scour, and also that the magnitude of the results will be sensitive to the specification of the periphyton respiration and natural death rates. In ongoing work, adjustments to model parameters are being investigated to provide an approximate match to the maximum chlorophyll *a* concentration relationships reported by Dodds *et al*.

5.10.4 EFFECTS OF LIGHT LIMITATION

In most streams, light limitation reduces growth of periphyton below the theoretical maximum. This light limitation may be caused by cloudy or foggy skies, topographic shading, turbidity in the water column, or canopy closure. Going from zero to 80 percent canopy closure reduces light by 90 percent and is predicted to reduce periphyton biomass (in the absence of nutrient limitation) by about 50 percent (Figure 5-67). Other factors that reduce average light availability (including time of year) will have similar effects. However, the results predicted by the model will depend on the value assigned to the light half-saturation constant.

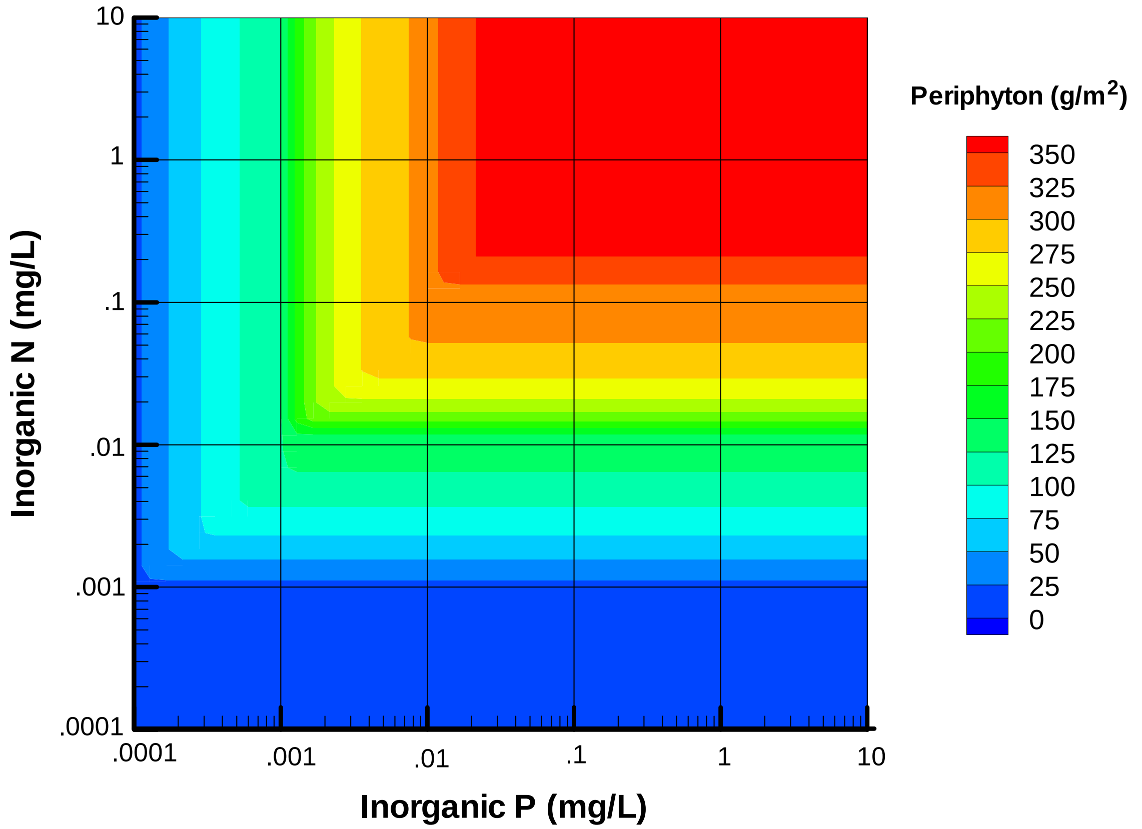


Figure 5-66 Response Surface for Maximum Periphyton Biomass (AFDW) versus Inorganic Nitrogen and Phosphorus Concentrations, QUAL2K Equations with Default Parameters

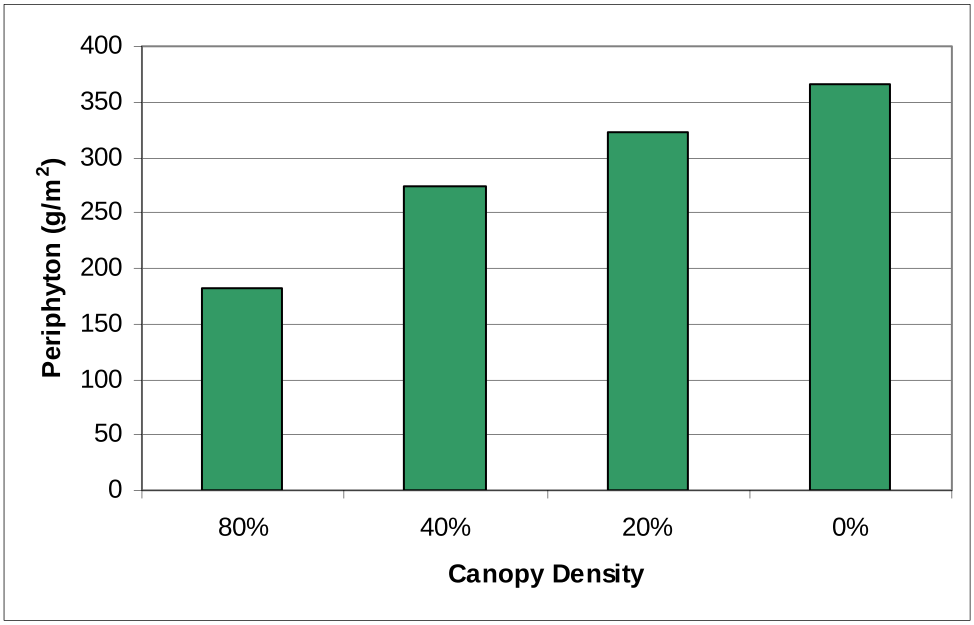


Figure 5-67 Steady State Benthic Algal Density Response to Canopy Closure without Nutrient Limitation (QUAL2K Default Parameters)

5.10.5 Reconciling the Model and Dodds' Estimates

The discussions in the previous sections relied on default kinetic parameters for QUAL2K. Many of these parameters are not well constrained or documented, and may not be appropriate for application in Ecoregion 6.

As discussed in Sections 4.10 and 4.11, Dodds *et al*. (1997, 2002) developed regression relationships to predict average and maximum benthic chlorophyll *a* concentrations in streams as a function of total nitrogen and total phosphorus concentrations. In theory, the maximum predicted using the Dodds equation should correspond to the steady-state QUAL2K prediction, without light limitation below the local theoretical maximum, converted from AFDW to chlorophyll *a*. As discussed above, the conversion to chlorophyll *a* density used the median from the California EMAP samples, or 1.14.

As QUAL2K works with inorganic fractions rather than total nitrogen and phosphorus, as used by Dodds, it is necessary to translate between total and inorganic concentrations to compare the estimates. These numbers vary widely in ecoregion 6 streams. For nitrogen, a plot of the inorganic fraction versus total nitrogen concentration shows a positive correlation (Figure 5-68). However, the correlation seems to be absent below about 1 mg/L total nitrogen. Presumably, the higher concentrations represent effluent dominated situations impacted by WWTPs with high inorganic N loads.

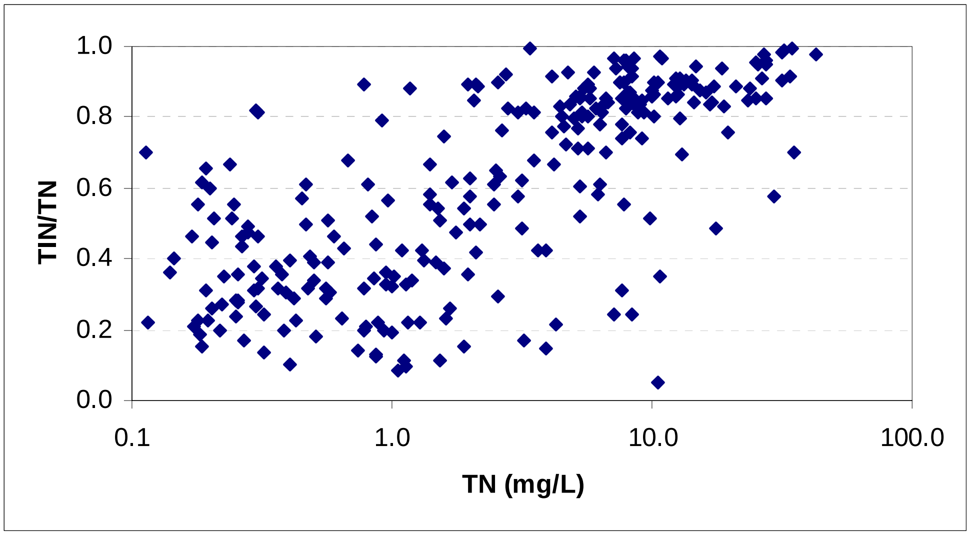


Figure 5-68 Inorganic Nitrogen Fraction vs. Total Nitrogen in Ecoregion 6 Streams

For phosphorus, there is no evident correlation between inorganic fraction and total phosphorus concentration (Figure 5-69), and observed values range from 0 to greater than 1 (not shown), reflecting dubious laboratory precision.

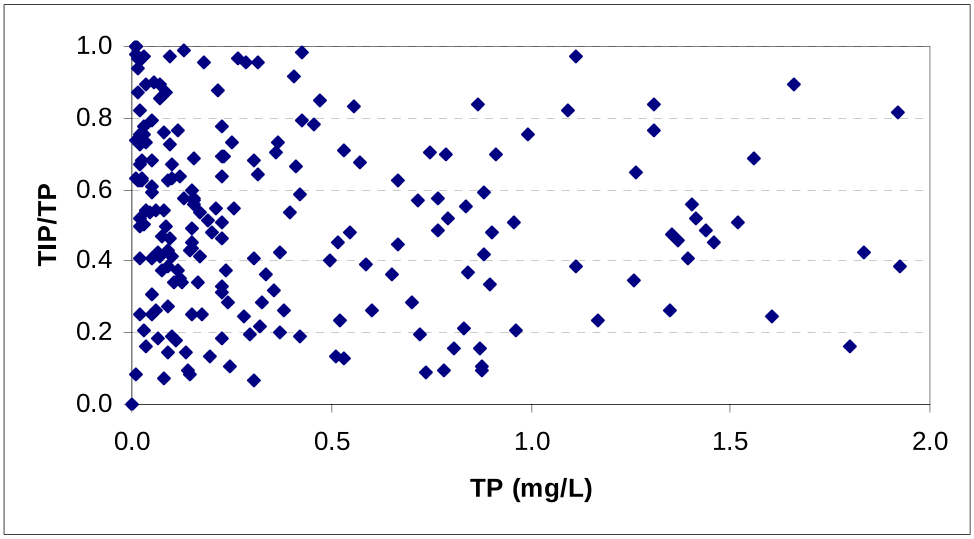
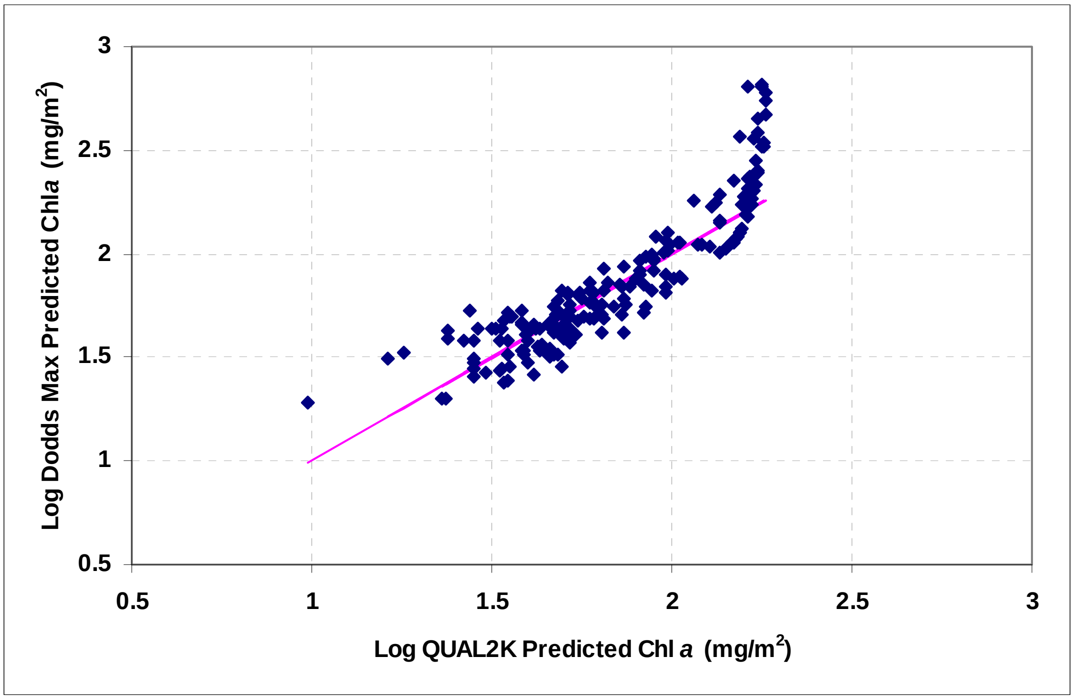


Figure 5-69 Inorganic Phosphorus Fraction vs. Total Phosphorus in Ecoregion 6 Streams

For the purpose of the analysis, it was assumed that the inorganic fraction for nitrogen could be represented as the median from sites with TN concentrations less than 1 (35 percent), while the inorganic fraction of phosphorus could be represented as the median from sites with TP concentrations less than 0.5 mg/L (63 percent). However, the large variability in actual fractions limits the applicability of a generic approach to setting total nutrient criteria based on simulation with inorganic nutrient frations.

An approximate fit between the QUAL2K steady-state results and the Dodds (2002) equation for maximum chlorophyll *a* density (as applied to the EMAP and RB3 nutrient data) was obtained by adjusting the kinetic parameters to minimize the squared difference in log space, using numerical optimization. The response of the Dodds equation appears to deviate from QUAL2K predictions above a concentration of about 250 mg-chl a/m^2, so the optimization was based on QUAL2K predictions below 250 mg/m^2 only. The resulting fit is shown in Figure5-70 and the corresponding parameter values in Table 5-12.



Note: Individual points represent observed nutrient pairs from the RB6 and EMAP California data sets.

Figure 5-70 Optimized Reconciliation of QUAL2K Steady-State Predictions to Dodds (2002) Equation for Maximum Periphyton Chlorophyll *a*

Table 5-12
QUAL2K Kinetic Parameters for Benthic Algae Adjusted to Dodds' (2002) Results

Parameter	Default Value	Optimized Value	Units
$K_{p\max}$ (maximum photosynthetic rate)	60	80.9	$g/m^2/d$
k_{rb} (respiration rate)	0.125	0.10	1/d
k_{db} (natural death rate)	0.02	0.251	1/d
k_{sNb} (nitrogen half-saturation constant)	0.015	0.0951	mg-inorganic N/L
k_{sPb} (phosphorus half-saturation constant)	0.002	0.00118	mg-inorganic P/L

The estimated nitrogen Michaelis-Menton half-saturation constant is substantially higher in the adjusted model than specified in the default parameters. This likely reflects the model assumption that the water column is completely mixed. In fact, there is likely to be a vertical gradient, particularly in the presence of an active periphyton community, with lower concentrations near the sediment-water interface. The half-saturation point relative to the averaged water column concentration will thus be higher than the true half-saturation constant relative to the water layer just above the sediment bed. The effect of this parameter change is to delay response to total nitrogen until higher concentrations are reached. On the other hand, the optimized phosphorus half-saturation constant is lower than the default, perhaps reflecting availability of phosphorus from sediment.

The revised parameters yield a model in which the maximum is 212 g/m^2 AFDW (241 mg/m^2 chlorophyll *a*) in the absence of nutrient or light limitation. This is lower than the estimate with default parameters (360 g/m^2 AFDW or 410 mg/m^2 chlorophyll *a*) because of the increase in the natural death rate relative to the default parameters. It will be recalled that the death rate used in the default set was recommended for planktonic, rather than benthic algae, and a higher value for benthic algae seems reasonable. However, the QUAL2K and Dodds predictions do diverge for higher concentrations, with the Dodds equation predicting much greater biomass above about 250 g/m^2 AFDW.

Using the median chlorophyll *a* to AFDW ratio of 1.14 from the EMAP data and the dissolved fractions of TN and TP cited above, concentrations corresponding to different benthic chlorophyll *a* targets may be generated, as shown in Table 5-13. The targets for TN are about 40 percent higher than the TN targets proposed in Dodds *et al*. (1996) to achieve similar maximum chlorophyll *a* targets in the Clark Fork River, while the TP targets are much lower. It appears likely that the TP targets derived in this way may not be realistic, as they ignore the availability of phosphorus from sediment and from luxury storage and recycling in the algal mat. It also should be noted that, even if quantitatively correct, the target concentrations derived from the QUAL2K formulation are "either/or" rather than "and" constraints. That is, the maximum chlorophyll *a* target is predicted to be met if either the N or P target is met.

Table 5-13
Nutrient Concentrations to Achieve Maximum Benthic Chlorophyll *a* Targets Estimated by Adjusted QUAL2K Equations for Ecoregion 6 (no light limitation)

Maximum Chlorophyll *a* (mg/m^2)	Corresponding Algal Biomass (g/m^2)	Total Inorganic N (μg/L)	Total Inorganic P (μg/L)	Total Nitrogen (μg/L)	Total Phosphorus (μg/L)
100	87.7	66.7	0.97	191	1.5
150	131.6	154	2.25	440	3.6
200	175.4	446	6.51	1274	10.3

5.10.6 EFFECTS OF HYDRAULIC REGIME

In higher-gradient natural streams, periphyton are periodically scoured out by high flow events. Variability in flow may also suppress periphyton growth relative to steady flowing streams in other ways, for instance by periodic drying of much of the wetted perimeter. In extensive work in New Zealand, Biggs (2000) determined that a simple, but useful statistical representation of the effects on biomass of hydrologic regime can be created based on an analysis of the mean number of days available for biomass accrual (d_0), which he defined as the average time between flood events greater than 3 times the median flow. Note that days of accrual is calculated as [1/(mean frequency of events per year >3x median flow) x 365 d), and is not the same as an actual measure of average days between flood events. Biggs' found that the frequency of high-biomass events increases greatly in response to nutrients when the average accrual period exceeds about 50 days.

Biggs' best fit regression for maximum monthly density of benthic algal biomass (mg/m^2 chlorophyll *a*) was written in terms of days of accrual and soluble inorganic nitrogen (SIN) concentration, although the parameters on d_0 are similar for regressions on accrual only and on accrual and soluble reactive phosphorus. This has the form:

$$\log_{10} B = -2.946 + 4.285 \log_{10} d_0 - 0.929 \left(\log_{10} d_0\right)^2 + 0.504 \log_{10} SIN$$

Figure 3 in Biggs (2000) shows that the response to d_0 flattens out by 200 days and may be taken as an estimate of the maximum response in the absence of scour, B_{max}. Because the equation is in logarithmic form, forming the ratio of B at a specified value of d_0 to the value of B_{max} results in cancellation of the constant and SIN terms. This yields an expression for the reduction in maximum monthly biomass as a function of days of accrual:

$$\log_{10}\left(B(d_0) \big/ B_{max} \right) = 4.285 \cdot (\log_{10} d_0 - 2.301) - 0.929 \cdot \left([\log_{10} d_0]^2 - 5.295 \right)$$

This results in a factor on the maximum potential periphytic algal biomass as a function of days of accrual as shown in Figure 5-71.

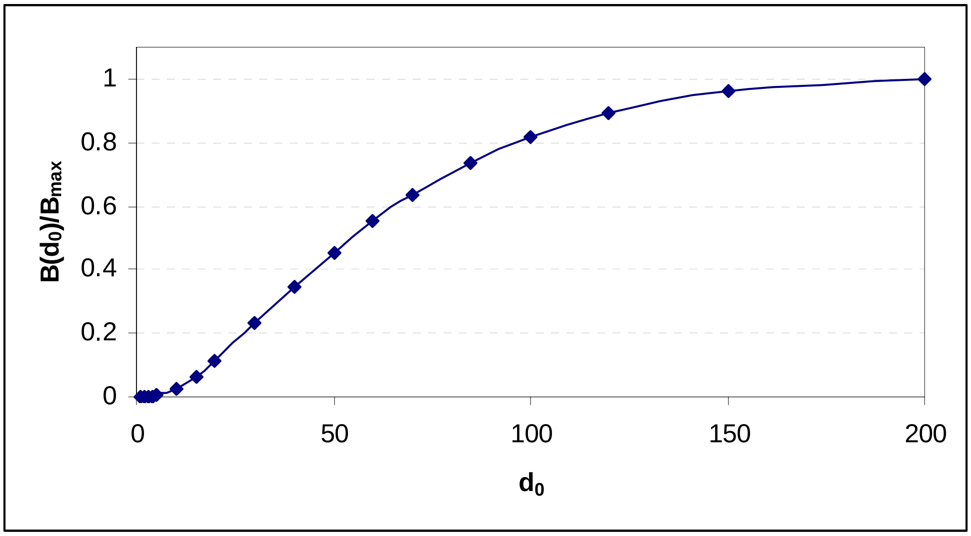


Figure 5-71 Fraction of Potential Maximum Periphyton Biomass Expected as a Function of Days of Accrual

The exact parameter values for this relationship are specific to the New Zealand streams studied by Biggs, and may not be applicable to California. However, data are not available to develop a similar analysis for periphyton response in California streams, and it is reasonable to expect that a similar relationship exists. Note that a 50-day or less accrual period is sufficient to reduce the density by more than half (using Biggs' regression).

Variability in the hydraulic regime is likely to have a large effect on whether a particular biomass target is attained. For example 30 or less days of accrual would reduce the average monthly maximum biomass to 23 percent of the theoretical maximum, using the Biggs (2000) equation. Under these conditions, the biomass would not be predicted to exceed a 100 mg/m^2 chlorophyll *a* target using the default QUAL2K parameters.

The Mediterranean climate of southern California experiences long dry periods during the summer. Intuitively, this would suggest that the value of d_0 should be large for streams in this area, and the effects on biomass small. This is not, however, necessarily the case, as d_0 is defined on the basis of frequency relative to the median flow, and the California climate also suppresses the median flow value. In fact, d_0 should be thought of as a measure of flow variability rather than as a direct index of flood frequency. This can be seen through an example application to the 1969-1994 flow record for the upper Santa Margarita River at Temecula, CA (USGS gage 11044000).

Flow in the Santa Margarita at this site is flashy, but is perennial in character, being supported by groundwater discharge (Figure 5-72). Like most California rivers, flows in the Santa Margarita are also affected by upstream impoundments, return flow from irrigation using imported water, and wastewater discharges.

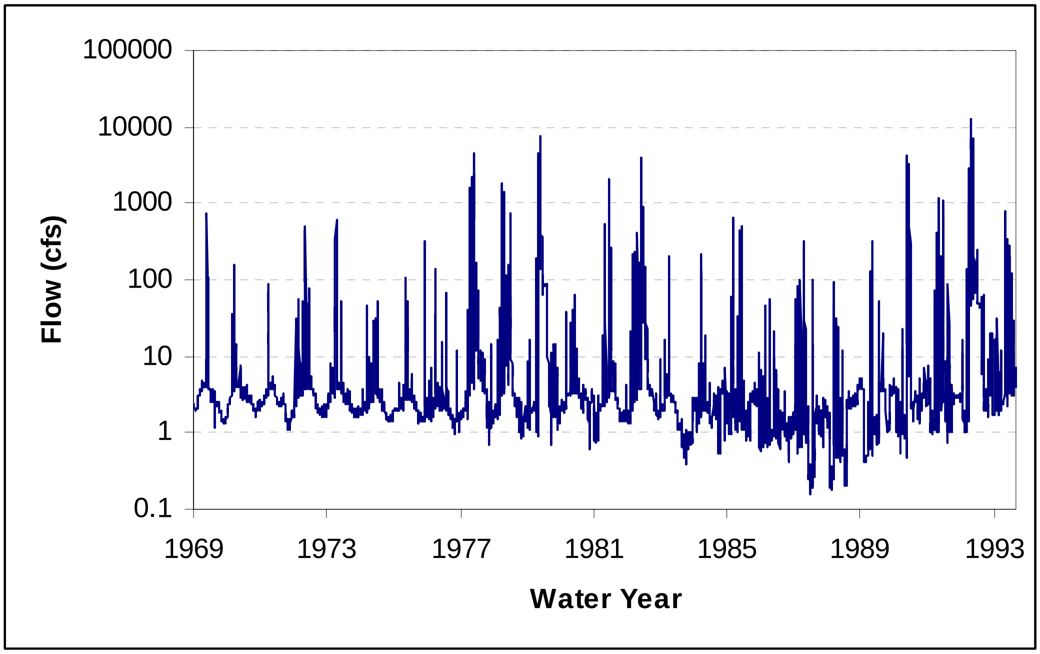


Figure 5-72 Daily Flows in the Santa Margarita River at Temecula, CA

Large flow events in the Santa Margarita typically occur for brief periods in the winter rainy season and are uncommon in the summer. For example, flows of 100 cfs or greater occur on less than 3 percent of days, and there are often greater than 200 days between flows of this magnitude. However, this does not mean that the value of d_0 is large.

The median flow in the Santa Margarita is 2.5 cfs, and 3 times the median flow is only 7.5 cfs, so flows greater than 3 times the median are actually fairly common. Over the period of record, 12.3 percent of flows exceed 7.5 cfs. The value of d_0 calculated as recommended by Biggs is thus only 8.13 days. This estimate is, however, strongly influenced by the flow regime in a few wet years. If, alternatively, the value of d_0 is calculated for each individual year and then averaged the estimate is 22.7 days.

In either case, the Santa Margarita example shows that the estimate of Biggs' accrual index, d_0, may be quite low for California streams – not because large floods occur frequently but because of variability about a naturally low median flow. If the Biggs relationship is valid for these types of streams this would imply a significant reduction in the maximum potential algal biomass as a function of flow variability. Another possibility is that d_0 should be redefined as a higher multiple of median flow for the California climate than is appropriate for New Zealand. Testing of the relationship is needed for California streams, but is not possible with the data currently available.

5.10.7 DISCUSSION OF PERIPHYTON MODEL RESULTS

Attempts documented in the literature to derive nutrient targets to control benthic algal biomass in streams have generally met with limited success. In essence, nutrient concentrations alone are not particularly good predictors of benthic algal biomass because there are other important controlling factors – most notably light availability and hydraulic regime. Nutrients, however, should establish an upper bound on the potential benthic algal growth that could occur in a stream in the absence of other limitations – although water column concentrations of nutrients alone are not necessarily a good indicator of the nutrients available for growth, as periphyton often obtain nutrients from the sediment or store excess nutrients within their biomass.

The analytical and empirical tools discussed above provide a structured means to assess the interaction of these many parameters. However, the problem is of sufficient complexity, with many of the parameter values poorly constrained, that the modeling analysis alone cannot be relied upon to give reasonable nutrient criteria without either site-specific calibration or cross-sectional calibration. It may, however, be useful as one among a number of lines of evidence in the evaluation of stream nutrient criteria.

5.10.8 TEST APPLICATION TO RWQCB MONITORING DATA

The large monitoring data set collected by RWQCB 3 provides an excellent basis for investigating the ranking of stream eutrophication risk using the periphyton modeling approach described above.

5.10.8.1 Nutrient-based Analysis

The first application to the data was based on nutrient limitation only, without consideration of additional light limitation due to shading and turbidity. This yields an indication of potential risk from nutrients alone. To apply this method to the RWQCB3 data, we extracted average inorganic N and P concentrations for each monitoring site from the RWQCB3 database. The application yields reasonable potential maximum benthic algal response on the assumption of minimal light limitation (the calculations were done assuming no shading, typical summer insolation of 690 cal/cm^2/d, a depth of 0.5 m, and an extinction coefficient of 1 m^{-1}). Further consideration of the potential role of light limitation is provided in the second part of the memo.

QUAL2K estimates of benthic biomass are as AFDW. There is not a direct translation to the more common metric of mg/m^2 benthic chlorophyll *a*. However, information from the California EMAP data set provides a median for the ratio of

AFDW (g/m^2) to benthic chlorophyll *a* (mg/m^2) of 36/41 – with plentiful scatter. Using this ratio, the oligotrophic-mesotrophic boundary of 60 mg/m^2 chlorophyll *a* recommended by Dodds *et al*. (1998) would be equivalent to 53 g/m^2 AFDW, and mesotrophic-eutrophic boundary of 200 mg/m^2 chlorophyll *a* would be equivalent to 175 g/m^2 AFDW.

A total of 146 sites from the RWQCB database were ranked, as shown in Figure5-73. The figure displays the cumulative distribution of sites, with the approximate trophic ranges superimposed. 56 percent of the sites are rated eutrophic by this method. However, the results are only approximate as they do not take light limitation into account – thus the actual maximum in many of these streams will generally be lower.

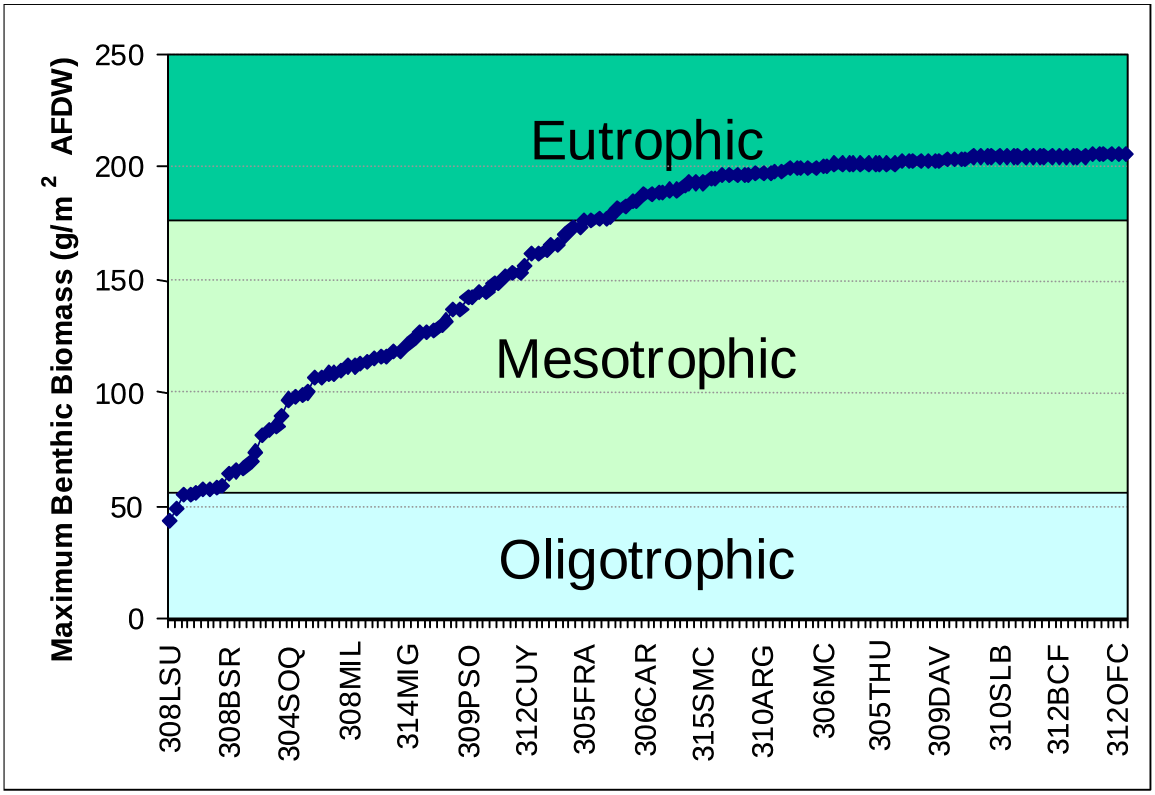


Figure 5-73 Cumulative Distribution of Maximum Potential Benthic Algal Biomass Predicted by QUAL2K Approach Applied to RWQCB 3 Monitoring Sites in the Absence of Light Limitation

5.10.8.2 Evaluation of Light Limitation

The RB3 database provides information on percent shading and turbidity for 125 out of the 146 available sites. Both shade and turbidity reduce the potential for algal growth.

Stream shading reduces the available incident light at the water surface (*I*). A graphical relationship (USACE, 1956) shows the general form of this reduction can be approximated as

$$I / I_0 = e^{-3.0\,Shade},$$

where *Shade* is the fractional amount of the water surface that is shaded. This relationship was developed for conifers in the northwest, which likely provide much greater shading per fraction of sky covered than vegetation in southern California. For an approximate analysis, the factor on *Shade* was reduced by half, to 1.5.

Turbidity affects the light extinction coefficient in the water column. The relationship is not exact or constant, however, and varies with the type of matter that is causing the turbidity. Lacking a general relationship between turbidity and light extinction, we have used a relationship developed for a large data set in the Upper Mississippi, which estimates the extinction coefficient (m^{-1}) as

$$K_e = 0.655 \cdot Turb^{0.5325},$$

where, *Turb* is the nephelometric turbidity (NTU).

Light extinction also depends on the depth of the water column. This information is not available for the RB3 sites, and an assumption of 0.5 m depth was again used. Given the lack of information on depth and the use of approximate relationships for shading and turbidity, only relative estimates can be developed at this time.

Light limitation is of high importance to the nutrient response of RB 3 streams. Canopy shading up to 100 percent is present at several sites, and observed turbidity measurements at some sites exceed 2,500 NTU. Average reported values at each site were used for this analysis; however, the turbidity estimates for small data sets can be highly biased depending on the type of weather events that have been sampled. This analysis, like the previous, also does not account for scour effects.

Application of the analysis with light limitation included yields very different results from the previous analysis, as shown in Figure 5-74. With light limitation included, only 8 percent of sites are rated eutrophic, versus 56 percent without consideration of light limitation.

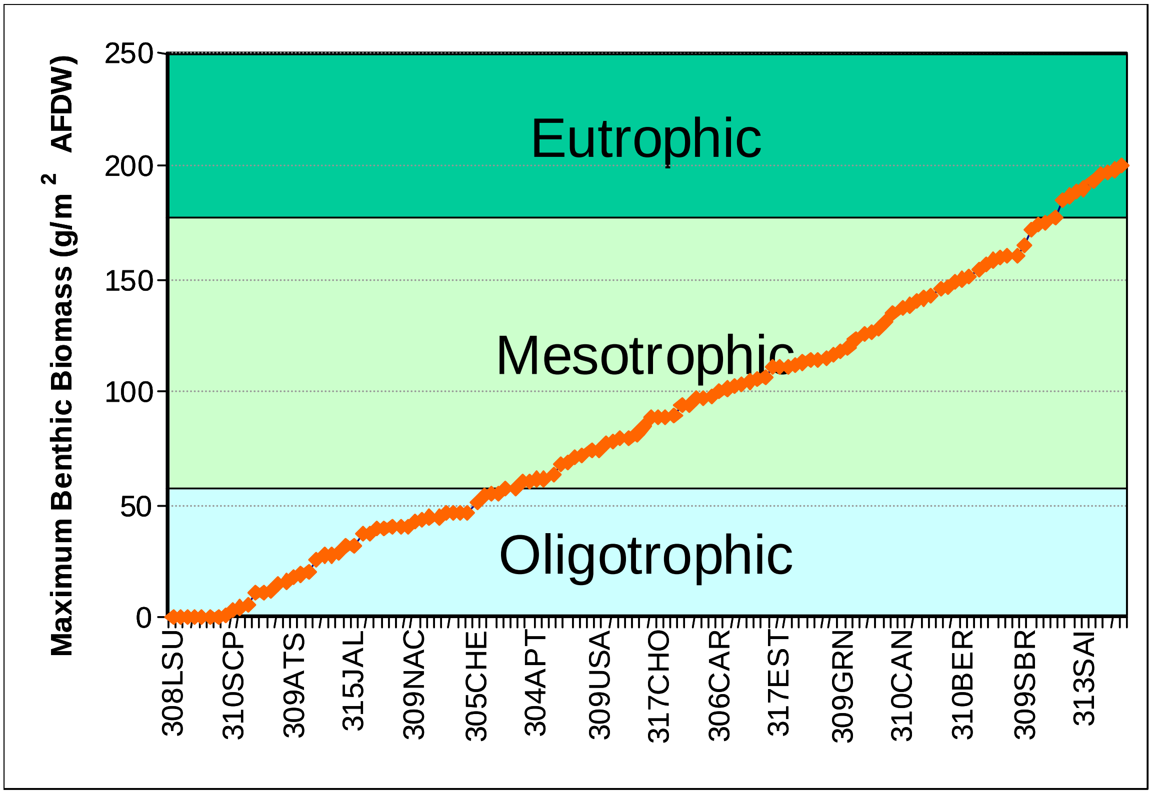


Figure 5-74 Cumulative Distribution of Potential Benthic Algal Biomass Predicted by QUAL2K Approach Applied to RWQCB 3 Monitoring Sites with Light Limitation

Figure 5-75 shows the effect of light limitations on individual sites. At many of the sites with a high potential for growth due to nutrient concentrations, a high degree of shading and high turbidity are likely to suppress most growth of benthic algae. For example, at site 305PAJ the predicted biomass without light limitation is 203 g/m^2, but only 12 g/m^2 with light limitation. Average turbidity at this site was 136 NTU, with 50 percent shading.

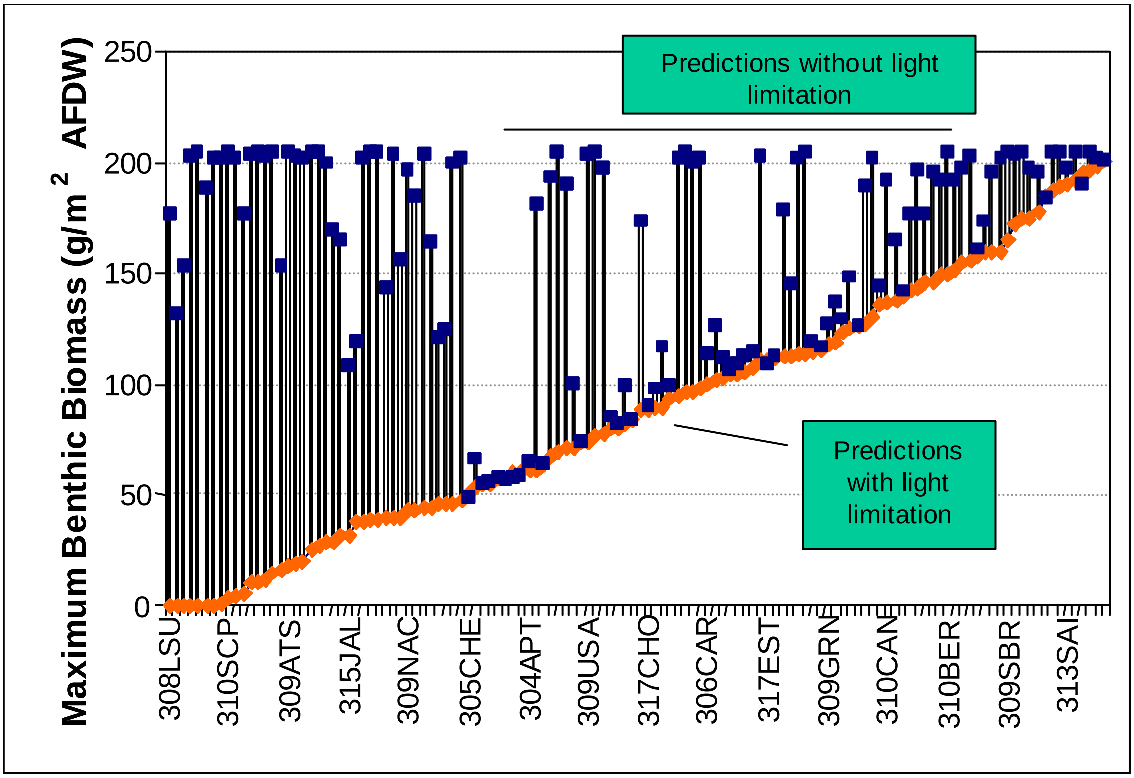


Figure 5-75 Effect of Light Limitation on Predictions

The distributions of light and nutrient limitation in the RB3 dataset is shown in Figure 5-76 (both measures range from 0 to 1, where a value of 1 indicates no limitation). In the upper right hand of the diagram are sites with high light availability and high nutrients. These are sites most at risk for excess benthic algal growth. The upper left of the diagram contains sites with elevated nutrients, but where sufficient light is not available to support benthic algal growth, due to canopy closure and/or turbidity. Finally, the lower right corner of the diagram contains sites that have ample light, but where algal growth is limited by nutrient availability. Relatively few sites are limited by both nutrients and light, probably due to a correlation between turbidity and nutrient load.

Results of the analysis with light limitation are provided in Table 5-15. As with Table 5-14, these should be treated as relative, rather than absolute estimates due to the many uncertainties still present in the analysis.

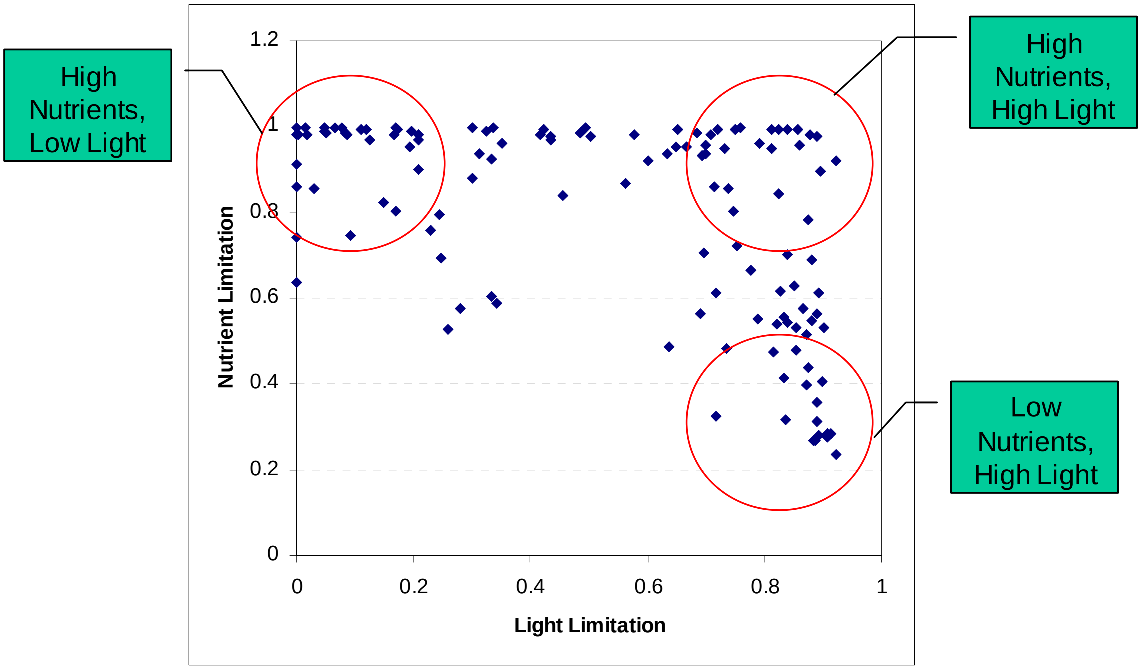


Figure 5-76 Nutrient and Light Limitation in RB 3 Data Set

Table 5-14
Results of QUAL2K Method Application to RWQCB 3 Sites without Evaluation of Light Limitation

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
308LSU	42.98	304APT	145.01
307CMD	48.61	304LOR	148.60
307CML	54.76	304SOQ	97.56
308LSR	55.19	305CHE	127.28
309SET	55.44	305CHI	202.33
309SEC	56.52	305COR	201.44
308BSU	57.44	305FRA	176.22
307CMU	58.23	305HOL	178.46
310SCP	58.79	305LLA	201.47
308BSR	64.07	305LUC	192.01
308SJC	64.90	305MON	173.05
310COO	66.52	305MUR	202.19
310TOR	69.75	305OAK	169.37
308WLO	73.84	305PAC	199.59
310SSU	81.56	305PAJ	202.78
307CMN	83.72	305PES	189.66
309ATS	84.97	305SAN	187.60
310PCO	90.14	305TES	199.08
304SOQ	97.56	305THU	201.71
315GAV	98.31	305TRE	188.86
308BGC	99.15	305UVA	190.33
307TUL	99.95	305VIS	181.10
310SLC	106.26	306CAR	187.28
310OLD	106.41	306ELK	161.03
315JAL	108.71	306MCM	200.30

Table 5-14
Results of QUAL2K Method Application to RWQCB 3 Sites without Evaluation of Light Limitation

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
317ESE	109.14	306MOR	182.01
312SIV	109.59	307CMD	48.61
308MIL	111.41	307CML	54.76
312BRE	112.04	307CMN	83.72
310SLM	112.64	307CMU	58.23
315CAU	113.61	307TUL	99.95
308GAR	114.55	308BGC	99.15
309NAC	116.11	308BSR	64.07
310ADC	116.27	308BSU	57.44
314SYC	118.40	308GAR	114.55
310SRO	118.56	308LIM	126.38
314MIG	121.24	308LSR	55.19
314SYL	124.44	308LSU	42.98
308LIM	126.38	308MIL	111.41
309SAN	126.39	308SJC	64.90
305CHE	127.28	308WLO	73.84
312SIS	129.54	309ALD	203.76
312CCC	131.45	309ALU	204.68
315GAI	136.83	309ATS	84.97
310SRU	137.34	309AXX	202.59
309PSO	141.91	309DAV	203.04
314SYI	142.85	309DSA	144.29
309DSA	144.29	309GAB	201.66
304APT	145.01	309GRN	196.50
304LOR	148.60	309KNG	184.98
310CAY	148.72	309LOK	176.84

Table 5-14
Results of QUAL2K Method Application to RWQCB 3 Sites without Evaluation of Light Limitation

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
310VIA	151.44	309LOR	152.84
309LOR	152.84	309NAC	116.11
312CUT	153.34	309OLD	204.49
312CUY	155.80	309POT	204.55
306ELK	161.03	309PSO	141.91
309USA	161.23	309QUA	205.22
314SAL	163.47	309SAC	197.99
310PIS	165.13	309SAN	126.39
312SBC	165.25	309SAT	184.16
305OAK	169.37	309SBR	202.16
305MON	173.05	309SDR	205.25
309SUN	173.31	309SEC	56.52
305FRA	176.22	309SET	55.44
317CHO	176.28	309SUN	173.31
309UQA	176.75	309TOP	195.97
309LOK	176.84	309UAL	205.01
305HOL	178.46	309UQA	176.75
305VIS	181.10	309USA	161.23
306MOR	182.01	310ADC	116.27
309SAT	184.16	310AGB	195.22
309KNG	184.98	310AGF	195.89
306CAR	187.28	310AGS	197.63
305SAN	187.60	310ARG	197.38
305TRE	188.86	310BER	201.66
313SAC	189.16	310CAN	199.55
305PES	189.66	310CAY	148.72

Table 5-14
Results of QUAL2K Method Application to RWQCB 3 Sites without Evaluation of Light Limitation

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
305UVA	190.33	310COO	66.52
305LUC	192.01	310MOR	198.86
312NIP	192.53	310OLD	106.41
317EST	192.71	310PCO	90.14
315SMC	192.88	310PIS	165.13
310SCN	195.06	310PRE	204.43
310AGB	195.22	310SCN	195.06
310AGF	195.89	310SCP	58.79
309TOP	195.97	310SLB	204.49
310TWB	196.27	310SLC	106.26
311SLE	196.49	310SLM	112.64
309GRN	196.50	310SLV	204.78
315RSB	197.04	310SRO	118.56
310ARG	197.38	310SRU	137.34
310AGS	197.63	310SSC	200.99
309SAC	197.99	310SSU	81.56
313SAE	198.01	310TOR	69.75
310MOR	198.86	310TUR	200.90
305TES	199.08	310TWB	196.27
310CAN	199.55	310VIA	151.44
305PAC	199.59	311SLE	196.49
315RIN	199.88	311SLN	204.87
306MCM	200.30	312BCD	201.69
310TUR	200.90	312BCF	204.93
310SSC	200.99	312BCU	203.49
305COR	201.44	312BRE	112.04

Table 5-14
Results of QUAL2K Method Application to RWQCB 3 Sites without Evaluation of Light Limitation

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
305LLA	201.47	312CCC	131.45
310BER	201.66	312CUT	153.34
309GAB	201.66	312CUY	155.80
312BCD	201.69	312MSD	204.90
312ORB	201.69	312NIP	192.53
305THU	201.71	312NIT	202.12
315CRP	201.81	312OFC	205.29
313SAB	201.94	312OFL	204.84
312NIT	202.12	312OFN	204.23
309SBR	202.16	312OLA	204.78
305MUR	202.19	312ORB	201.69
305CHI	202.33	312ORC	205.14
309AXX	202.59	312ORI	205.32
305PAJ	202.78	312SAL	204.74
309DAV	203.04	312SBC	165.25
314SYN	203.17	312SIS	129.54
312BCU	203.49	312SIV	109.59
313SAI	203.63	312SMA	205.07
309ALD	203.76	312SMI	205.26
314SYF	204.17	313SAB	201.94
312OFN	204.23	313SAC	189.16
310PRE	204.43	313SAE	198.01
309OLD	204.49	313SAI	203.63
310SLB	204.49	314MIG	121.24
309POT	204.55	314SAL	163.47
309ALU	204.68	314SYC	118.40

Table 5-14
Results of QUAL2K Method Application to RWQCB 3 Sites without Evaluation of Light Limitation

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
312SAL	204.74	314SYF	204.17
312OLA	204.78	314SYI	142.85
310SLV	204.78	314SYL	124.44
312OFL	204.84	314SYN	203.17
311SLN	204.87	315APC	205.28
312MSD	204.90	315CAU	113.61
312BCF	204.93	315CRP	201.81
309UAL	205.01	315FRC	205.08
312SMA	205.07	315GAI	136.83
315FRC	205.08	315GAV	98.31
312ORC	205.14	315JAL	108.71
309QUA	205.22	315RIN	199.88
309SDR	205.25	315RSB	197.04
312SMI	205.26	315SMC	192.88
315APC	205.28	317CHO	176.28
312OFC	205.29	317ESE	109.14
312ORI	205.32	317EST	192.71

Table 5-15
Results of QUAL2K Method Application to RWQCB 3 Sites with Light Limitation Included

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
309LOK	0.00	304APT	112.87
312CCC	0.00	304LOR	125.33
309LOR	0.00	304SOQ	89.00
305CHI	0.05	305CHE	118.09
309QUA	0.06	305CHI	0.05
305SAN	0.07	305COR	97.75
305MUR	0.12	305FRA	5.77
309GAB	0.54	305HOL	112.33
312BCF	3.32	305LLA	113.42
305THU	4.30	305LUC	148.86
305FRA	5.77	305MON	88.13
312BCU	10.55	305MUR	0.12
312OFC	10.96	305OAK	28.13
305PAJ	11.75	305PAC	97.15
312ORI	14.79	305PAJ	11.75
312CUT	15.91	305PES	195.72
312ORC	17.95	305SAN	0.07
309AXX	19.13	305TES	46.60
312BCD	19.82	305THU	4.30
309ALU	25.14	305UVA	71.13
309OLD	27.34	305VIS	60.95
315RIN	27.92	307CMD	50.23
305OAK	28.13	307CML	54.36
312SBC	31.62	307CMN	84.18
315JAL	31.66	307CMU	59.65

Table 5-15
Results of QUAL2K Method Application to RWQCB 3 Sites with Light Limitation Included

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
310SRO	37.30	307TUL	71.29
313SAB	37.39	308BGC	81.73
309UAL	38.95	308BSR	63.77
312SMA	39.07	308BSU	57.32
314SYI	39.70	308GAR	106.95
314SYF	39.71	308LIM	101.59
312CUY	40.00	308MIL	102.54
310TWB	42.69	308SJC	60.71
309KNG	43.25	308WLO	73.49
313SAI	44.46	309ALD	74.00
314SAL	44.56	309ALU	25.14
314MIG	46.38	309ATS	79.41
314SYL	46.57	309AXX	19.13
305TES	46.60	309DAV	155.92
315CRP	46.77	309DSA	135.50
307CMD	50.23	309GAB	0.54
310COO	53.51	309GRN	142.61
307CML	54.36	309KNG	43.25
309SET	54.81	309LOK	0.00
308BSU	57.32	309LOR	0.00
309SEC	57.36	309NAC	89.77
307CMU	59.65	309OLD	27.34
310SCP	59.72	309PSO	140.08
308SJC	60.71	309QUA	0.06
305VIS	60.95	309SAC	78.11
308BSR	63.77	309SAN	126.31

Table 5-15
Results of QUAL2K Method Application to RWQCB 3 Sites with Light Limitation Included

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
315SMC	67.25	309SAT	184.92
315APC	68.82	309SBR	130.60
305UVA	71.13	309SDR	174.09
307TUL	71.29	309SEC	57.36
308WLO	73.49	309SET	54.81
309ALD	74.00	309SUN	159.80
312SMI	77.14	309UAL	38.95
309SAC	78.11	309UQA	141.53
309ATS	79.41	309USA	158.08
310SSU	79.59	310ADC	115.80
308BGC	81.73	310AGB	177.73
307CMN	84.18	310AGF	146.24
305MON	88.13	310AGS	174.87
310PCO	88.28	310ARG	190.25
304SOQ	89.00	310BER	198.04
309NAC	89.77	310COO	53.51
315GAV	93.89	310PCO	88.28
312ORB	94.35	310PIS	138.09
312MSD	96.74	310PRE	164.70
305PAC	97.15	310SCN	159.92
305COR	97.75	310SCP	59.72
315CAU	100.14	310SLB	188.73
308LIM	101.59	310SLC	103.90
308MIL	102.54	310SLM	111.15
310SLC	103.90	310SLV	192.69
317ESE	104.22	310SRO	37.30

Table 5-15
Results of QUAL2K Method Application to RWQCB 3 Sites with Light Limitation Included

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
312BRE	105.26	310SSC	200.30
308GAR	106.95	310SSU	79.59
314SYN	110.32	310TWB	42.69
312SIV	110.76	311SLN	196.83
310SLM	111.15	312BCD	19.82
305HOL	112.33	312BCF	3.32
304APT	112.87	312BCU	10.55
305LLA	113.42	312BRE	105.26
315FRC	113.52	312CCC	0.00
314SYC	114.72	312CUT	15.91
310ADC	115.80	312CUY	40.00
305CHE	118.09	312MSD	96.74
315GAI	119.03	312NIP	136.61
312SIS	123.31	312NIT	159.95
304LOR	125.33	312OFC	10.96
309SAN	126.31	312OFL	186.49
313SAC	127.56	312OFN	171.79
309SBR	130.60	312OLA	149.49
309DSA	135.50	312ORB	94.35
312NIP	136.61	312ORC	17.95
310PIS	138.09	312ORI	14.79
309PSO	140.08	312SBC	31.62
309UQA	141.53	312SIS	123.31
309GRN	142.61	312SIV	110.76
317CHO	145.80	312SMA	39.07
310AGF	146.24	312SMI	77.14

Table 5-15
Results of QUAL2K Method Application to RWQCB 3 Sites with Light Limitation Included

Station	Maximum Potential Biomass (g/m^2 AFDW)	Station	Maximum Potential Biomass (g/m^2 AFDW)
305LUC	148.86	313SAB	37.39
312OLA	149.49	313SAC	127.56
317EST	150.86	313SAI	44.46
315RSB	154.05	314MIG	46.38
309DAV	155.92	314SAL	44.56
309USA	158.08	314SYC	114.72
309SUN	159.80	314SYF	39.71
310SCN	159.92	314SYI	39.70
312NIT	159.95	314SYL	46.57
310PRE	164.70	314SYN	110.32
312OFN	171.79	315APC	68.82
309SDR	174.09	315CAU	100.14
310AGS	174.87	315CRP	46.77
310AGB	177.73	315FRC	113.52
309SAT	184.92	315GAI	119.03
312OFL	186.49	315GAV	93.89
310SLB	188.73	315JAL	31.66
310ARG	190.25	315RIN	27.92
310SLV	192.69	315RSB	154.05
305PES	195.72	315SMC	67.25
311SLN	196.83	317CHO	145.80
310BER	198.04	317ESE	104.22
310SSC	200.30	317EST	150.86

6.0 Draft Tiered Criteria for Streams and Lakes in Ecoregion 6

From the standpoint of regulatory concern, nutrients matter when they produce a direct or indirect biological response (such as excess algal productivity or depressed dissolved oxygen levels), which in turn impairs a particular beneficial use for a water body of interest. As discussed in Chapter 2 in conjunction with conceptual models of nutrient impairment, different beneficial uses have different pathways of impact and different degrees of sensitivity to nutrient-induced biological changes. Thus, a relatively direct effect may be elevated nutrients causing unaesthetic algal blooms, which impair the recreational use of a water body. A more subtle effect may be nutrient-induced changes at the base of the food-web which impairs the wildlife habitat beneficial use. With respect to nutrient sensitivity, it is generally understood that a cold-water fishery is far more likely to be adversely affected by elevated algal biomass in planktonic and periphytic forms than a warm-water fishery.

The long-term objective of the nutrient criteria development effort undertaken in California is to identify nutrient levels and selected direct biological responses (such as chlorophyll levels and dissolved oxygen) **appropriate for each beneficial use**. Where multiple beneficial uses in a single water body may be affected by nutrients, the most sensitive use will determine the allowable nutrient levels. This task is complicated by the absence of co-located information on beneficial uses, nutrient levels, and direct biological responses. However, the collection of such information, either through monitoring, or through interviews with regional experts, is a critical step in the development of scientifically defensible nutrient criteria.

To illustrate the eventual form of the criteria, we propose ranges of numbers for the tiered nutrient criteria for the cold-water beneficial use for streams and lakes in Ecoregion 6, using the combination of approaches presented in the previous sections, i.e., data analysis, modeling, and review of published literature, and using our best professional judgment. Values are specified for the following exposure variables:

total nitrogen, total phosphorus, nitrate, and ammonia. Ammonia values are related to EPA toxicity criteria. Values are also proposed for the following response variables: benthic and planktonic chlorophyll a. These numbers are a first draft, based on the analyses presented in the preceding chapters and must be re-evaluated in light on any new information that is collected in the future. In preparing these initial estimates, it is anticipated that sub-regional variations within an ecoregion will be addressed through additional investigation only when water bodies are initially classified into Tier II.

As presented, these numbers are preliminary and not intended for use in a regulatory setting. The main point of including specific numeric criteria in this report is to provide a tangible guide to the proposed form of the nutrient criteria we are developing. It is possible that, in future development work, either by us, or by the Regional Water Boards, specific numeric values in Tables 6-1 and 6-2 may change; however, the structure of the tables is not expected to change.

Table 6-1

Draft nutrient criteria for protecting the cold water beneficial use in streams

Parameter	Tier I Range	Tier II Range	Tier III Range	Rationale
Exposure variables				
Total Nitrogen (TN) (mg/l)	<0.5	0.5-2.0	>2.0	Background concentrations through modeling; data from Ecoregion 6; stream benthic chlorophyll model
Total Phosphorus (TP) (mg/l)	<0.05	0.05-0.2	>0.2	Background concentrations through modeling; Land-use nutrient relationships from Ecoregion 6; stream benthic chlorophyll model
Nitrate (NO_3) (mg/l)	<0.2	0.2-2.0	>2.0	Ecoregion 6 data; Land-use nutrient relationships; background concentrations in modeling; drinking water criteria
Ammonia (NH_3) (mg/l)	<0.5[1]	0.5-2.0[2]	>2.0	Ecoregion 6 data and CCC and CMC for ammonia[1,2]
Response Variables				
Benthic Chlorophyll a (mg/m^2)	<50	50-200	>200	Literature sources; Data from Regional Water Board 6
Planktonic Chlorophyll a ($\mu g/l$)	<10	10-30	>30	Literature sources

[1] The criterion continuous concentration (CCC) for ammonia at pH 8.5 and 26°C when early life stages of fish are present is selected as the Tier I/II boundary (value = 0.52 mg/l). Source: EPA 1999 Update of Ambient Water Quality Criteria for Ammonia.
[2] The criterion maximum concentration (CMC) for ammonia when salmonids are present is 5.62 mg/l at pH 8; because this is higher than the total nitrogen value, the total nitrogen concentration is proposed as the Tier II/III boundary. Source: EPA 1999 Update of Ambient Water Quality Criteria for Ammonia.

Table 6-2

Draft nutrient criteria for protecting the cold water beneficial use in lakes

Parameter	Tier I Range	Tier II Range	Tier III Range	Rationale
Exposure variables				
Total Nitrogen (TN) (mg/l)	<0.3	0.5-1.0	>1.0	Background concentrations through modeling; data from Ecoregion 6
Total Phosphorus (TP) (mg/l)	<0.03	0.05-0.1	>0.1	Background concentrations through modeling; data from Ecoregion 6
Nitrate (NO_3) (mg/l)	<0.1	0.1-1.0	>1.0	Ecoregion 6 data; drinking water criteria
Ammonia (NH_3) (mg/l)	<0.5[1]	0.05-2.0[2]	>2.0[2]	Ecoregion 6 data and CCC and CMC for ammonia[1,2]
Response Variable				
Planktonic Chlorophyll a (μg/l)	<10	10-30	>30	Literature sources

[1] The criterion continuous concentration (CCC) for ammonia at pH 8.5 and 26°C when early life stages of fish are present is selected as the Tier I/II boundary (value = 0.52 mg/l). Source: EPA 1999 Update of Ambient Water Quality Criteria for Ammonia.
[2] The criterion maximum concentration (CMC) for ammonia when salmonids are present is 5.62 mg/l at pH 8; because this is higher than the total nitrogen value, the total nitrogen concentration is proposed as the Tier II/III boundary. Source: EPA 1999 Update of Ambient Water Quality Criteria for Ammonia.r

7.0 HOW NUTRIENT CRITERIA AND TMDLS ARE RELATED

There are substantial areas of overlap between the technical requirements / needs of a TMDL and the development of regional nutrient criteria. Because numeric criteria are often used as endpoints in TMDL analyses, the results of this study have an important bearing on future nutrient TMDLs that will be conducted in the state. There are also substantial distinctions that must be made between the two. In this section, we describe how the two processes are related, and lay out a roadmap for ongoing and future work on the criteria development efforts.

7.1 APPROACH

As a first and critical step, it is proposed in this study that nutrient criteria not be defined solely in terms of the concentrations of various nitrogen and phosphorus species, but also include consideration of primary biological responses to nutrients. It is these biological responses that correlate to support or impairment of uses. It is proposed that the consideration of biological responses be **in addition to** chemical concentrations in the final form of the nutrient criteria. Further, the development of chemical concentration criteria should be closely linked to the evaluation of biological responses.

Although the definition of the term primary biological responses can be somewhat arbitrary, for the purpose of this discussion, it includes measurements such as chlorophyll *a* (Chl *a*) in the plankton and periphyton, dissolved oxygen (DO), and benthic indices of biological integrity (IBI). Other choices for biological response may also be included as and when suitable data become available. The basis for this approach is explored more thoroughly in Chapter 2, where it is shown that elevated nutrient concentrations and impairment of beneficial uses consists of multiple interactions between different components of the aquatic ecosystem, and nutrient concentrations alone cannot be used to predict the likelihood of impairment. For example, a stream with elevated nutrient concentrations may not exhibit excessive algal growth and consequent impairment if it has a good canopy cover. If criteria were defined only in terms of nutrient concentrations, and excluded other considerations, this stream could be thought of as nutrient impaired, an inappropriate designation in the absence of excessive algal growth.

The proposed approach for criteria development above is broadly consistent with the approach that has been recommended for performing TMDLs by USEPA. To perform a TMDL, loads must be related to concentrations in water bodies that are often the numerical criteria against which the success of the TMDL will be measured. In the absence of numeric criteria in water, other numeric criteria may apply, such as concentration in fish tissue. However, if no numeric criteria are applicable, EPA recommends identification of potential indicators of impairment, selecting a numeric target for the indicator that protects designated uses, and developing the TMDL on this numeric target.

In our proposed methodology for nutrient criteria development, we propose to take the above TMDL-type approach, where both chemical concentrations as well as biological responses will be part of the criteria. Figure 7-1 illustrates in a simplified form the relationship between the loads and beneficial uses, and identifies the interactions that are the typical focus of TMDL analyses and the subset of interactions that will be studied in the nutrient criteria development process. The role of exogenous factors such as flow, sediment load, habitat quality, temperature, and shade, on biological responses is also shown. Because TMDL analyses are focused on an individual water body, as opposed to groups of water bodies in the criteria development process, it is possible to do a much more detailed analysis of the connections between initial biological responses and beneficial uses in that water body. For practical purposes, therefore, we have chosen to limit our analysis to those types of primary biological responses for which data are more likely to be available, either now or in future, such as Chl *a*, DO, and IBI. Aside from being one step closer to the actual impairment, a significant benefit of including some form of biological response in the criteria is the automatic consideration of some of the hard-to-measure exogenous factors that affect the likelihood of impairment. To take the example of the stream with elevated nutrient levels but low algal biomass, this approach would clearly indicate no nutrient impairment without requiring specific data on canopy cover. The inclusion of chemical concentrations in the criteria is also important. For example, a stream could have an unacceptable IBI, caused by poor habitat quality, where lowering of nutrient concentrations may provide no ecological benefit. In such cases, a nutrient concentration criterion should identify when an observed impairment is unlikely to be due to nutrients.

Measures of biological responses can be used directly to assess attainment of uses in a waterbody. They are more difficult to apply for planning and management, for instance to determine the appropriate level of nutrient loading that should be allowed in a NPDES wasteload allocation. In a TMDL analysis, this problem is addressed by establishing a quantitative linkage between chemical concentrations and biological responses. A chemical criterion target should implicitly recognize the same linkage. That is, a chemical criterion (e.g., phosphorus concentration) should be established at a particular level that protects against occurrence of a biological response sufficient to impair a designated use.

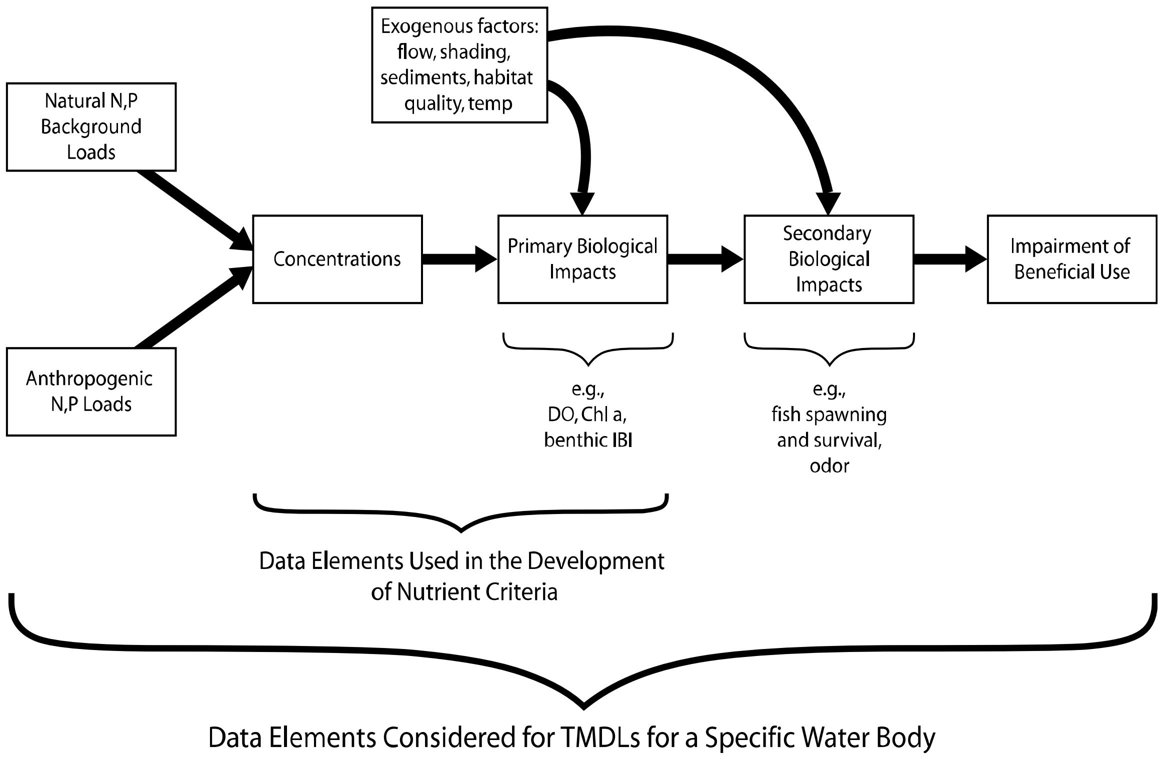


Figure 7-1. Schematic showing the relationship between nutrient loads and impairment of beneficial uses, including data elements used for nutrient criteria development and for development of TMDLs. Impairment of beneficial use will be considered in both cases, but because criteria development deals with groups of water bodies the linkage will be more uncertain than in a single water body TMDL analysis.

8.0 FUTURE WORK FOR 2004-2005 AND RECOMMENDATIONS FOR MONITORING

The nutrient criteria development work for EPA Region IX has been an ongoing effort since 1999. Over this time, the project team has explored different approaches for developing robust, scientifically defensible nutrient criteria, including analysis of available data and modeling. This section provides a brief overview of related future work to be conducted. The general direction for work in 2004-2005 is to:

- link nutrient levels with biological responses that have the potential to impact beneficial uses;
- refine the analytical tools so that they can be transferred to the Regional Boards to expand criteria assessment efforts that will result in better definition of recommended tier boundaries for Ecoregion 6, and for other California ecoregions.

To help address critical shortages of existing data this section also includes general recommendations for future data collection efforts for parameters to be measured in existing monitoring and research programs. These are parameters that are related to nutrient criteria development necessary to improve the quantity and quality of information needed to address key uncertainties in the nutrient criteria development process.

The following recommendations continue to expand on the alternate path chosen by the EPA Region IX Regional Technical Advisory Group (RTAG) and the State and Regional Technical Advisory Group (STARTAG). The alternate path consists of the following components, some of which will be accomplished in 2004-2005.

1) Verification and calibration of criteria development tools (e.g., SWAT, SPARROW, BATHTUB, empirical analytical methods),

2) Training workshops for Regional and State Board staff with the criteria development tools,

3) Data collection by existing monitoring programs expanded for criteria related parameters,

4) Regional and State Board staff contribute to population of criteria numeric endpoints for California ecoregions,

5) Expanded project database to update existing tier boundaries and to propose tier boundaries for remaining California ecoregions.

6) Evaluate impact of proposed nutrient criteria on California estuaries and near-coastal waters.

The involvement of State and Regional board staff into the nutrient criteria development process has several key benefits:

- Consistency in approach across the State,
- More rapid development of numeric endpoints for nutrient TMDLs and information upon which to base nutrient criteria,
- A statewide database of background and existing nutrient conditions, and
- Regional and State Board staff will develop first-hand knowledge of the criteria which can be applied to Basin Plan Objectives and in other regulatory uses of the criteria (e.g., NPDES permit limits).

8.1 FUTURE WORK

Building on past work the project team proposes to undertake some of the following tasks in 2004-2005, following the consensus and support of the nutrient RTAG:

- Communicate results of existing study more widely across the state
- Work with Regional Boards to identify biological endpoints and associated target levels, specific for each major beneficial use that can be tied to nutrients (such as benthic chlorophyll concentrations) using the tools created in the previous phases of this project.
- Work with Regional Boards to develop appropriate monitoring approaches where existing information is insufficient
- Complete SWAT analysis to estimate watershed non-point loads of nutrients
- Conduct an analysis similar to Ecoregion 6 on other ecoregions in EPA Region IX

- Develop a California-specific strategy for nutrient criteria in estuaries

Funding for the upcoming project year is not final and may dictate either expansion or contraction of the list above. In addition, other areas of focus may be substituted for the tasks listed above should the RTAG so decide.

8.2 RECOMMENDATIONS FOR MONITORING

Nutrient monitoring is conducted by a variety of agencies with several chemical parameters represented (typically, total nitrogen, total Kjeldahl nitrogen, nitrate, nitrite, ammonia, total phosphorus, and orthophosphate). These efforts should continue, and emphasis should be placed on obtaining total nutrient concentrations (i.e., including the organic component), rather than just inorganic species. Information on turbidity and TSS (in streams) and Secchi disk depth or depth of light penetration (in lakes), along with temperature, pH, and other standard field parameters should be obtained with all nutrient samples. However, our experience assessing data across California confirmed that information related to the biological impacts of nutrients is fairly limited. Biological impacts are key to relating nutrients to attainment of beneficial uses in water bodies, and collection of additional data in this area is essential for development of meaningful nutrient criteria. Because many biological growth processes are seasonal, this also implies that future measurements consider characterizing intra-annual variability of nutrients, flow, and biological responses.

Some specific biological parameters that should be considered for measurement simultaneously with nutrients are:

- Seasonal chlorophyll-*a* concentrations in lakes and larger rivers.

- Benthic algal densities (as chlorophyll-*a*) in shallow streams, coupled with measurements of turbidity and extent of riparian canopy closure. Agreement on a standard methodology will be important here.

- Diurnal dissolved oxygen range (maxima and minima) to evaluate impacts of algae due to elevated nutrients.

- Identification of non-algal biological responses in selected water bodies, such as macrophytes, benthic invertebrates, and fish communities.

- Quantitative evaluation of biological responses and beneficial use, e.g., identification of levels of chlorophyll that leads to a subjective assessment of impairment.

While recommending additional parameters for monitoring, we recognize that limited resources often constrain water quality monitoring programs. It is our suggestion that, faced with such constraints, efforts be focused on conducting detailed characterization of a smaller number of water bodies, rather than making fewer relevant measurements across a larger number of locations.

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- Srinivasan et al 2000
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