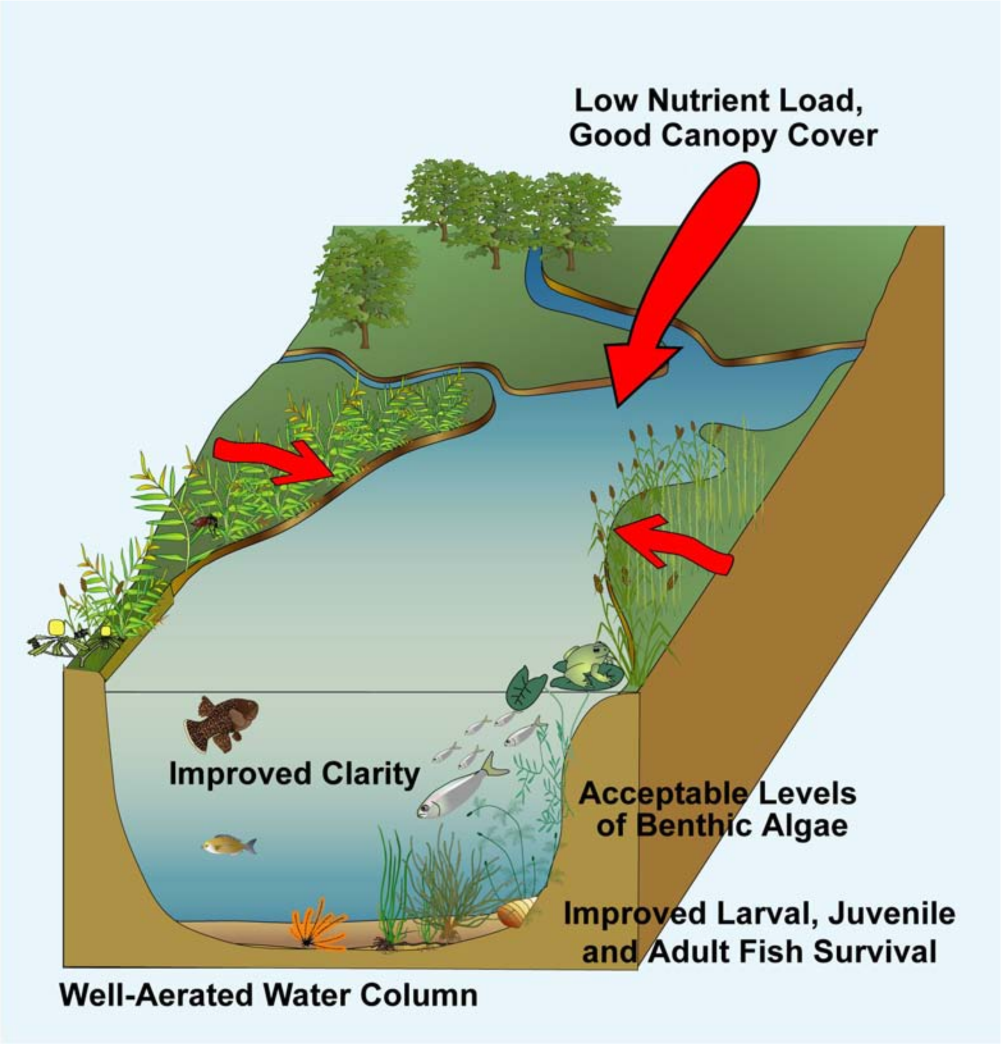
Technical Approach to Develop Nutrient Numeric Endpoints for California

July 2006



Prepared for:

U.S. EPA Region IX
(Contract No. 68-C-02-108-To-111)

California State Water Resource Control Board;
Planning and Standards Implementation Unit

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1 Nutrient Numeric Endpoints for California Water Quality Programs

1.1 THE PURPOSE OF THIS DOCUMENT

This report provides an approach for the development of nutrient (nitrogen and phosphorus) numeric endpoints for use in the water quality programs of the California's State Water Resources Control Board (State Water Board) and Regional Water Quality Control Boards (Regional Water Boards). The approach provides a methodology to support several water quality program components including: setting numeric limits for National Pollutant Discharge Elimination System (NPDES) permits; development of Total Maximum Daily Load (TMDL) nutrient numeric endpoints; and for those Regional Water Boards that choose to, the development of numeric nutrient criteria. Guidance for this technical report comes from participants at workshop (Nutrient Numeric Endpoints Training Workshop held on May 18 & 19 2005 in Sacramento, CA) that is a part of a longer running process that has been sponsored by EPA Region IX and the State Water Board. The workshop summary (Summary of May 18-19, 2005 Nutrient Workshop) of this workshop can be downloaded from the project website: http://rd.tetratech.com/epa/. The CA Nutrient Numeric Endpoints process includes a Regional Technical Advisory Group (RTAG); and a collaborative forum of Regional Water Boards, Tribes, and other state and federal agencies in the State Regional Technical Advisory Group (STRTAG). The result of the RTAG / STRTAG process is an approach that has achieved a high level of consensus among participants. The purpose of this document is to describe the approach that has been reviewed and approved by the STRTAG.

The intention of the proposed approach is to select nutrient response indicators that can be used to evaluate risk of use impairment, rather than using pre-defined nutrient limits that may or may not result in eutrophication for a particular water body. The report provides a description of the proposed approach that includes innovative elements such as:

- A water body classification framework that uses three Beneficial Use risk classification categories;
- Risk-based secondary indicators that are more closely linked to Beneficial Use condition than water column nutrient concentrations; and
- Modeling tools that provide the necessary linkage analysis between secondary indicators and water column nutrient concentrations. The modeling tools also account for site-specific cofactors such as flow, light availability, and others.

This report provides the starting point for a process that will lead to refinements in the classification framework, secondary indicators, and linkage analysis modeling tools through the development of site-specific endpoints. The report has been reviewed by workshop participants; other State, Federal, and Tribal staff; and the Technical Evaluation Committee identified in Appendix 1. This report includes changes made in response to comments received. As an adaptive management process technical updates will be made and new dated editions of this document will be made available. A potential outcome of this process is the adoption of the framework and endpoints by various Regional Water Quality Control Boards for use as nutrient numeric criteria.

1.2 BACKGROUND FOR DEVELOPMENT OF NUTRIENT CRITERIA

The process for developing nitrogen and phosphorus nutrient criteria for the region started in 1998 with the publication of the U.S. Environmental Protection Agency's *National Strategy for the Development of Regional Nutrient Criteria* (USEPA, 1998). USEPA then proceeded to develop national criteria recommendations based on aggregated Level III ecoregions. Data sets from Legacy STORET, NASQAN, NAWQA, and EPA Region 10 were used by EPA to assess nutrient conditions from 1990 to 1998. EPA proposed that the 25^{th} percentiles of all nutrient data could be assumed to represent unimpacted reference conditions for each aggregate ecoregion, and also provided a comparison of reference condition for the aggregate ecoregion versus the subecoregions. These 25^{th} 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." USEPA also encouraged that States and Tribes "critically evaluate this information in light of the specific designated uses that need to be protected."

Several researchers have demonstrated the shortcomings of using ambient nutrient concentrations within a waterbody alone to predict eutrophication, particularly in streams (Heiskary and Markus, 2001; Prairie et al., 1989; Welch et al., 1989; Chételat, et al., 1999; Dodds et al., 2002; Fevold, 1998; Van Nieuwenhuyse and Jones, 1996). Ambient concentration data may not be effective in assessing eutrophication and the subsequent impact on water use because algal productivity depends on several additional factors such as morphology, light availability, flooding frequency, biological community structure, etc.

The problems associated with using nutrient concentrations alone to predict use-support status are demonstrated by a California pilot study conducted in Ecoregion 6 (Tetra Tech, 2003). Tetra Tech categorized 22,000 data points from streams and lakes classified as minimally impacted, unimpaired, impaired by nutrients, or impaired by non-nutrients. Box plots for each available nutrient parameter (ammonia or NH_3, nitrite or NO_2, nitrate or NO_3, total Kjeldahl nitrogen or TKN, phosphate or PO_4, and total phosphorus or TP) were created separately for lakes and streams with data points partitioned for each of the use attainability classes. Yearly and summer-season analyses were performed. Though an increase in the median of each parameter across all data points was correlated with degradation in attainability status, the range of concentrations found in each category overlapped across orders of magnitude. For example (Figure 1-1b), the median nitrate concentration for all summer stream data increased from 0.08 to 0.30 to 5.43 mg/L for minimally impacted streams, unimpaired streams, and streams impaired by nutrients, respectively. But nitrate concentrations as low as 0.01 mg/L and as high as 0.9 mg/L were detected in all three classes of streams, and some unimpaired streams had higher nitrate concentrations than those classified as impaired. Setting a nitrate criteria of 0.30 mg/L to define unimpaired stream segments would incorrectly classify some minimally impacted streams as impaired and some impaired streams as unimpaired. Nitrate was chosen for this example because its median showed the strongest correlation with use attainability.

Welch et al. (1989) suggest that a dynamic modeling approach is necessary to quantitatively evaluate nutrient-biomass relationships for a particular system. It is not feasible to set up such models for each water body in California to determine use status. It is against this background that the California approach described in this report suggests secondary response indicators in place of complex models or simplistic nutrient concentration limitations to assess use support status.

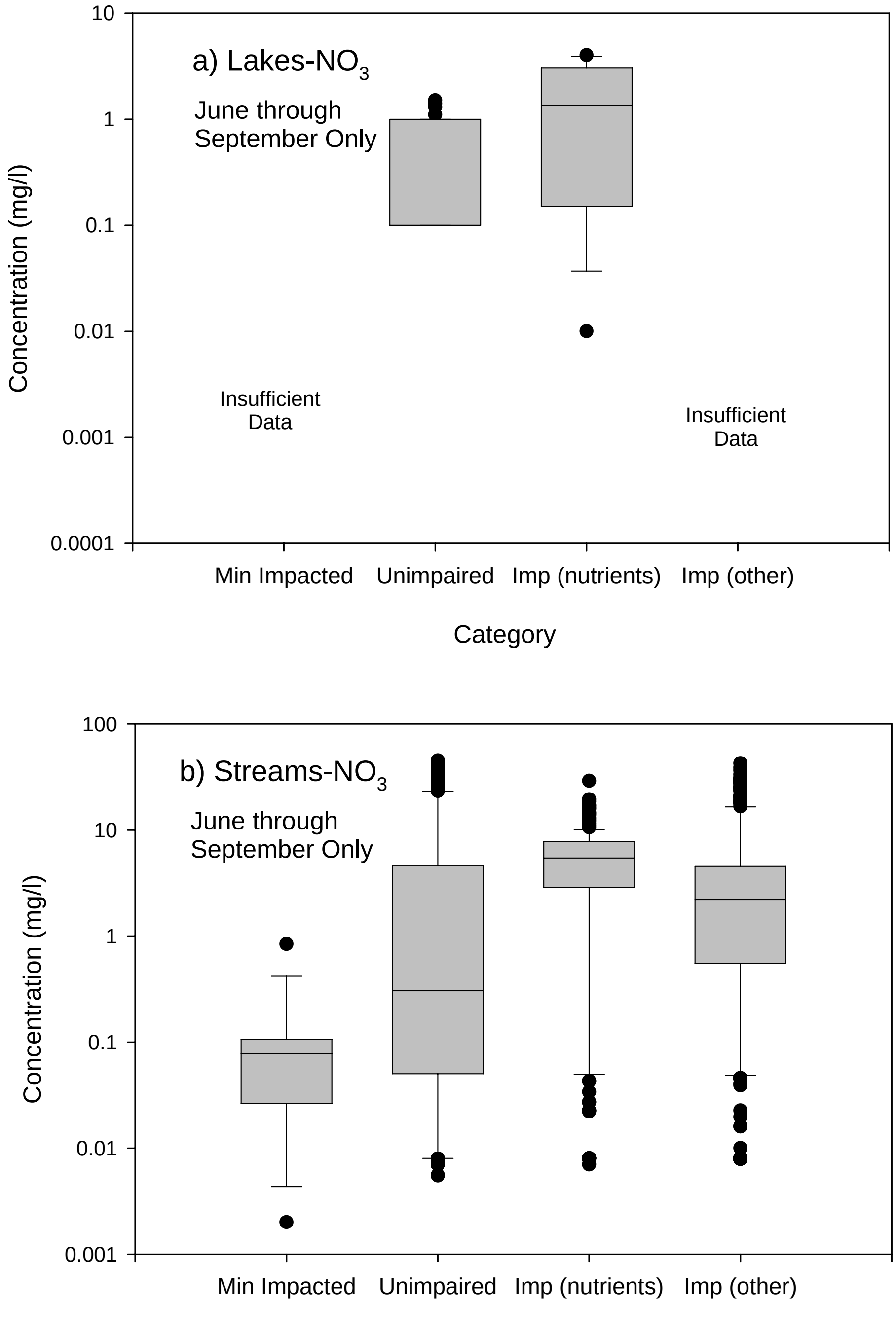

Figure 1-1. Example of Weak Correlation Between a Nutrient Parameter (Nitrate) and Use Attainability (from Tetra Tech, 2003)

1.3 How Did We Get To This Point?

The purpose of this section is to provide a brief summary of the stakeholder and technical process that has led to the development of the proposed California approach.

USEPA 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 USEPA guidance documents for developing nutrient criteria. The RTAG included representatives from all State water quality agencies in Region IX, Tribes, other State and Federal agencies, and representatives from industry and environmental groups. In 2001, the California State Water Board created the State Regional Board Technical Advisory Group (STRTAG) to work in parallel with the RTAG and assume responsibility for continuing to move nutrient development forward for California and to better coordinate the activities of the individual Regional Water Boards. The RTAG and STRTAG have collaborated on the direction of technical support activities for this initiative.

The RTAG conducted a pilot project in 1999 and 2000 to develop a water quality database organized by ecoregion to assess the availability of existing water quality and biological data to support nutrient criteria development, and 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 USEPA would require some refinement and supporting studies to ensure that the adopted criteria were appropriate. In 2000 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 (TP) and total nitrogen (TN) suggest that if the EPA reference-based values (draft 304(a)) are adopted that a large number of probably un-impaired water bodies would be misclassified as impaired. Therefore the RTAG and STRTAG responded to this potential for misclassification by adopting a resolution to pursue the USEPA approved alternative to development alternate nutrient criteria.

The current proposed approach is the result of refinements developed through a series of pilot studies undertaken at the direction of the RTAG / STRTAG from 2000 through 2005. The pilot studies evaluated the feasibility of using an ecoregional and a sub-ecoregional approach employing a landscape stratification strategy. Several parameters have also been evaluated for inclusion in a nutrient criteria index. Many elements evaluated in previous pilot studies have not been adopted due to technical issues or a lack of data. The results of the pilot studies have been documented in a series of annual reports that have been compiled on the project website: http://rd.tetratech.com/epa/.

Several important elements of the current approach were developed through these pilot studies including several elements described in this report: the risk classification categories, secondary indicators, and linkage analysis models. The current approach relies on the need to develop site-specific nutrient numeric endpoints for TMDLs and NPDES permit limit determinations to gradually accumulate a database. The information on nutrient numeric endpoints will be available through SWAMP and CIWQS allowing California water quality programs to move beyond these site-specific applications to the development of water quality objectives for inclusion in Basin Plans. The framework that evolved from this regional process is further described in the following section.

1.4 The Proposed California Approach To Develop Nutrient Numeric Endpoints

The purpose of this section is to provide a summary of the California approach for developing nutrient numeric endpoints. Several portions of this approach are described in greater detail in later sections of this report. As stated above the California approach was developed through a series of studies funded by USEPA Region IX and the State Water Board between 1999 and 2005.

Except in extreme cases, nutrients alone do not impair beneficial uses. Rather, they cause indirect impacts through algal growth, low DO, and so on, that impair uses. These impacts are associated with nutrients, but result from a combination of nutrients interacting with other factors. Appropriate nutrient targets for a waterbody should take into account the interactions of these factors to the extent possible. For instance, the nutrient concentration that results in impairment in a high-gradient, shaded stream may be much different from the one that results in impairment in a low-gradient, unshaded stream. Instead of setting criteria solely in terms of nutrient concentrations, it is preferable to use an analysis that takes into account the risk of impairment of uses. Conceptually this is similar to the allocation procedure for Biochemical Oxygen Demand (BOD), under which BOD loads are controlled to achieve acceptable levels of indirect impacts on Dissolved Oxygen (DO), rather than to meet an arbitrary concentration criterion for BOD in the receiving water.

The nutrient criteria framework needs to contain, in addition to nutrient concentrations, targeting information on secondary biological indicators such as benthic algal biomass, planktonic chlorophyll, dissolved oxygen, dissolved organic carbon, macrophyte cover, and clarity. These secondary indicators provide a more direct risk-based linkage to beneficial uses than the nutrient concentrations alone.

Nutrients occur naturally, and vary in relationship to soils, geology, and land cover. Indeed, nutrient concentrations that are too low may also impair certain uses. It makes little sense to set a nutrient criterion that is lower than natural background for a specific waterbody, as may occur through application of ecoregional statistical criteria.

For many of the biological indicators associated with nutrients there is no clear scientific consensus on a target threshold that results in impairment. To address this problem, we propose to classify water bodies into the three Beneficial Use Risk Categories (BURCs) illustrated in Figure 1-2. Beneficial Use Risk Category I water bodies are not expected to exhibit impairment due to nutrients; BURC III water bodies have a high likelihood of exhibiting impairment due to nutrients; and BURC II water bodies may require additional information and analysis. We believe this three-tiered approach is better than a binary meet/does not meet criteria approach. For a given beneficial use designation, the BURC I/II boundary represents a level below which there is general consensus that nutrients will not present a significant risk of impairment. (This boundary should also be set so that is not less than the expected natural background.) Conversely, the BURC II/III boundary represents a level that is sufficiently high that there is consensus that risk of use impairment by nutrients is probable. Within BURC II, additional water body-specific cofactors may be brought into the analysis to determine an appropriate target. Permitting discharges to waters that remain within BURC II after additional analysis would require an antidegradation or reasonable potential effect analysis.

The California NNE approach proposes preliminary numeric targets (BURC boundaries) for each of the secondary indicators using literature sources and elicitation from the Regional Water Boards. A summary of many of the studies used in developing the endpoint recommendations is included in Appendix 2. It is thought that these values will not change very much from region to region within California. Thus, benthic algal biomass levels that impair the spawning beneficial use are considered to be similar for different parts of the state. The same is true for the other secondary indicators, with the exception of macrophyte cover, which may not be usable as a generalized indicator. The CA NNE approach is based on lines of evidence that incorporates natural background conditions; the status of risk cofactors (e.g., habitat integrity, flow); and the relationship between secondary indicator response variables (e.g., chlorophyll a, clarity, DO, and pH maximums). The CA NNE approach also includes spreadsheet modeling tools to evaluate various nutrient concentration targets to achieve the desired condition for secondary indicators. However, it is critical that these tools be used in the context of the overall approach as a single line of evidence. The CA NNE approach requires a good understanding of the individual waterbody being evaluated and consideration of all of the lines of evidence.

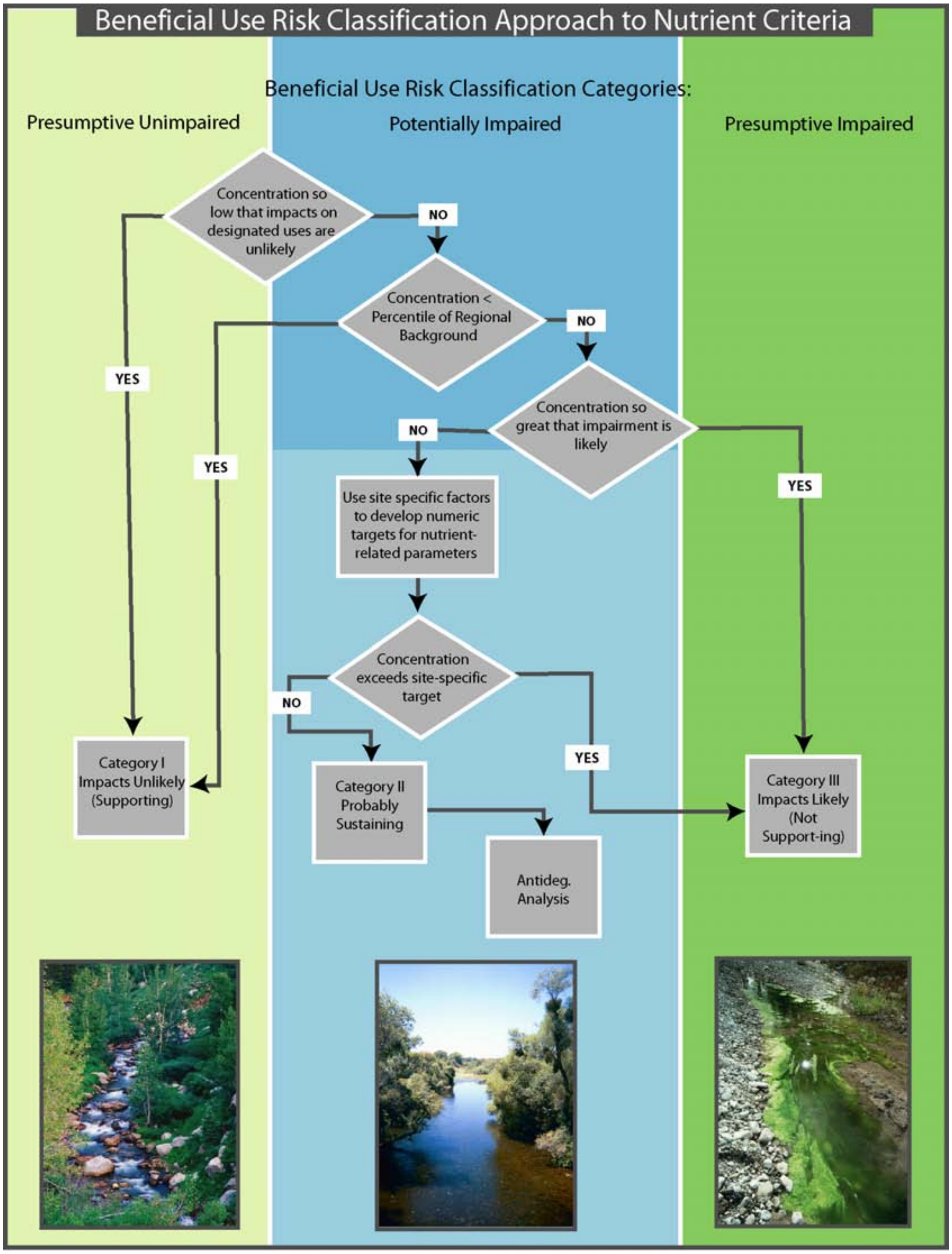

Figure 1-2. Beneficial Use Risk Classification (BURC) Categories and Nutrient Assessment Process.

Secondary indicator targets can be converted to nutrient concentration targets appropriate for assessment, permitting, and the calculation of TMDLs by using simulation models for biological responses in reservoirs/lakes and rivers/streams. Relatively simple tools can provide initial targets, although site-specific refinements may be needed for individual waterbodies. Description and documentation for use of simplified tools is included as Appendices 3 and 4 of this report. In addition, the tools are included on the accompanying CD-ROM disk. These modeling tools are available for general use and can be downloaded from the project website: http://rd.tetratech.com/epa/

File names:

- CA_NNE_Benthic_Biomass_Predictor_V12
- CA_NNE_BATHTUB_V11

The nutrient targets so derived may be compared with reference nutrient levels in different regions in California. Nutrient concentration targets derived from secondary indicators are acceptable if they are not lower than background levels in that region. Depending on the use, user perceptions, data availability, and economic impact of the decision, other, more detailed and site-specific tools may be needed for translating secondary indicator targets to nutrient concentration targets.

To limit the potential for downstream impacts nutrient criteria may require reach-specific limits on upstream concentrations consistent with TMDL allocations. Achieving nutrient reductions to control downstream impacts may require more stringent restrictions in upstream reaches than would be otherwise necessary for uses within those reaches alone. For instance a stream entering a reservoir may need lower nutrient numeric endpoints upstream, not to protect against upstream secondary impacts but to protect against impacts within the reservoir.

The lessons learned from the experience gained through several years of pilot studies for the development of nutrient criteria suggests that no one approach will be suitable for all the diverse water bodies within California. However, we believe that the proposed risk-based approach will provide solutions to many if not most of the issues that need to be addressed in setting numeric nutrient endpoints for California.

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2 Uses and Impairments – A Risk-Based Approach

The approach taken for California is to propose nutrient numeric endpoints 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 designated 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 to link specific designated 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 designated 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 Designated 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 designated 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, water bodies are assigned multiple beneficial uses.

Agricultural Supply: 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: Designated by the State Water Resources Control Board. 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: 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: 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 designated uses.

Groundwater Recharge: 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: 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: 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: Uses of water for hydroelectric power generation. Elevated nutrients are unlikely to result in significant impairment of this use.

Municipal and Domestic Supply: 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: 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: 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: 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: 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: 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: 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: 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: Uses of water that support warm water ecosystems including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish, or wildlife, including waterfowl. 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: 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 not conducive to full support of aquatic life. 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: 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 designated 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 designated 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 (SPWN), 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 (USEPA, 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 USEPA's watershed-level ecological risk assessment case studies (Butcher et al., 1998). In each of the five USEPA-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 (USEPA, 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 [USEPA (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 (USEPA, 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 designated uses. General conceptual models for the impairment of key uses in lakes and streams by nutrients are presented in Figure 2-1 and Figure 2-2. The illustrated conceptual models also include major exogenous factors that influence how nutrients are processed within a water body, and / or have a direct impact on the endpoints. Exogenous factors are included in the California approach because they are critical to the decision-making process to maintain or restore water body integrity. These exogenous factors, identified in the conceptual model, also affect the allowable nutrient levels necessary to maintain or protect the desired beneficial uses. Additional linkages may be significant in individual waterbodies; however, most of the major linkage connections are captured in these figures.

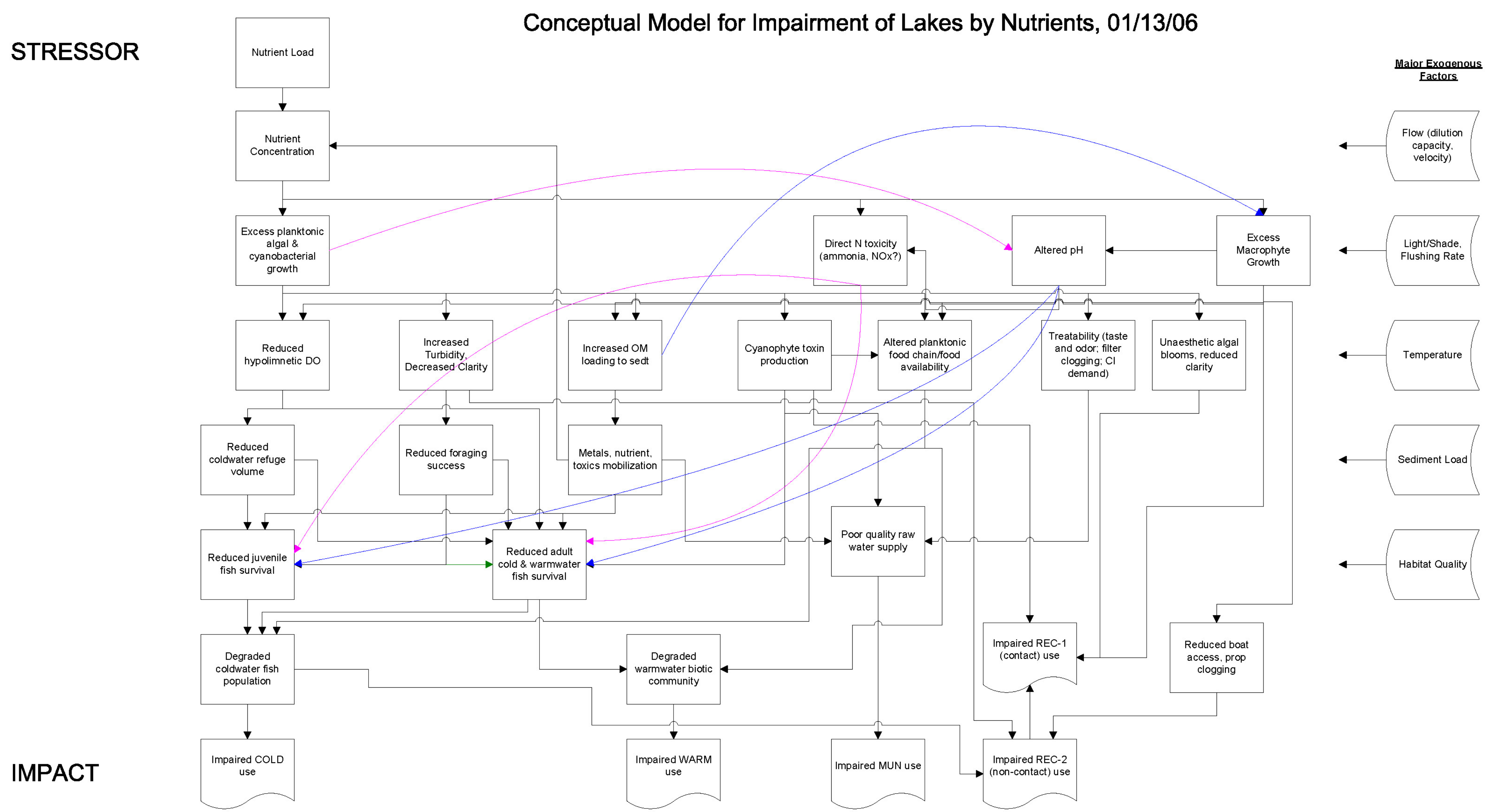

Figure 2-1. Conceptual Model for Lakes

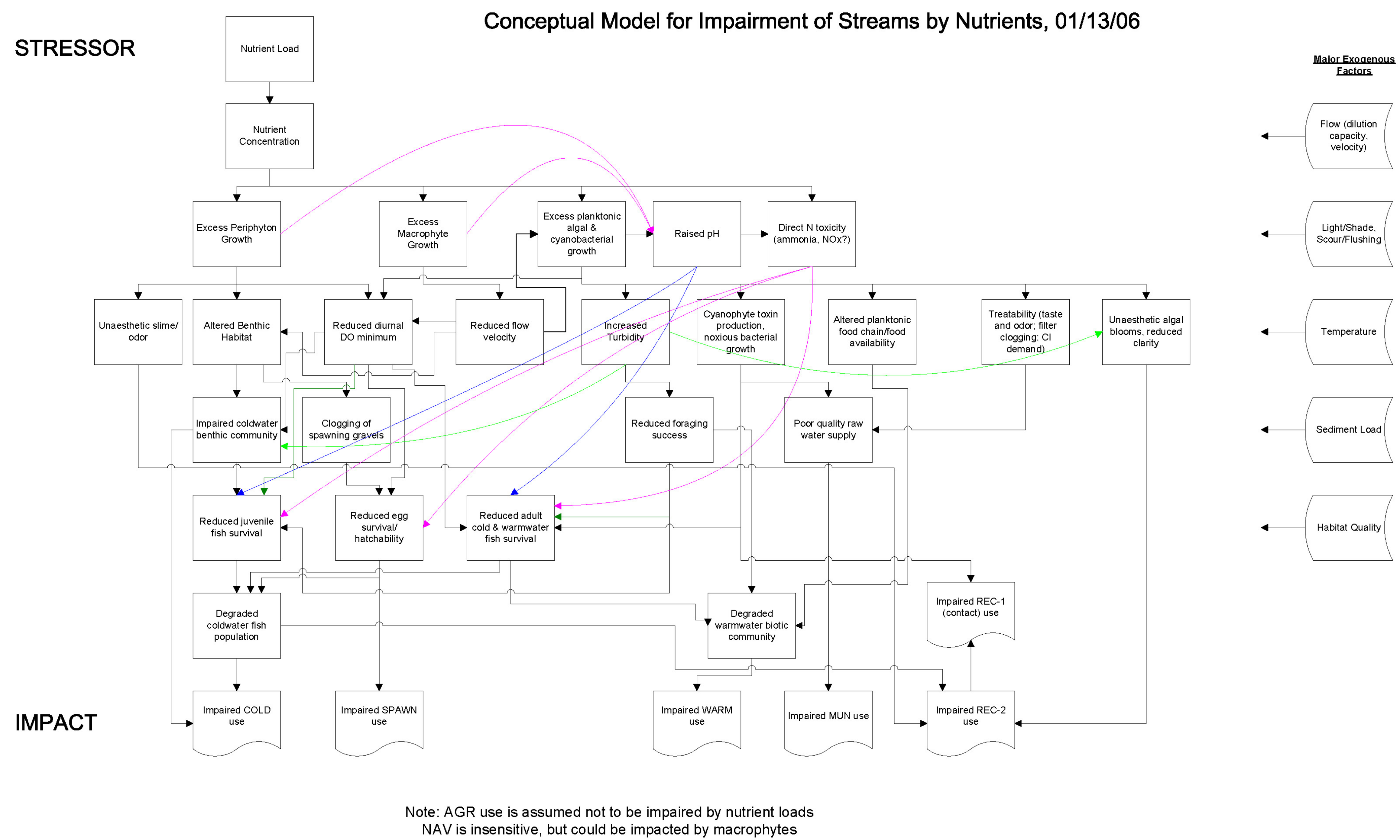

Figure 2-2. Conceptual Model for Streams

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, and 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 and pH, 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. These key indicators are discussed in greater detail in the following section.

Table 2-1
Stressor-Response Factors

Use	Key Stressor Response Factors								Secondary Factors
	Reduced hypolimnetic DO	**Increased Turbidity**	**Cyanophyte toxins**	**Altered food chain**	**Toxic metal, NH4 cycling**	**Taste & Order**	**Unaesthetic Blooms**	**Excess Macrophytes**	
COLD	x	x	x	x					Summer chl a, cyanophyte blooms, turbidity, 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

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3 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 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," 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 measures of ecosystem and receptor characteristics (USEPA, 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.

3.1 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 designated 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 – that is, 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 because they represent a key intersection along the complex path from nutrient loading to impairment of designated uses.

Some states have addressed nutrient criteria through direct measures of exposure – setting target concentrations of nutrients applicable to a class of water bodies. Other states have focused on intermediate measures or indicators. 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 on 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 level 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 indicators 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 numeric endpoints 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 indicators assigned to ensure support of a designated 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 from 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 indicator). 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 nutrient targets, but are applicable without a detailed, site-specific study. One outcome of the assessment process associated with numeric endpoints of this type is to identify those marginal sites for which a more detailed analysis is warranted before allowing additional nutrient loads. The assessment outcome is consistent with the risk classification framework (described in Section 1.3) used in the California approach to assign each water body to its appropriate BURC. There are three categories: I. Presumptive Unimpaired; II. Potentially Impaired; and III. Presumptive Impaired. The framework requires that each indicator be assigned a value for the boundary between categories I and II, and categories II and III.

3.2 Indicators and Targets for Lakes and Reservoirs

The intermediate indicators most relevant to support of specific uses in lakes and reservoirs are primarily measures of 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 surface or epilimnetic 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 biomass 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 or empirical approach that takes into account the major stratification factors that cause site specific differences in response.

3.2.1 Chlorophyll *a*

The chlorophyll *a* content of water samples is a surrogate measure of algal biomass and is commonly used to assess eutrophication of lakes and other lentic water bodies. In California, water quality objectives for chlorophyll *a* range from narrative descriptions to numeric criteria. This section describes methods used across the US to incorporate chlorophyll *a* as a water quality indicator.

Nine Regional Water Boards were created to provide legislative guidance to the State of California for the purpose of protecting human health and water quality in relation to waters within their boundaries. Each board was assigned the task of setting water quality objectives in addition to State Water Board objectives, that would “ensure reasonable protection of beneficial uses and the prevention of nuisance” (Porter-Cologne Act of the California Water Code, Section 13000, Water Quality). The nine plans can be viewed at the following website: http://www.epa.gov/ost/standards/wqslibrary/ca/ca.html#basin.

In regards to biostimulatory substances, each Regional Water Board cites the same narrative criteria: “Waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growths cause nuisance or adversely affect beneficial uses.” None of the Regional Water Boards has defined a quantitative limit for nuisance growth.

Only the San Diego Regional Water Board places regional numeric limits on biostimulatory substances by explicitly limiting total phosphorus concentrations in streams to 0.05 mg/L at the point of entry to a body of standing water. Bodies of standing water have a limit of 0.025 mg-P/L. The limit for streams or other flowing waters is 0.1 mg-P/L. These objectives are not to be exceeded more than 10% of the time, excluding approved exceptions. Region-wide nitrogen limits have not been set, though natural ratios of nitrogen to phosphorus should be maintained. In the absence of data, a nitrogen to phosphorus ratio of 10:1 should be implemented. The Rainbow Creek TMDL produced by the San Diego Regional Water Board (2005), sets the TP target to 0.1 mg/L and the TN target to 1 mg/L, not to be exceeded more than 10 percent of the time. No chlorophyll *a* objectives are suggested for this region.

The Santa Ana Regional Water Board has set nutrient targets for Lake Elsinore and Canyon Lake (SAR, 2004) to 0.1 mg-P/L and 0.75 mg-N/L based on the 25th percentile of data collected in Lake Elsinore. Average chlorophyll *a* concentrations should not exceed 25 μg/L. The Lahontan RWQCB limits annual average nutrient concentrations in the Bridgeport Reservoir (Horne, 2003) to 0.5 mg-N/L and 0.06 mg-P/L. The 90th percentile targets are 0.8 mg-N/L and 0.1 mg-P/L.

Several states quantify the usability of a water body by its trophic state. The State of Michigan relies on the Carlson Trophic State Index for Secchi Depth, chlorophyll *a*, and total phosphorus to differentiate among oligotrophic, mesotrophic, eutrophic, and hyper-eutrophic lakes (MDEQ, 1999). Oligotrophic lakes are capable of supporting cold-water fish because they are minimally productive and maintain high DO levels Eutrophic lakes with high levels of aquatic productivity support warm-water fish, which are not as sensitive to low dissolved oxygen concentrations as cold-water species. Lakes experiencing nuisance algal blooms are termed hypereutrophic. The Michigan criteria for summer mean chlorophyll *a* concentrations for cold-water fish lakes is less than 3 μg/L. To support warm-water fish, chlorophyll *a* concentrations should be less than 40 μg/L. Research is not available on chlorophyll a levels protecting spawning uses of waters.

For the protection of cold-water fish, a target chlorophyll *a* concentration of 3 μg/L may seem unrealistic. Nevertheless, a review of the data used to develop the USEPA Ecoregion regression approach for California shows that the range of 25th percentile values for chlorophyll *a* (proposed targets for each Ecoregion) is 0.9 to 4.4 μg/L. However, only four of the 12 Ecoregions in the State have at least four data points with which to determine the chlorophyll *a* criteria. Data from two Ecoregions in California result in suggested chlorophyll *a* criteria less than 3 μg/L. All four Ecoregions have data with 25th percentiles less than 5 μg/L.

The review of the USEPA Ecoregion dataset also confirmed the problems inherent to nutrient-only water quality criteria. The 25^{th} percentile total phosphorus concentrations across the Ecoregions ranged from 7.1 to 172 μg-P/L. However, the response as measured by chlorophyll *a* was fairly consistent, with a range of 0.9 to 4.4 μg/L.

The North Carolina State University Water Quality Group (NCSU, 2005) suggests that water supply reservoirs maintain mean chlorophyll *a* concentrations less than 15 μg/L and waters designated for recreation less than 25 μg/L. The State of Oregon has phytoplankton water quality standards of 10 μg/L for lakes that thermally stratify and 15 μg/L for lakes that do not thermally stratify or for rivers and streams (ODEQ, 2004). The State of Iowa typically sets chlorophyll *a* targets at 33 μg/L to correspond with a desired Carlson TSI of 65 (IDNR, 2005).

The Indian Creek Reservoir in California has designated uses of municipal water supply, agricultural supply, groundwater recharge, navigation, contact and non-contact water recreation, cold freshwater habitat, and wildlife habitat. The nutrient TMDL assigns a Secchi Depth of no less than 2 ft and a maximum summer chlorophyll *a* concentration of 10 μg/L to protect designated uses.

Across the nation chlorophyll concentrations have been incorporated into nutrient TMDLs. Lake Linganore, Maryland is designated for public water supply and recreational trout fishing. The target chlorophyll *a* concentration is 10 μg/L. In Oregon, the Upper Klamath and Agency Lakes have chlorophyll *a* targets of 15 μg/L. They are both designated for the protection of aquatic life. McPherson Lake in Kansas has a target chlorophyll *a* concentration of 12 μg/L and is designated for primary and secondary contact recreation and aquatic life support. McDaniel Lake, Missouri is designated for primary drinking supply, warm-water fish habitat, and aquatic life support. The chlorophyll *a* criterion for this lake is also 10 μg/L.

Selection of a summer mean chlorophyll *a* target also has implications for the frequency of severe bloom conditions (defined as concentrations greater than 30 µg/L). In work on USACE reservoirs, Walker (1985, 1987) determined that the distribution of chlorophyll *a* concentrations in an impoundment could generally be described as lognormal. An estimate of the frequency of time that concentrations are greater than any target value can then be made from the arithmetic mean target concentration and a coefficient of variation on the log-transformed values (CV; standard deviation divided by the mean), using the algorithm found in Walker (1985).

Results of the analysis depend on the selection of an appropriate CV value. Walker (1987) states that the temporal CV for chlorophyll *a* concentrations in ACOE reservoirs was 0.62; however, the accompanying computer program defaults to 0.26. In an apparent later reanalysis, Figure 7.6 in Welch and Jacob, (2004) appears to have been calculated with a CV of 0.17, citing personal communication from Walker “for calibration to Corps of Engineers reservoirs.” Temporal CVs will likely differ for other datasets.

Table 3-1 shows the frequency of severe bloom conditions (concentrations greater than 30 µg/L) for different summer mean chlorophyll *a* targets and various assumptions regarding CV. Based on this analysis, setting a summer mean target of 5 µg/L means that blooms will almost never occur, while a target of 10 µg/L implies that such blooms will be rare. A target of 20 µg/L suggests blooms will occur about 15-20 percent of the time, which is suggested as the maximum allowable level consistent with full support of contact recreation use. A target mean concentration of 25 corresponds to blooms about one quarter of the time.

Table 3-1
Frequency of Chlorophyll *a* Concentrations Greater than 30 µg/L using the Method of Walker (1985)

Summer Mean Chlorophyll a (µg/L)	CV = 0.62	CV = 0.26	CV = 0.17
5	0.4 %	0.0 %	0.0 %
10	5.9 %	1.2 %	0.1 %
20	16.7 %	17.7 %	14.1 %
25	20.4 %	26.8 %	27.1 %

Because it is relatively easy to measure, a response target defined as a concentration of chlorophyll *a* provides a natural basis for assessing use support status in response to nutrient enrichment. Clear consensus does not exist, however, as to what target levels are appropriate for attainment of different uses. The selection of a target will need to combine both scientific and policy components.

Evaluation of a target also needs to consider questions of temporal and spatial applicability consistent with the desired use protection. Temporally, a chlorophyll *a* target can be defined as a point-in-time measurement (or frequency of such measurements) or as an average over a year, season, or other period. Spatially, the target could be applied as a lakewide average, a concentration at a specific point (e.g., outlet forebay), or in relation to specific sub-habitat areas.

3.2.2 Cyanobacteria

Exposure to biotoxins produced by certain cyanobacteria (blue green algae) can cause a wide range of human health problems ranging from skin irritation to organ damage (Cadmus, 1996). There are currently no water quality criteria in the U.S. for cyanobacteria or the biotoxins they produce.

Australian authorities have set limits on one of the toxic byproducts of cyanobacteria (Brookes et al., 2004). Some species such as *Microcystis aeruginosa* produce the hepatoxin microcystin. The Australian drinking water criterion for total microcystin is 1.3 µg/L. Other species of cyanobacteria produce neurotoxic saxitoxins, or other hepatoxins such as nodularin and cylindrospermopsin. The Australian government is currently gathering data for the determination of appropriate drinking water standards for these toxins. Eventually, criteria will be set for the protection of recreational contact waters.

Cyanobacterial blooms are also of concern for other reasons. For instance, a shift in algal species composition toward cyanobacteria in a waterbody can alter the ecology of zooplankton, in turn affecting the ability of the water to support a native fishery. From an aesthetic perspective, various cyanobacteria form mats or scums that can render a waterbody unsuitable for recreation.

In British Columbia, waters classified for primary recreation and aquatic life must not be dominated by cyanobacteria (less than 50 percent of cells per volume) (MELP, 1992). Though no limitations were suggested for water supply uses, MUN waters should be as protected.

3.2.3 Macrophyte Density

The biostimulatory narrative criteria apply to nuisance "aquatic growth" – thus including macrophytes as well as algae. Excess nutrient loads can promote macrophyte growth, although in many cases of use impairment the introduction of non-native nuisance species is a greater cause of use degradation. Unfortunately, models to predict response of rooted macrophytes to loads are not well developed, in part because many such plants can withdraw nutrients stored in sediments (Welch and Jacoby, 2004). Further, there seems to be little agreement as to what measure of macrophyte coverage represents problem conditions. Some TMDLs addressing macrophytes have thus focused on macrophyte-driven impacts on DO and pH.

Nutrient loading to lakes and reservoirs can affect the density, type, and distribution of aquatic macrophytes. Impairments of use can potentially arise either through enhancement of macrophyte growth to levels that impede boat traffic and alter the DO and pH cycles, or through suppression of submersed macrophyte growth by increased algal turbidity, resulting in degraded fish habitat.

There are significant difficulties in predicting macrophyte response to cultural eutrophication. Some studies (e.g., Remillard and Welch, 1993) have suggested that macrophyte coverage in lakes is correlated to nitrogen concentrations, but not sensitive to water column phosphorus concentrations. However, most of the cases in which excessive macrophyte growth is reported as a cause of impairment in lakes appear to result from a combination of the introduction of noxious non-native aquatic weeds with lake morphometry that favors growth (unshaded area at proper depth range).

Various nutrient TMDLs (e.g., Tetra Tech, 1999) have addressed excessive macrophytes in lakes as an endpoint and as an intermediate link in the recycling of nutrients into the water column. However, these have typically not identified a specific quantitative linkage between nutrient load and macrophyte density but rather recommended an adaptive management approach. As another example, a northeastern nutrient/macrophyte TMDL (MA DEP, 1999) takes the approach of reducing phosphorus concentrations to a value typical of lakes in the region, recognizing that “there is no loading capacity *per se* for nuisance aquatic plants”, then recommends achieving uses through the combination of nutrient load reduction and direct harvesting of macrophyte biomass.

As with streams, it appears unlikely that macrophyte biomass is a useful general measure for quantitative target determination for nutrients. Where macrophytes need to be addressed as an endpoint it appears that one must either take a site-specific approach or develop targets based on more general relationships, such as the ability of nutrient concentrations to support a given level of planktonic algal growth.

3.2.4 Transparency and Secchi Depth

Secchi depth is a measure of lake water clarity that depends on the amount of floating and suspended algae, suspended sediment, and dissolved organic compounds (color) in the water column. When estimating lake productivity, Secchi depth measurements should be used in conjunction with chlorophyll *a* measurements to determine if the reduced clarity is due to the algal component or sediment and organic content (UFL, 2005). High Secchi depths mean clear water, while low Secchi depths indicate reduced visibility. Secchi depth is inversely related to algal productivity such that high Secchi depths are associated with clear water with low algal populations. Low Secchi depths are common in highly productive waters with high biomass of floating algae.

The State of Michigan uses the Carlson Trophic State Index discussed in Section 3.2.1 (MDEQ, 1999) to set appropriate Secchi depths for WARM and COLD waters. Secchi depths greater than 4 m (13.1 feet) are typical of oligotrophic lakes and should be indicative of conditions supporting the COLD use. Eutrophic lakes generally have much lower Secchi depths, but those with Secchi depths greater than 0.6 m (2 feet) are likely to support WARM uses (MDEQ, 1999).

For recreational water uses, visibility is a safety issue. In British Columbia, the Secchi depth objective is a minimum of 1.9 meters (~6 ft) for waters designated for primary recreation or aquatic life uses (MELP, 1992). No guidelines for US streams were found.

Secchi depth is not a direct indicator of negative impacts to water supply uses. High turbidity may adversely affect treatment processes while also reducing Secchi depth, but is usually driven by inorganic suspended sediment concentrations, No clarity targets directly connected to nutrient criteria are suggested for the MUN use in this report.

3.2.5 pH

Algae can alter the pH of water through the uptake or release of CO_2. The following reactions demonstrate how photosynthetic organisms convert CO_2 and water to sugar and oxygen during photosynthesis and how during respiration the reaction is reversed. During daylight hours, photosynthesis and respiration occur simultaneously though photosynthesis occurs at a much faster rate. In the absence of sunlight, only respiration occurs.

$$\text{Photosynthesis: } 6CO_2 + 6H_2O + Sunlight \Rightarrow C_6H_{12}O_6 + 6O_2$$

$$\text{Respiration: } C_6H_{12}O_6 + 6O_2 \Rightarrow 6CO_2 + 6H_2O$$

During photosynthesis, CO_2 is consumed and pH increases. During respiration, CO_2 is released and dissolves in water to form carbonic acid (H_2CO_3), which lowers pH by adding hydrogen ions to the water:

$$H_2O + CO_2 \Leftrightarrow H_2CO_3 \Leftrightarrow H^+ + HCO_3^-$$

$$HCO_3^- \Leftrightarrow H^+ + CO_3^{2-}$$

The impact of photosynthesis on pH is governed by the alkalinity or buffering capacity of the water. Alkalinity is typically present in the carbonate or bicarbonate form and is measured in units of mg-$CaCO_3$/L. Carbonate and bicarbonate is capable of reacting with both acids and bases to minimize changes in pH. Alkalinity concentrations should be greater than 20 mg-$CaCO_3$/L to prevent large swings in pH due to photosynthesis and respiration (Wurts and Durborow, 1992). The buffering mechanisms of calcium carbonate are shown below.

$$CaCO_3 \Leftrightarrow Ca^{2+} + CO_3^{2-}$$

$$CO_3^{2-} + H^+ \Leftrightarrow HCO_3^-$$

$$HCO_3^- + H^+ \Leftrightarrow H_2CO_3$$

$$Ca^{2+} + 2OH^- \Leftrightarrow Ca(OH)_2$$

During acidic conditions, carbonate (CO_3^{2-}) and bicarbonate (HCO_3^-) remove hydrogen ions from the water to form bicarbonate and carbonic acid, respectively. During basic conditions, calcium (Ca^{2+}) binds to hydroxyl ions (OH^-) to form calcium hydroxide. Removal of excessive hydrogen or hydroxyl ions prevents the system from experiencing large swings in pH.

The water quality objectives for hydrogen ion concentration vary for each Regional Water Board in California. In the Central Coast Region Basin Plan, the objective for COLD, WARM, or general use waters is a pH range of 7.0 to 8.5. Waters classified MUN, REC-1, or REC-2 have an allowable range of 6.5 to 8.3. The Central Valley Basin Plan has a site-specific objective for Tulare Lake that sets the pH objective to 6.5 to 8.3 for all water uses with the additional requirement that normal ambient pH should not be changed by more than 0.3 unit. In the San Diego Region Basin Plan, the allowable range of pH is 6.5 to 8.5. Waters specified COLD or WARM should not have a change in ambient pH greater than 0.5 unit. The allowable range in the Colorado River Basin Plan is 6.0 to 9.0, and no discharge should cause a pH change detrimental to beneficial water uses. The Santa Ana, San Francisco, Los Angeles, Lahontan, and North Coast Basin Plans all specify an allowable pH range from 6.5 to 8.5. The San Francisco and Los Angeles Regional Water Basin Plans do not allow changes of more than 0.5 unit from normal ambient pH due to discharge. The Lahontan and North Coast Regional Basin Plans limit changes in the pH (regardless of cause) to 0.5 unit for all waters designated COLD or WARM.

Site-specific objectives in the North Coast Basin Plan range from 6.5 to 9.0 for freshwaters. In the Lahontan Basin Plan, site-specific objectives generally state that a change in normal ambient pH greater than 0.5 units is unacceptable (Little Truckee River, Truckee River, West Fork Carson River, and East Fork Carson River). In Eagle Lake, the pH of the hypolimnion should not be less than 7.6, and all other Eagle Lake waters must not have a change in normal ambient pH greater than 0.1 unit. At Honey Lake, the pH should be no less than 8.0 and no more than 10.0 based on the average of at least three samples collected from three different locations. An assessment of potential water quality stress to fish conducted as a supplement to effects of water quality and lake level on the biology and habitat of selected fish species in the Upper Klamath Lake (Loftus 2001) severe stress and mortality for rainbow trout was noted at pH 9.5 in controlled experiments. Finally, in Lake Tahoe, the allowable pH range is 7.0 to 8.4.

The USEPA (1986) has set criteria for pH according to water use. Freshwaters supporting aquatic life should have a pH between 6.5 to 9.0. Waters designated MUN have an allowable range of 5.0 to 9.0.

3.2.6 Dissolved Oxygen

The DO concentration in a waterbody reflects the balance between reaeration and internal oxygen consumption, primarily through bacterial respiration. In many cases, the dominant factor for oxygen consumption is the decay of non-living organic matter, including both anthropogenic wastes and natural substances. Excess growth of algae can affect DO concentrations in a variety of ways. As a direct effect, photosynthetic production by algae releases oxygen, while respiration consumes oxygen. This leads to a diurnal cycle in which the presence of algae increases oxygen concentrations during the day (if sufficient light is present) and decreases oxygen concentrations at night – generally evinced as a sine curve imposed on the oxygen concentration due to other factors. In addition, as an indirect effect, algae that die contribute to the pool of non-living organic matter subject to bacterial decomposition. This can result in dramatic DO depression during periods of algal bloom die-off.

Ambient water quality criteria for the support of aquatic life are well established for dissolved oxygen (DO). Dissolved oxygen criteria in California, however, vary from region to region. The most stringent criteria are set in the Colorado River Basin Plan, where the minimum allowable DO is 5.0 mg/L for WARM uses and 8.0 mg/L for COLD uses.

The Central Valley Basin Plan objectives for Tulare Lake, Central Coast Basin Plan, and the San Francisco Bay Basin Plan all assign minimum DO for WARM waters to 5.0 mg/L and for COLD waters to 7.0 mg/L. The Central Coast Basin Plan also has a minimum DO for SPWN of 7.0 mg/L and for waters carrying no specific use, 5.0 mg/L. These three regions also include supplemental criteria for DO as stated below.

> Central Coast Basin Plan: "Median values should not fall below 85 percent saturation as a result of controllable water quality conditions."
>
> Central Valley Basin Plan objective for Tulare Lake: "Waste discharges shall not cause the monthly median dissolved oxygen concentrations (DO) in the main water mass (at the centroid of flow) of streams and above the thermocline in lakes to fall below 85 percent of saturation concentration, and the 95 percentile concentration to fall below 75 percent of saturation concentration." Site-specific minimum DO criteria in this region range from 8.0 mg/L to 9.0 mg/L.
>
> San Francisco Bay Basin Plan: "The median dissolved oxygen concentration for any three consecutive months shall not be less than 80 percent of the dissolved oxygen content at saturation…In areas unaffected by waste discharges, a level of about 85 percent of oxygen saturation exists."

The Los Angeles, North Coast, San Diego, and Santa Ana Basin Plans have set DO criteria for WARM waters to 5.0 mg/L and for COLD waters to 6.0 mg/L. The Los Angeles and North Coast Basin Plans have also set minimum DO concentrations in waters designated as SPWN to 7.0 mg/L. The North Coast Basin Plan further assigns a minimum DO of 9.0 mg/L for SPWN waters during “critical spawning and egg incubation periods.” Site-specific criteria in the North Coast Basin Plan range from 5.0 to 9.0 mg/L (minimum DO concentration). The Los Angeles, San Diego, and Santa Ana Regional Basin Plans have additional requirements as follows:

> Los Angeles Basin Plan: “At a minimum, the mean annual dissolved oxygen concentration of all waters shall be greater than 7 mg/L, and no single determination shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations.”
>
> San Diego Basin Plan: “The annual mean dissolved oxygen concentration shall not be less than 7 mg/L more than 10% of the time.”
>
> Santa Ana Basin Plan: “…waste discharges shall not cause the median dissolved oxygen concentration to fall below 85% of saturation or the 95th percentile concentration to fall below 75% of saturation within a 30-day period.”

The Lahontan Basin Plan follows EPA guidance in setting dissolved oxygen criteria for warm and cold-water uses (USEPA, 1986). During the early life stages SPWN (embryonic, larval, and less than 30 day post hatching), the 1-day minimum dissolved oxygen concentration for cold-water fish is 8.0 mg/L and for warm-water fish is 5.0 mg/L. During other life stages, daily minimum concentrations are 4.0 mg/L and 3.0 mg/L for cold and warm-water fish, respectively. The 7-day mean for early life stages of cold-water fish is 9.5 mg/L and for warm-water fish is 6.0 mg/L. For other life stages, means are calculated over a 30-day period and are 6.5 mg/L for cold-water fish and 5.5 mg/L for warm-water fish. For other uses, the minimum dissolved oxygen concentration should not be less than 80 percent of saturation. Site-specific criteria in the Lahontan are the more restrictive of either a minimum dissolved oxygen concentration of 7.0 mg/L or not less than 10 percent below 80 percent saturation (Susanville, Little Truckee River, Truckee River, West Fork Carson, and East Fork Carson).

Typically, dissolved oxygen concentrations above 5 mg/L are protective of most aquatic life uses. The State of Colorado has a one-day minimum DO concentration of 6 mg/L to protect cold-water fishes and 7 mg/L to protect spawning habitats (CDPHE, 2004b). Water supply and recreational uses have a suggested minimum DO concentration of 5 mg/L for general protection.

3.2.7 DOC and Trihalomethanes

Trihalomethanes (THMs) are byproducts of drinking water treatment that result from the chlorination or bromination of certain organic compounds particularly dissolved organic compounds (DOC). THMs include a number of known and suspected carcinogens, creating concern for drinking water safety. Algal metabolites and decomposition products present in raw water are candidates for THM production (USEPA, 2000b; Nigel, et al., 1998; Plummer and Edzwald, 2001). In a recently proposed rule, the USEPA (2003) suggests a maximum contaminant level of 0.080 mg/L total THM at any point in the water distribution system.

The DOC content of natural waters can be increased by algal production; however, in most cases, the total DOC supply is dominated by loading of organic compounds from the watershed. It is unlikely that response criteria for nutrients would be defined directly in terms of DOC or specific THM precursors. However, information from water treatment system operators on acceptable levels of algae in raw water consistent with meeting THM guidelines could be an important input to the determination of a chlorophyll *a* target for MUN water uses. Despite the importance of DOC and TOC in terms of potential adverse impacts on water quality the complexity of the relationship between water column nutrient concentrations and DOC reduces the feasibility of using DOC as a secondary indicator for developing nutrient numeric endpoints.

3.3 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 effects of velocity on growth..

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. In-depth reviews of periphyton related to growth controlling factors and water quality constituents are provided in USPEA (2000a) and Welch and Jacoby (2004).

3.3.1 Stream Benthic Algal Biomass

As discussed in Section 3.2.1, the nine California Regional Water Boards have narrative criteria regarding biostimulatory substances: “Waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growths cause nuisance or adversely affect beneficial uses.” They do not, however, specify what levels of algal growth constitute a nuisance.

The San Diego Regional Water Board places numerical limits on biostimulatory substances by explicitly limiting total phosphorus concentrations in streams to 0.05 mg/L at the point of entry to a body of standing water. The general limit for streams or other flowing waters is 0.1 mg-P/L. These objectives are not to be exceeded more than 10% of the time, excluding approved exceptions.

There is consensus in the literature that nuisance conditions can be expected if levels of benthic algal biomass exceed 100 to 200 mg chl-*a*/m^2 (Welch et al., 1988; Dodds et al., 1998; Sosiak, 2002; Dodds and Welch, 2000; USEPA, 2000a, Biggs 2000a). Aesthetic nuisance conditions are caused by the fraction of stream surface covered by visible periphyton mats, especially filamentous green algae and in particular *Cladophora*.

Both recreational use categories (REC-1 and REC-2) include aesthetics, fishing, and wading activities. High levels of benthic biomass are aesthetically displeasing and may present a hazard during instream foot travel for fisherman, hikers, etc. Biomass levels above 100 mg chl-*a*/m^2 typically relate to more than 20 percent coverage of bottom surface area with filamentous green algae (Welch et al., 1988). Biggs (2000a) found that 120 mg chl-*a*/m^2 related to 20% cover by filamentous greens. Several species of filamentous greens, especially *Cladophora*, represent a risk to invertebrate communities as well (Biggs, 2000a). Therefore, for recreation and most other uses, I/II boundary of seasonal maximum benthic algal biomass of 100 mg chl-*a*/m^2 is suggested with a II/III boundary of 150 mg chl-*a*/m^2.

These levels have been used by other agencies to protect aesthetic and aquatic community uses. For example, in British Columbia, the algae biomass objective is 100 mg chl-*a*/m^2 for streams designated for aquatic life uses. For recreational-use streams, the objective is 50 mg chl-*a*/m^2. Seasonal biomass levels of 100 and 150 mg chl-*a*/m^2, as mean and maximum, respectively, are used in the Clark Fork River, Montana (Watson and Gestring, 1996) and a maximum of 150 mg chl-*a*/m^2, along with a cover 0f 40% is used as a guideline to protect aesthetics and fishing in New Zealand (Quinn, 1991). This is consistent with results of Lohman et al. (1992) who found that average benthic biomass typically exceeded 150 mg chl-*a*/m^2 at sites considered highly enriched and was < 75 mg chl-*a*/m^2 at unenriched sites.

On a broader scale, Dodds et al. (1998) have suggested a classification of stream trophic state based on frequency distributions of chlorophyll *a*, nitrogen, and phosphorus concentrations in 200 temperate streams in North America and New Zealand. Oligotrophic streams capable of supporting cold-water fishes had mean biomass levels less than 20 mg chl-*a*/m^2 and maximum chlorophyll *a* concentrations less than 60 mg chl-*a*/m^2. Eutrophic streams capable of supporting warm-water fishes had a suggested mean benthic biomass of 70 mg chl-*a*/m^2 or greater and maximum of 200 mg chl-*a*/m^2. Information is not available on levels that protect spawning habitats. Spawning limitations should be protective of the cold or warm water fish species present to ensure proper food supply during subsequent life stages, which should be ensured if biomass does not exceed the suggested criteria..

Water supply plants with intakes from lotic systems are typically not impacted (clogged filters, taste/odor problems) by benthic biomass at concentrations less than 600 mg chl-*a*/m^2 (Welch et al., 1988). For general nuisance control, the suggested mean benthic algal biomass criteria is 200 mg chl-*a*/m^2. The maximum limit is 600 mg chl-*a*/m^2.

Several TMDLs in California have incorporated response indicators into water quality criteria (Tetra Tech, 2003). Two nutrient TMDLs in California (Malibu Creek and Calleguas Creek) have benthic algal biomass criteria of 150 mg chl-*a*/m^2. Malibu Creek is designated for water contact recreation, non-contact recreation, warm freshwater habitat, cold freshwater habitat, wildlife habitat, rare/threatened/endangered species, spawning/reproduction/development habitat, and estuarine habitat. Calleguas Creek is designated for water contact recreation, non-contact recreation, warm freshwater habitat, cold freshwater habitat, wildlife habitat, rare/threatened/endangered species, industrial service and process supply, agricultural supply, and groundwater recharge.

3.3.2 Macrophyte Density in Streams

Growth of aquatic macrophytes is affected by light availability, by flow velocity (positively by providing greater exchange at the leaf surface, negatively by scour), by temperature, and by carbon availability. Like other plants, aquatic macrophytes also require nutrients for growth. However, attempts to predict macrophyte response to water column nutrient concentrations are fraught with difficulties. In a review article, Carr et al. (1997) summarize the issues as follows:

> There has been a long-standing debate in the literature regarding the importance of phosphorus (P) and nitrogen (N) in regulating macrophyte production in freshwater, with some researchers maintaining that these nutrients are important growth-limiting factors…. and other suggesting that high P and N background concentrations and growth limitations due to light and carbon availability preclude phosphorus and nitrogen as being influential in plant growth…. Nevertheless, there are numerous reported cases of luxuriant plant growth in culturally-eutrophicated waters…indicating that resource managers could make valuable use of models that include nutrient-mediated growth rates. The fact that rooted macrophytes in lakes and rivers incorporate varying amounts of P and N from both the open-water and bottom sediments…further confounds attempts to develop models describing nutrient-mediated growth.

Generally, models have met with more success in estimating response to nutrients of periphytic algae, such as *Cladophora*. However, even for such species the role of nutrient storage in plant biomass may confound relationships to ambient nutrient concentrations. Further, if flow conditions are such as to allow significant growth of planktonic algae or periphytic algae on macrophytes, the resulting feedback of reduced light availability may serve to suppress growth of submersed macrophytes.

Physiologically based models of macrophyte growth developed by USACE (e.g., Best and Boyd, 2003) have met with some success in modeling individual species, but generally assume that nutrients are not a limiting factor. The ability of models to predict the effects of changes in nutrient loading on macrophyte biomass appears to be largely uncertain and untested at this time – although there clearly are effects of cultural eutrophication as sediments become enriched (see Welch and Jacoby, 2004). A recent comparative study by Barendregt and Bio (2003) suggests that anthropogenic nutrients play a more important role in determining the community composition of macrophytes than in determining total biomass.

Given the current state of the science, it appears that macrophyte density is not useful as a general numerical effect measure to set quantitative TMDL targets or criteria in streams. Effects of cultural eutrophication on macrophytes may be significant in individual streams, and analysis will likely need to be done on a site-specific basis or through surrogate variables, such as potential periphytic algal biomass. For example, dense macrophyte beds largely disappeared from a 10-km stretch of the Bow River, downstream of Calgary, Alberta, following removal of nitrogen and phosphorus from wastewater (Sosiak, 2002).

3.3.3 Water Clarity and Turbidity in Streams

Planktonic (floating) algae in streams can increase turbidity and decrease water clarity. The general objective for turbidity in each region of California is to limit changes in turbidity that will cause nuisance or adversely affect beneficial uses. In the Colorado Region, no further objective is stated. The remaining eight regions limit percent increases in turbidity by varying degrees.

The Lahontan Region states that increases in turbidity should not exceed natural levels by more than 10 percent. The North Coast Region limits the increase to 20 percent. In the San Francisco Region, in areas where the naturally occurring turbidity is greater than 50 nephelometric turbidity units (NTU), increases due to discharge should not be greater than 10 percent above background.

In the Los Angeles Region, in areas where natural turbidity is between 0 and 50 NTU, increases in turbidity should not exceed 20 percent. Areas with a natural turbidity greater than 50 NTU, should not have increases greater than 10 percent.

In the Santa Ana and San Diego Regions, areas where natural turbidity is between 0 and 50 NTU should not have increases in turbidity greater than 20 percent. If natural background turbidity is between 50 and 100 NTU, increases should not exceed 10 NTU. Areas with a natural turbidity greater than 100 NTU should not have increases greater than 10 percent. The Central Coast Region has the same objectives, but specifies turbidity in JTU rather than NTU. The Tulare Lake Region specifies similar criteria (as NTU), but adds a requirement that waters with natural turbidity between 0 and 5 NTU should not have increases greater than 1 NTU.

Site-specific turbidity limits have been set in the Lahontan Region. Waters in the Little Truckee River and Truckee River Hydrologic Units should not have a mean of monthly means greater than 3 NTU. Clarity requirements for Lake Tahoe state that when water is too shallow to determine a reliable light extinction coefficient, turbidity should not exceed 3 NTU. In shallow waters not directly influenced by stream discharges, turbidity should not exceed 1 NTU. Waters in the West Fork Carson River Hydrologic Unit should not have a mean of monthly means greater than 2 NTU.

Though algae contribute to turbidity in streams, suspended sediment and dissolved organic compounds are usually the more important causes. In fast-flowing or shaded streams, planktonic algal concentrations are usually low and not significant relative to other sources of turbidity. In some slow moving rivers, however, algal blooms may significantly increase turbidity. Because the basin plan objectives are defined in terms of increases in turbidity, the increase due to algal growth could also be used as a response target.

3.3.4 Toxicity of Nitrogen Species

Various forms of nitrogen have proven or suspected toxic effects on aquatic life. In the case of ammonia, the unionized fraction (NH_3) is the toxic form and varies with pH. There are national ambient water quality criteria for unionized ammonia. The USEPA updated its ammonia criteria for freshwater aquatic life based on ambient pH and water temperature in 1999 (USEPA, 1999b). Tables in the document summarize the acute Criteria Maximum Concentration (CMC) and the chronic Criteria Continuous Concentration (CCC) over the pH range of 6.5 to 9.0.

Over a three-year period, one-hour average ammonia concentrations should not exceed the CMC more than once, nor should 30-day average concentrations exceed the CCC. In addition, the highest four-day average within 30 days should not exceed 2.5 times the CCC.

These unionized ammonia criteria constitute direct targets for nutrient TMDLs (as measures of exposure), and do not require an analysis of measures of effect.

Nitrite can be toxic to both humans and aquatic life by reducing the capacity of hemoglobin to carry oxygen. However, ritrate is more common in the environment because nitrite is quickly converted to nitrate under aerobic conditions. Human babies are particularly sensitive to nitrate commonly found in drinking water because their immature digestive systems convert nitrate to nitrite with no counter regulation. The nitrite then enters the blood stream, binds to hemoglobin, and decreases the amount of oxygen transported by the blood. To protect drinking water supplies (MUN), the USEPA has set the nitrate limit to 10 mg-N/L and the nitrite limit to 1 mg-N/L (USEPA, 2002).

There is debate as to what levels of nitrate are toxic to aquatic animals. However, there is always the possibility that nitrate will be converted to nitrite in anaerobic environments by denitrifying bacteria. Most studies indicate that both nitrite and nitrate are toxic to fish at levels that vary by species and life stage. Nitrite is considered generally toxic to fish at 0.1 mg-N/L to 0.6 mg-N/L (ADDL, 1998). Chronic nitrate toxicity to amphibian and salmonid embryos has been observed at levels as low as 1.1 mg N/L (Kincheloe et al., 1979; Marco et al., 1999; Crunkilton, 2000; Krottje and White, 2003). These are general criteria that should be modified on a species-specific basis if necessary. For waters with high chloride content, the State of Colorado has developed equations to account for the buffering affects of chloride to nitrite toxicity for fish (CDPHE, 2004b).

There is thus a possibility of developing ambient water quality criteria to protect aquatic life from exposure to nitrate/nitrite. If such criteria were developed, they would be directly applicable measures of exposure for nutrient assessments. In the absence of such criteria, analysis of the corresponding measure of effect (toxicity) on a site-specific basis may be warranted.

3.3.5 pH

Photosynthesis and respiration impact the pH of an aquatic system by altering the CO_2 balance and pH can exceed 10 in poorly buffered, nutrient-enriched waters (as explained in Section 3.2.5). The USEPA has set criteria for pH for water supply resources and aquatic life (USEPA, 1986). For the protection of water supply processing equipment, the pH should be maintained within the range of 5.0 to 9.0. For the protection of freshwater aquatic life, the range is 6.5 to 9.0.

The water quality objectives for hydrogen ion concentration vary by region in California. In the Central Coast Region, the objective for COLD, WARM, or general use waters is a pH range of 7.0 to 8.5. Waters classified MUN, REC-1, or REC-2 have an allowable range of 6.5 to 8.3. The Tulare Lake Region sets the pH objective to 6.5 to 8.3 for all water uses with the additional requirement that normal ambient pH should not be changed by more than 0.3 unit. In the San Diego Region, the allowable range of pH is 6.5 to 8.5. Waters specified COLD or WARM should not have a change in ambient pH greater than 0.5 unit. The allowable range in the Colorado Region is 6.0 to 9.0, and no discharge should cause a pH change detrimental to beneficial water uses. The Santa Ana, San Francisco, Los Angeles, Lahontan, and North Coast Regions all specify an allowable pH range from 6.5 to 8.5. The San Francisco and Los Angeles Regions do not allow changes of more than 0.5 unit from normal ambient pH due to discharge. The Lahontan and North Coast Regions limit changes in the pH (regardless of cause) to 0.5 unit for all waters designated COLD or WARM.

Site-specific objectives in the North Coast Region range from 6.5 to 9.0 for freshwaters. In the Lahontan Region, site-specific objectives generally state that a change in normal ambient pH greater than 0.5 units is unacceptable (Little Truckee River, Truckee River, West Fork Carson River, and East Fork Carson River). In Eagle Lake, the pH of the hypolimnion should not be less than 7.6, and all other Eagle Lake waters must not have a change in normal ambient pH greater than 0.1 unit. At Honey Lake, the pH should be no less than 8.0 and no more than 10.0 based on the average of at least three samples collected from three different locations. Finally, in Lake Tahoe, the allowable pH range is 7.0 to 8.4.

3.3.6 Dissolved Oxygen

Dissolved oxygen criteria also vary from region to region. Generally, dissolved oxygen concentrations above 5 mg/L are protective of most aquatic life uses. However, cold-water fishes require higher DO concentrations as do all species in stages of early development. The most stringent criteria are set in the Colorado River Basin, where the minimum allowable DO is 5.0 mg/L for WARM uses and 8.0 mg/L for COLD uses.

The Tulare Lake, Central Coast, and San Francisco Bay Regions all assign minimum DO for WARM waters to 5.0 mg/L and for COLD waters to 7.0 mg/L. The Central Coast Region also has a minimum DO for SPWN of 7.0 mg/L and for waters carrying no specific use, 5.0 mg/L. These three regions also include supplemental criteria for DO as stated below.

> Central Coast: “Median values should not fall below 85 percent saturation as a result of controllable water quality conditions.”
>
> Tulare Lake: “Waste discharges shall not cause the monthly median dissolved oxygen concentrations (DO) in the main water mass (at the centroid of flow) of streams and above the thermocline in lakes to fall below 85 percent of saturation concentration, and the 95 percentile concentration to fall below 75 percent of saturation concentration.” Site-specific minimum DO criteria in this region range from 8.0 mg/L to 9.0 mg/L.
>
> San Francisco Bay: “The median dissolved oxygen concentration for any three consecutive months shall not be less than 80 percent of the dissolved oxygen content at saturation…In areas unaffected by waste discharges, a level of about 85 percent of oxygen saturation exists.”

The Los Angeles, North Coast, San Diego, and Santa Ana Regions have set DO criteria for WARM waters to 5.0 mg/L and for COLD waters to 6.0 mg/L. The Los Angeles and North Coast Regions have also set minimum DO concentrations in waters designated as SPWN to 7.0 mg/L. The North Coast Region further assigns a minimum DO of 9.0 mg/L for SPWN waters during “critical spawning and egg incubation periods.” Site-specific criteria in the North Coast Region range from 5.0 to 9.0 mg/L (minimum DO concentration). The Los Angeles, San Diego, and Santa Ana Regions have additional requirements as follows:

> Los Angeles: “At a minimum, the mean annual dissolved oxygen concentration of all waters shall be greater than 7 mg/L, and no single determination shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations.”
>
> San Diego: “The annual mean dissolved oxygen concentration shall not be less than 7 mg/L more than 10% of the time.”
>
> Santa Ana: “…waste discharges shall not cause the median dissolved oxygen concentration to fall below 85% of saturation or the 95th percentile concentration to fall below 75% of saturation within a 30-day period.”

The Lahontan Region follows EPA guidance in setting dissolved oxygen criteria for warm and cold-water uses (USEPA, 1986). During the early life stages SPWN (embryonic, larval, and less than 30 day post hatching), the 1-day minimum dissolved oxygen concentration for cold-water fish is 8.0 mg/L and for warm-water fish 5.0 mg/L. During other life stages, daily minimum concentrations are 4.0 mg/L and 3.0 mg/L for cold and warm-water fish, respectively. The 7-day mean for early life stages of cold-water fish is 9.5 mg/L and for warm-water fish is 6.0 mg/L. For other life stages, means are calculated over a 30-day period and are 6.5 mg/L for cold-water fish and 5.5 mg/L for warm-water fish. For other uses, the minimum dissolved oxygen concentration should not be less than 80 percent of saturation. Site-specific criteria in the Lahontan are the more restrictive of either a minimum dissolved oxygen concentration of 7.0 mg/L or not less than 10 percent below 80 percent saturation (Susanville, Little Truckee River, Truckee River, West Fork Carson, and East Fork Carson).

Water supply and recreational uses have a suggested minimum DO concentration of 5 mg/L for general protection.

Algae produce oxygen during photosynthesis and consume oxygen during respiration. In oligotrophic and mesotrophic systems, oxygen production is typically greater than consumption, and algae help maintain dissolved oxygen concentrations for use by other organisms. However, in eutrophic systems, nighttime respiration may cause DO concentrations to drop below critical levels. Eventual bloom die-off can also cause large DO deficits as organic matter is consumed. Water quality criteria for DO should specify allowable deficits from the saturation concentration and require pre-dawn sampling to capture concentrations when they are typically lowest.

3.4 Summary of Measures of Effect

The State Water Board Nutrient Numeric Training Workshop held on May 18 & 19, 2005 in Sacramento, CA provided representatives from Federal, State, and Tribal resource management agencies the opportunity to review and comment on the draft California approach to setting nutrient numeric endpoints. The workshop outcome included agreement on the following elements:

- The use of risk categories to classify water bodies into three levels of beneficial use status: presumptive unimpaired; potentially impaired; and presumptively impaired;
- The use of secondary indicators (described above) to assess the status of water bodies for assignment to the appropriate risk category;
- The use of quantitative models to conduct the linkage analysis to determine the nutrient concentrations necessary to maintain the secondary indicators within an acceptable range;
- The development of statewide monitoring guidance for the measurement of secondary indicator variables classify water bodies; and
- The maintenance of a statewide database of nutrient numeric endpoint assessments that will be used to update and refine the risk category classification boundaries.

Workshop participants provided direction to the technical project team to use available information to propose beneficial use risk category boundary values for as many of the secondary indicators as possible. The results of this synthesis are provided in Table 3-2, which summarizes the measures of effect discussed in this document. For simplicity, references are not attached to the table, but can be found in each relevant section of the document and in the annotated bibliographic table (Appendix 2).

Table 3-2
Nutrient Numeric Endpoints for Secondary Indicators - Proposed Risk Classification Category Boundaries: I & II and II & III

Beneficial Use Risk-Category I. Presumptive unimpaired (use is supported)
Beneficial Use Risk Category II. Potentially impaired (may require an impairment assessment)
Beneficial Use Risk Category III. Presumptive impaired (use is not supported or highly threatened)

RESPONSE VARIABLE	RISK – CATEGORY BOUNDARY	BENEFICIAL USE						
		COLD	WARM	REC-1	REC-2	MUN[1]	SPWN	MIGR
Benthic Algal Biomass in streams (mg chl-*a*/m^2) Maximum	I / II	100	150	**C**	**C**	100	100	**B**
	II / III	150	200	**C**	**C**	150	150	**B**
Planktonic Algal Biomass in Lakes and Reservoirs (as µg/L Chl-*a*)[2] – summer mean	I / II	5	10	10	10	5	**A**	**B**
	II / III	10	25	20	25	10	**A**	**B**
Clarity (Secchi depth, meters.)[3] – lakes summer mean	I / II	**A**	**A**	2	2	**A**	**A**	**B**
	II / III	**A**	**A**	1	1	**A**	**A**	**B**
Dissolved Oxygen (mg/l) Streams – the mean of the 7 daily minimums	I / II	9.5	6.0	**A**	**A**	**A**	8.0	**C**
	II / III	5.0	4.0	**A**	**A**	**A**	5.0	**C**
pH maximum – photosynthesis driven	I / II	9.0	9.0	**A**	**A**	**A**	**C**	**C**
	II / III	9.5	9.5	**A**	**A**	**A**	**C**	**C**
DOC (mg/l)	I / II	**A**	**A**	**A**	**A**	2	**A**	**A**
	II / III	**A**	**A**	**A**	**A**	5	**A**	**A**

A = No direct linkage
B = More research needed to quantify linkage
C = Addressed by Aquatic Life Criteria

[1] For application to zones within water bodies that include drinking water intakes.
[2] Reservoirs may be composed of zones or sections that will be assessed as individual water bodies
[3] Assumes that lake clarity is a function of algal concentrations, does not apply in waters of high non-algal turbidity

4 Next Steps

This document represents another significant milestone in the process to develop and implement the California approach for nutrient numeric endpoints. The purpose of this section is to describe several recommendations for continuing refinement of the California approach. These are described below.

Recommendation 1: The State Water Board sponsored two technical training workshops on March 27, 2006 at the North Coast Regional Water Board in Santa Rosa and on March 29, 2006 at the Los Angeles Regional Water Board. Staff from all nine Regional Board offices attended the training. The purpose of the training was to provide the opportunity for agency staff to become familiar with the nutrient endpoint framework and to learn how to use the linkage models described in Appendices 3 and 4 of this document. These workshops will allow staff from all nine Regional Water Boards to use the approach to develop nutrient numeric endpoints for TMDLs and NPDES permits limits. Finally, the workshops provided a forum to discuss review comments received on this document and to ensure support for technical responses. Following the workshops technical support is being provided through EPA Region IX to apply the CA NNE framework on up to eight pilot waterbodies throughout the state. These waterbodies either have a TMDL currently underway or have completed the TMDL process. The CA NNE framework will be applied to either develop initial numeric targets or to update existing targets.

Recommendation 2: Supply information of nutrient numeric endpoints into SWAMP and CIWQS California State Water Boards databases. They will serve as valuable functions in the implementation process including:

- Reference for other ongoing endpoint studies; and
- Use as a repository for refinement of the approach.

Recommendation 3: If the Regional Water Boards approve the approach they should consider development of draft evaluations of water body status based on the proposed BURC. This may require additional monitoring of the parameters identified as secondary indicators (e.g., stream benthic algal biomass).

Recommendation 4: Develop monitoring guidance for all secondary indicator parameters and procedures for conducting BURC 2 impairment assessments.

Recommendation 5: Regional Water Boards could use the regional database as a resource to identify key trends and patterns among affected water bodies allowing possible Basin Plan amendments for addressing nutrient criteria and implementation. This option would be at the discretion and direction of each individual Regional Water Board.

The California approach lays out a realistic and technically defensible method for developing nutrient numeric endpoints. It provides the flexibility to allow for regional differences and timetables. The approach can be used as a focused water quality program tool for TMDLs or NPDES permit limits. The approach could also over time be used to develop broader water quality objectives for use in Basin Plans. The timetable for implementation of the approach will be largely driven by its success in developing technically defensible nutrient numeric endpoints that provide protection for affected water bodies and that lead to sound nutrient management and control programs.

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APPENDIX 1
Nutrient Numeric Endpoint Technical Advisory Committee

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APPENDIX 1 Nutrient Numeric Endpoint Technical Advisory Committee

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APPENDIX 2
ANNOTATED BIBLIOGRAPHIC TABLE

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APPENDIX 2 ANNOTATED BIBLIOGRAPHIC TABLE

How to read the table: For each study reviewed, the table lists the waterbody use that was addressed and the associated nutrient concentration, chlorophyll *a* concentration, or turbidity measurement. Reading across the row for each study, you generally will find that the listed use impairment occurred when the listed parameter value was exceeded (or not exceeded, in the case of Secchi depth) in the study. For example, when TP was greater than 30 ug P/L or if there were greater than 10 filaments of *Oscillatoria tenuis*/mL, then Nakanishi et al. (1999) concluded that the drinking water use was negatively affected. At the end of the table are a few studies that relate chlorophyll *a* and nutrient concentrations but did not indicate any specific use impairment.

** This table is only provided as a starting point to understand relationships between causal and response variables and designated uses. The information provided in the table should be used with caution. Values taken from the literature may have associated limitations that are not noted in the comments section. User Beware!

Designated Use	Water Body Type	N (μg/L)	P (μg/L)	Chl a (mg/m^2 or μg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Drinking water/ Aquatic life, growth of musty odor-producing algae supported			>30 μg/L TP	>10 filaments of *Oscillatoria tenuis*/ml				Japan	Nakanishi et al. 1999	Based on correlation between P concentrations and maximum standing crop (filaments/ml) of O. tenuis, a musty odor-producing algae.
Drinking water, production of odor-producing algae	Reservoirs							*Kansas*	Arruda and Fromm 1989	A panel ranked odors of 6 Kansas reservoirs and correlated odor rankings with TSI (trophic state index). Found a positive correlation between the two parameters (r=0.81, p=0.05), but a TP threshold could not be determined from this study
Drinking water, production of trihalomethanes	Reservoirs		>24 μg/L						Aruda 1988 (as cited in NC guidance manual-Lakes)	Trihalomethane concentration exceeded 100 mg/L at a TSI of 50, which can be related to the TP concentration listed.
Human consumption, toxicity		10,000 μg/L NO_3							USEPA 2004	
Aquatic life, toxicity (acute)		30-5000 μg/L NH_3							Russo 1985	Based on fish and invertebrate data.
Aquatic life, toxic to amphibians		>3000μg/L NO_3						S. Ontario	Hecnar 1995	96-hr LC50 tests showed physical and behavioral abnormalities in tadpoles

Designated Use	Water Body Type	N (μg/L)	P (μg/L)	Chl a (mg/m^2 or μg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
STREAMS/ RIVERS										
Aquatic life, decreased biotic integrity using fish and invert indices (inc in tolerant and omnivore spp, dec in EPT taxa)	Streams, headwater and wadeable	1370 μg/L inorg-N, >1000 NH_3-N	*170 μg/L TP*					Ohio	Miltner and Rankin 1998	Numbers represent exceedance of 50th percentile of all nutrient concentrations. No relationship for large rivers was found. Large data set used to develop regression models.
Aquatic life, 50% decline in "clean water" EPT invertebrates (and increase in rel abundance of chironomids and oligochaetes)	Streams, New Zealand			>13-20 mg/m^2 (mean monthly)				*New Zealand; 21 streams*	Biggs 2000, Ministry of Environment	Based on diatom/cyanobacteria assemblages. Low biomass does not suggest only EPT will be dominant, but high proportions of EPT were only found where biomass was low. Recommends max does not exceed 50mg/m^2 based on average peak biomass values of 16 streams with diverse benthic communities.
Aquatic life, shift in invertebrate community composition	Streams, New Zealand			>100 mg/m^2				New Zealand; 4 streams	Biggs 2000, Ministry of Environment	Based on diatom/cyanobacteria assemblages. 20 mg/m^2-(oligotrophic)-stone-, may-, and caddisflies are dominant; 100 mg/m^2 (mesotrophic)-may-, caddisflies, midge, and beetle larvae dominant; 600 mg/m^2 (eutrophic)-snails, midge and beetle larvae, and oligochaetes dominant; Based on relative abundances; Chl levels based on 90th percentile of values in each trophic group.

Designated Use	Water Body Type	N (μg/L)	P (μg/L)	Chl a (mg/m^2 or μg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aquatic life, shift in energy source altered food web structure	Stream, 4th order clearwater tundra; channel slope = 3%, drainage area (DA) = 143 km^2		Enriched with 10μg/L PO_4-P	5 mg/m^2 increased to 60 mg/m^2				Alaska	Peterson et al. 1985; Deegan and Peterson 1992	Enriched stream with PO_4-P (10μg/L) and increased SRP concentrations 10-fold. Stream changed from heterotrophy to autotrophy, algal biomass increased (by a factor of 10), diatom richness decreased; growth of Prosimuliids also increased compared to upstream control. Fertilization resulted in a 1.4 to 1.9-fold increase in size of age 0+ grayling and a 1.5-2.4-fold increase in weight gain of adult grayling.
Aquatic life, protection of trout habitat	*Streams, NZ*			~200 mg/m^2 (diatom communities) ~120 mg/m^2 (filamentous communities)				New Zealand	Biggs 2000, Ministry of Environment	Recommendations based on potential for fish kills; potential impairment may increase with duration of low flows and increases in temperature.
Aquatic life, trout biomass increases from oligotrophic to mesotrophic streams, but declined threefold in eutrophic streams	Streams							New Zealand	Quinn and Hickey 1990	Trophic status determined by percent of catchment developed for agriculture: oligo-<1%, meso-1-30%, and eutrophic->30 %.

Designated Use	Water Body Type	N (μg/L)	P (μg/L)	Chl a (mg/m^2 or μg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aquatic life, algal diversity decreased, nuisance growth increased	Rivers, temperate, lowland, DA range =400-90,000 km^2; rocky substrate		>20μg/L TP					S. Ontario and W. Quebec	Chetelat et al. 1999	Periphyton communities with TP <20μg/L had the highest diversity of algal taxa, but was not analyzed statistically. Cladophora (accounted for >65% of green algal biomass), Audouinella (red filamentous), and/or Melosira (diatom) dominated when TP >20g/L. The 3 genera above were positively correlated to an increasing TP.
Recreation, *nuisance levels*	Streams, NZ			~120 mg/m^2 (max)				New Zealand	Biggs 2000 Ministry for the Environment	Based on the relationship between chl a and percent of the substrate covered by filamentous algae. 120 mg/L is about 30 % cover by filamentous green or brown algae (basis unclear); most relevant to shallow reaches of cobble/gravel streams (<0.75m deep) during periods of recreational use.
Recreation/ Aesthetics	Streams, NZ			100 mg/m^2 and 40% coverage				New Zealand	Quinn, 1991	Values to protect contact recreation beneficial uses
Recreation/ Aesthetics	Streams			100-150 mg/m^2				National?	Horner et al. 1983	Max biomass to avoid recreational and aesthetic use impairment; Articles state that levels are based on 19 enrichment studies
Recreation/ Aesthetics, nuisance levels	Rivers, streams	300-350 μg/L	>30 μg/L					Clark Fork River, MT (some data from NA, Europe, and NZ)	Dodds et al. 1997	Guidelines for Clark Fork River, MT based on reference, probabilistic, and regression approaches that led to similar conclusions. Nuisance level set at 100 mg/m^2 (mean) and 150 mg/m^2 (max).

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Recreation/ Aesthetics, nuisance levels	Streams			150 mg/m^2 max. limit				Clark Fork River, MT	Watson and Gestring 1996	
Recreation/ Aesthetics/ Aquatic Life	Streams			50-100 mg/m^2				British Columbia	Nordin 1985	Approved water quality criteria for British Columbia (includes criteria for recreation and aquatic life uses). Basis unclear.

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Recreation/ Aesthetics, *nuisance level*	Rivers/ Streams			150-200 mg/m^2				Washington, Spokane River	Welch et al. 1989	Nuisance level based on perceived impairment.
Recreation/ Aesthetics, *nuisance level*	Streams			>150 mg/m^2 at high to moderately enriched sites vs. <75 mg/m^2 at un-enriched sites				Ozarks, US	Lohman et al., 1992	Data from 12 Ozark streams.
Aesthetics	Streams, rocky substrate			>100 mg/m^2				Pac NW (mainly)	Welch et al. 1988	Led to greater than 20% coverage by filamentous algae. Based on correlation of chl and % coverage from 25 streams sites ($r=0.78$). SRP and/or DO were not related to periphyton biomass in these streams (probably due to other limiting factors).
LAKES/ RESERVOIRS										
Aquatic life*, toxic algal blooms*	Lake			Cyanobacteria comprised >80% biovolume; cyanobacteria dominant during bloom				Washington	Jacoby et al. 2000	Found relationships between SD and cyanobacteria and SRP and cyanobacteria: Ln[cyanobacteria]=1.93-1.41SD ($R^2=0.82$); [Microcystin]=199.4+100(surface SRP) ($R^2=0.96$), Toxic algal blooms were associated with TN:TP ratios <30 (Blooms may have been limited by P)

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aquatic life, trout habitat	Lakes			>15 µg/L (unsuitable for trout) *>40 µg/L (severe nuisance)*				North Carolina	McGhee 1983 (cited in Heiskary and Walker 1988)	Based on use impairment classification for North Carolina lakes.
Aquatic life, peak in the relative abundance for: Lake trout Walleye Black crappies White crappie	Lakes		<10 µg/L 25 µg/L 70 µg/L 100 µg/L					Northern U.S.	extrapolated from Schupp and Wilson 1993 (cited in Ney 1996)	Based on comparison of total P to relative abundances of certain sport fish species in natural lakes in the northern U.S. Type of comparison unclear from Ney 1996.
Aquatic life, increased sockeye salmon production	Lakes, clearwater meromictic and holomictic	TKN inc 47 to 91 µg/L (ammonia did not change)	TP sign. increased from 8.0 to 9.8 µg/L, (~20% incr in mean P conc)	*Chl a sign. incr from 0.64 to 2.05 µg/L (220% incr)*	Mean turbidity incr from 4.6 to 4.8 NTU (p=0.067)			Alaska (Coghill Lake)	Edmundson et al. 1997	P and N enrichment led to increases in algal biomass (edible spp), zoops (>100%), and the salmon smolt popn (>300%) after enrichment. Data based on 3 years of pre- and 3 years of post-enrichment data. Attempted to maintain N:P >18:1.
Aquatic life, *Maximum fish biomass*	Reservoirs		>81 ug/L (see explanation)					Appalachian reservoirs	Ney et al. 1990	Based on regression of total fish standing stock (FSS) versus TP for 21 reservoirs. Log FSS=1.24+1.02logTP (r^2=0.84); FSS increased linearly over their TP range (8-81 µg/L) so they suggest that maximum fish production would occur at a higher concentration. Note: regression based on all fish, not just sport fish species.

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aquatic life, max biomass of sport fish	Reservoirs/ Impoundments (>200 ha)		>100 µg/L					Virginia, Arkansas, and Nevada reservoirs	Ney 1996;	Suggests that sport fish biomass does not peak at less than 100µg/L based on correlations and regressions of TP and total fish standing stock and sport fish standing stock (because relationship was linear for the range of values measured; ~3 to 85 µg/L). Recommendation based on Ney's data and literature findings for southeastern reservoirs. Relationship between TP and planktivores (r^2=0.84) was much stronger than for TP and piscivores (r^2=0.51). Suggested that oligotrophication led to summer habitat expansion for bass, which may explain the poorer relationship between piscivores and TP.
Aquatic life, biomass of planktivorous and piscivorous fishes	Reservoirs (>200 ha)		**see comments					Smith Mountain Lake, Virginia	Yurk and Ney 1989	Regression showed linear relationship for TP ~20-120 µg/L. Significantly explained variation in biomass of planktivores (R^2=0.63), but not piscivores (R^2=0.01, p=0.80). May be confounded by changed fish management practices or due to expanded habitat for sport fishes with decreased TP levels.
Aquatic Life, stunted pan fish populations	Lakes		>40-50 µg/L TP		< 1m			?	Lee and Jones 1991	*Levels noted by the authors solely based on their experiences (no data shown to support values).*

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aquatic Life, peak in sportfish yield	Lakes (depth <6 m; Area 6-24000 ha)			*>70 mg/m3*				Missouri and Iowa	Jones and Hoyer 1982	Found positive correlation between fish yield and chl a ($r=0.91$, $N=25$, $p<0.01$). Also found a positive correlation between fish yield and TP ($r=0.72$, $p<0.01$). Data set included natural and artificial lakes, data from multiple years, lakes differed in morphology, hydrology, trophic status, and fish communities. Did not mention which sport fish were present in these systems.
Aquatic Life, greater LM bass growth and harvest	Reservoirs		50-100 µg/L TP					Alabama	Bayne et al 1994	Compared LM bass growth and harvest in eutrophic (50-100 µg/L TP) to mesotrophic (10 µg/L TP) reservoirs

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aquatic Life, Potential water quality stress to fish	Lakes					9.0 (upper limit)	**Warm Water:** 6 mg/l (7-day mean); 4 mg/l (daily minimum) **Cold Water:** 5 mg/l (7-day mean); Early life stages: 9.5 mg/l (7-day mean); 8.0 mg/l (daily minimum)	Klamath Lake, Oregon	Loftus, M.E. 2001.	
Aquatic Life, change in algal community structure to dominance by less edible species	Lakes		>30 µg/L TP					National? Canada?	Watson et al. 1992	Used a wide range of published data from 362 different lakes to develop relationships between algal biomass (edible and inedible) and TP. Found that at TP>30µg/L, inedible algae became dominant (inedible algae defined by size—larger=inedible), whereas edible algae dominated below 8-10 µg TP/L. Paper contains the regr coefficients for TP and chl a relationship. Also suggests that the shift in phytoplankton community structure should lead to shift in herbivore community.

Designated Use	Water Body Type	N (μg/L)	P (μg/L)	Chl a (mg/m^2 or μg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Recreation	Lakes				1.2 m			New York	Effler et al. 1984 (cited in Heiskary and Walker 1988)	State standard for beaches. Basis unclear.

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m² or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Recreation, *impaired swimming*	Lakes		40-60 µg/L	20-40 µg/L	<1 m			Minnesota	Heiskary and Walker 1988	Cross-tabulation of lake response (chl, SD, and P conc) with user response (lake observer survey).
Recreation, *water clarity*	Lakes		30 µg/l	30 µg/l					Lillie and Mason 1983	"Chlorophyll-a appears to have the greatest impact on water clarity when levels exceed 30 µg/l." (pg. 5) "Thirty µg/l of total phosphorus appears as a more reliable predictor of visible chlorophyll a levels (10 µg/l or greater) than 20 µg/l total phosphorus." (pg. 5)
Recreation	Lakes				1.2 m			Mass.	MDPH 1969 (cited in Heiskary and Walker 1988)	State standard for beaches. Basis unclear.

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Recreation/ Aesthetics, nuisance blooms	Lakes	See Key 1 = 578 2 = 637 3 = 744 4 = 785 5 = 2091	See Key 1 = 15 2 = 27 3 = 37 4 = 65 5 = 220	See Key 1 = 7 2 = 12 3 = 14 4 = 17 5 = 80	See Key 1 = 2.3 2 = 2.0 3 = 1.6 4 = 1.7 5 = 0.8			116 Florida Lakes	Hoyer, et al. 2004	Citizen volunteer survey on aesthetic/use impairment based on Total P, Total N, Chl-a, and Secchi depth: **Key:** 1 = Beautiful; 2 = Very minor aesthetic problems, excellent for swimming; 3 = Swimming and aesthetic enjoyment slightly impaired because of algae levels; 4 = Desire to swim and level of enjoyment of the lake substantially reduced because of algae levels; 5 = Swimming and aesthetic enjoyment of the lake nearly impossible because of algae levels.
Recreation/ Aesthetics, nuisance blooms	Lakes, general		>30µg/L	>15 µg/L	<1.5 m			Wisconsin	Lillie and Mason 1983 (cited in Heiskary and Walker 1988)	Aesthetic/use impairment classification based on chl a and secchi depth: <1 µg/L (>6 m) = excellent *1-5 (3-6 m) = very good* 5-10 (2-3 m) = good 10-15 (1.5-2) = fair 15-30 (1-1.5) = poor >30 (<1 m) = very poor Secchi depths are noted in ( ). Based on data from >500 lakes. Contains regression coeff for chl + water clarity and chl + TP. Basis for category "labels" is unclear.
Recreation/ Aesthetics	Lakes		5-15 µg/L TP						Nordin 1985	Approved water quality criteria for British Columbia. Basis unclear.

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aesthetics, water clarity	Lakes		>20 µg/L	*>10 mg/m^3*	*<1.5 m SD*				Bachmann and Jones 1974	Based on relationships between chl *a* and TP and between chl *a* and secchi depth they found that in general, when TP exceeded 20 mg/L then chl *a* values exceed 10 mg/m^3 which led to SD below 1.5 m. Based on ~ 16 lakes (mostly published literature values).
Aesthetics/ Bathing, suitable water clarity	Lakes and Rivers				~1.6 m- bathing ~1.7m- aesthetics (black disc depth)			*New Zealand*	Smith and Davies-Colley 1992	Perceived suitability for bathing/aesthetics is suitable when black disc depth > ~1.6 m. Conditions are marginally suitable when bd visibility >1.0 m. Suitability curves developed from surveys of NZ water resource officers who were asked to draw their perception of the suitability curves.
Aesthetics, nuisance algal blooms	Lakes			>32 µg/L	<0.7 m			Louisiana	Burden et al. 1985 (cited in Heiskary and Walker 1988)	Classification based on mean chl-a and secchi depths for classes: 14 µg/L (1.2m) = Excellent to Good 30 µg/L (0.8 m) = Good to acceptable 32 µg/L (0.7 m) = acceptable to marginal Basis for category "labels" is unclear.

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m² or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
Aesthetics, nuisance algal blooms	Lakes			25-100 µg/L (moderate blooms)	0.4-1 m			CAN prairie ponds	Barica 1975 (cited in Heiskary and Walker 1988)	Aesthetic classification based on chl-a and secchi depth: 0-25 µg/L (>1 m) = clear, no blooms 25-100 (1.4-1) = moderate blooms 100-200 (<0.4) = dense colonies and scums
Economics/ **Aesthetics**, property prices declined with decreased secchi depth	Lakes				<3m			Maine	Michael et al. 1996	Correlations based on limited no. of lakes; property prices were sign different for lakes with SD >6m (highest prices) compared to <3m (lowest prices).
CHL a – NUTRIENT RELATIONSHIPS										
	River/ stream		>1-4 µg/L SRP	>150 mg/m²				Washington, Spokane River	Welch et al. 1989	Relationship based on Horner et al.'s (1983) model using data from Spokane River. Model also incorporates uptake rate, light, and velocity to predict periphytic biomass.

Designated Use	Water Body Type	N (μg/L)	P (μg/L)	Chl a (mg/m^2 or μg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
	River/ stream		~100 μg/L ~500 μg/L 300 μg/L 300 μg/L	16.2 μg/L 48 μg/L 21.6 μg/L ($100\ km^2$ CA) 49.5 μg/L ($100{,}000\ km^2$ CA)				Mainly N.A., some Europe	Van Nieuwenhuyse and Jones 1996	Recommendations based on regression analyses of data from literature; relationship between TP and chl was curvilinear (log Chl = -1.65 + 1.99log TP-0.28LogTP^2, R^2 = 0.67)); more variation accounted for by incorporating catchment area into models (log chl = -1.92 + 1.96(logTP)-0.3(logTP^2) + 0.12(logCA), R^2 = 0.73); based on water column measurements of sestonic algae.
	Streams/ Rivers; runoff fed, most unshaded	~20 μg/L SIN	~2 μg/L SRP	> 200 mg/m^2 (max)				New Zealand, temperate streams	Biggs 2000. JNABS	Recommendation for unshaded streams with accrual periods of > 50 d. Days available for biomass accrual explained as much if not more variation in mean monthly and max chl a than nutrients (SRP and SIN). $\log chl_{max} = -2.946 + 4.285\log d_a - 0.929(\log d_a)^2 + 0.504\log SIN$ (R^2 = 0.74) $\log chl_{max} = -2.714 + 4.716\log d_a - 1.076(\log d_a)^2 + 0.494\log SRP$ (R^2 = 0.72) (d_a = mean days of accrual)

Designated Use	Water Body Type	N (µg/L)	P (µg/L)	Chl a (mg/m^2 or µg/L)	Turbidity (secchi depth)	pH	Dissolved Oxygen (mg/l)	Geographic Location	Source(s)	Additional Comments
	Streams; DA range =8-860 km^2	350µg/L	100 µg/L	18µg/L 4 µg/L				Missouri Ozarks	Lohman and Jones 1999	Measured sestonic chlorophyll; these sites were also used in Van Nieuwenhuyse and Jones 1996 but they made up <10% of the data in their global model; CA = catchment area: Log Chl = -1.15 + 1.20log TP (R^2=0.85) Log Chl = -4.83 + 2.14log TN (R^2=0.65) With catchment area (CA): Log Chl = -1.53 + 0.98log TP + 0.33log CA (R^2=0.94) Log Chl = -4.53 + 1.65log TN + 0.45log CA (R^2=0.84)
	Rivers, temperate lowland; DA range =400-90000km^2; rocky substrate		47 µg/L	100 mg/m^2				Ontario and Quebec	Chetelat et al. 1999	Measured periphyton and TP at 33 riffles (in 13 rivers). Log chla = 0.490 + 0.905 log TP (R^2=0.56)
Other Relationships:										
Drinking Water, relationship btwn TP and TOC (surrogate for trihalomethanes)	*Reservoirs*							United States	Walker 1983	Regression: TOC = 0.56(TP)$^{0.63}$; R^2=0.85. Data based on 34 reservoirs and 3 lakes in the U.S. Only states that TOC can be used as a surrogate measure for THM, but does not state the levels of TOC that would lead to THM levels that exceed US EPA standards.

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APPENDIX 3
BENTHIC BIOMASS SPREADSHEET TOOL USER GUIDE AND DOCUMENTATION

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Appendix 3 Benthic Biomass Spreadsheet Tool User Guide and Documentation

This section provides background and instructions for using the California Benthic Biomass Tool. The tool is a Microsoft Excel spreadsheet, and is intended to be a simple but effective tool for predicting in-stream benthic Algal density and other metrics in response to a number of inputs. The tool calculates algal density as ash free dry weight (g/m^2) and benthic chlorophyll *a*. Both are estimated using a variety of methods as described in Sections 0 and 2 of this appendix.

The maximum algal contribution to dissolved oxygen deficit is also calculated. Lastly, the tool allows the user to supply a target (either algal density or benthic chlorophyll *a*), select a calculation method, and the tool will display a graph of allowable TN and TP to meet the target.

Three basic methods of calculation are included in the tool: Dodds 1997 method (Dodds et al., 1997), Dodds 2002 method (Dodds et al., 2002), and the QUAL2K model method (Chapra and Pelletier, 2003).

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1 Stream Criteria for Periphyton – Dodds' Method

1.1 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 River 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). Total Nitrogen (TN) and Total Phosphorus (TP) were found to be better regressors than inorganic Nitrogen (N) and inorganic Phosphorus (P).

The relationships that were identified were relatively weak, with a maximum adjusted R^2 value of 0.430 in log space (see Figure 1).

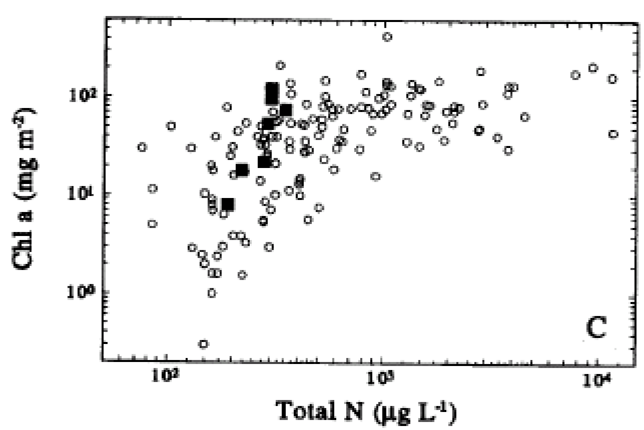

Figure 1. Relationship between Benthic Chlorophyll *a* and Total N shown in Dodds et al. (1997; Figure 2C)

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. dataset might result in a similar improvement in predictive ability.

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

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

and

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

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 1.

Table 1. 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	1,500
TP (µg/L)	25	75

The values shown in Table 1 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 87th 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 2,500 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:

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

and

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

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 µg/L and TP of 42 µg/L, both slightly higher than the amounts (275 and 35) estimated with the earlier regression.

1.2 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.

1.2.1 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 were 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.

Figure 2 and Figure 3 plot the RWQCB 6 benthic chlorophyll *a* observations against TN and TP, respectively. There appears to be a positive correlation with Total Nitrogen 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.

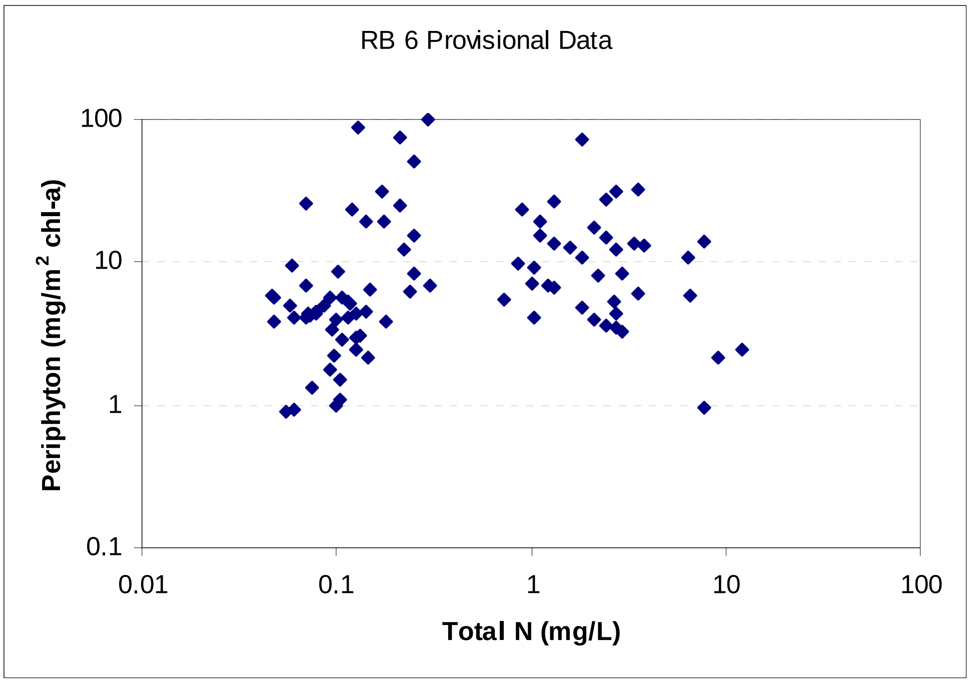

Figure 2. Provisional RWQCB 6 Benthic Chlorophyll *a* Observations vs. Total Nitrogen

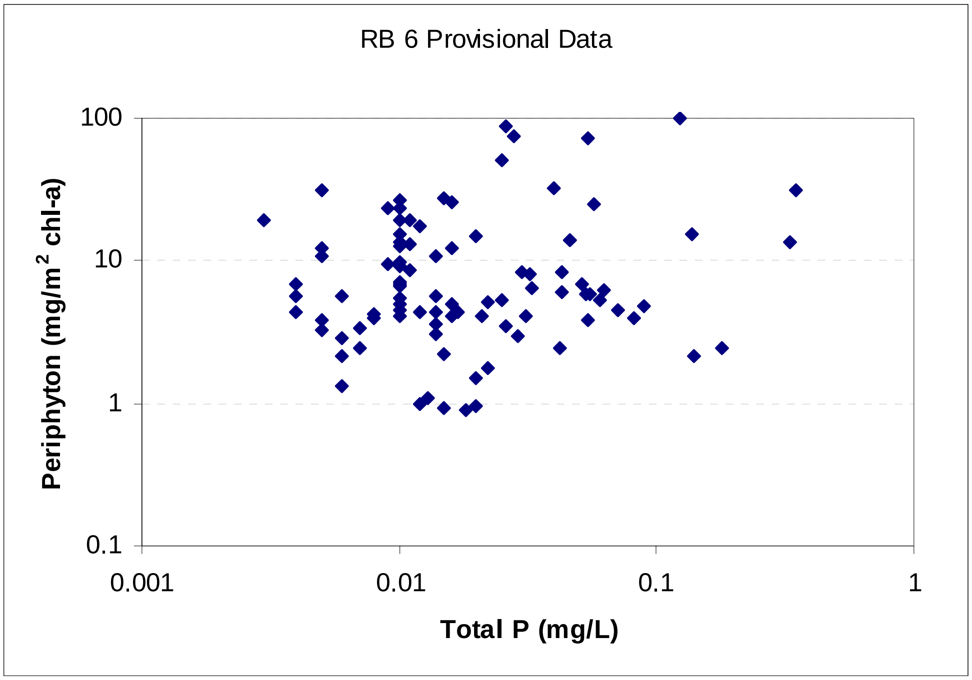

Figure 3. Provisional RWQCB 6 Benthic Chlorophyll *a* Observations vs. Total Phosphorus

None of the observations in the data set exceeded 100 mg/m^2 despite TP 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. Chlorophyll *a* density predicted by this equation is lower than the result of 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, 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.

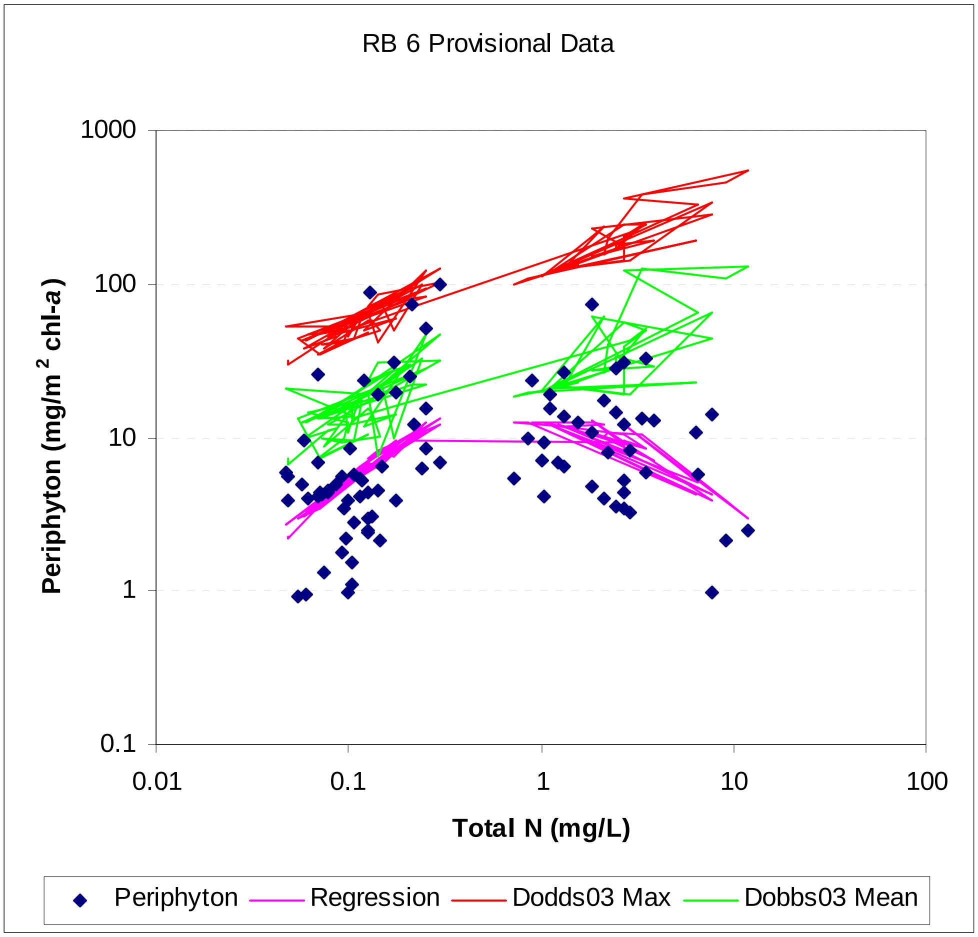

Figure 4. SWQCB 6 Periphyton Chlorophyll *a* Compared to Various Regression Equations

The relation of the SWQCB 6 data to the Dodds et al. (2002) equation for mean chlorophyll *a* is further explored in Figure 5, 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 6 compares the data to the maximum benthic algal regression from Dodds et al. (2002). In all but one case, the predictions are greater than the observations, suggesting that the Dodds equation does indeed provide an upper bound.

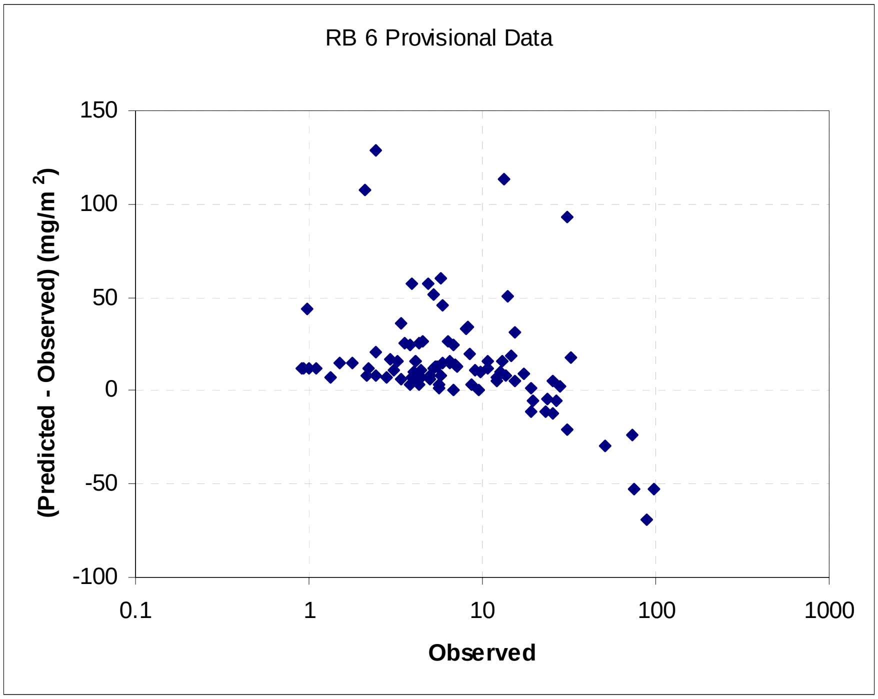

Figure 5. Deviations of RWQCB 6 Provisional Benthic Chlorophyll *a* Data (mg/m^2) from Mean Predictions using Dodds et al. (2002)

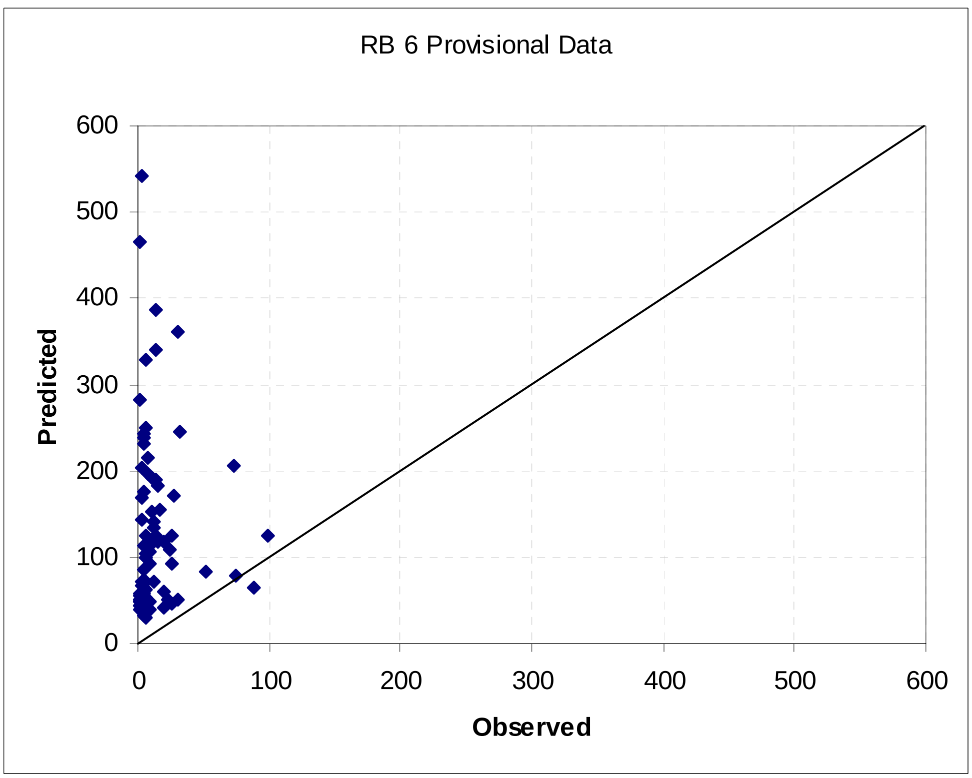

Figure 6. RWQCB 6 Provisional Data for Benthic Chlorophyll *a* Compared to Maximum Concentrations Predicted by Dodds et al. (2002) Regression

1.2.2 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.

Figure 7 and Figure 8 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 (1,000 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 9, while deviations relative to the mean prediction are shown in Figure 10. 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).

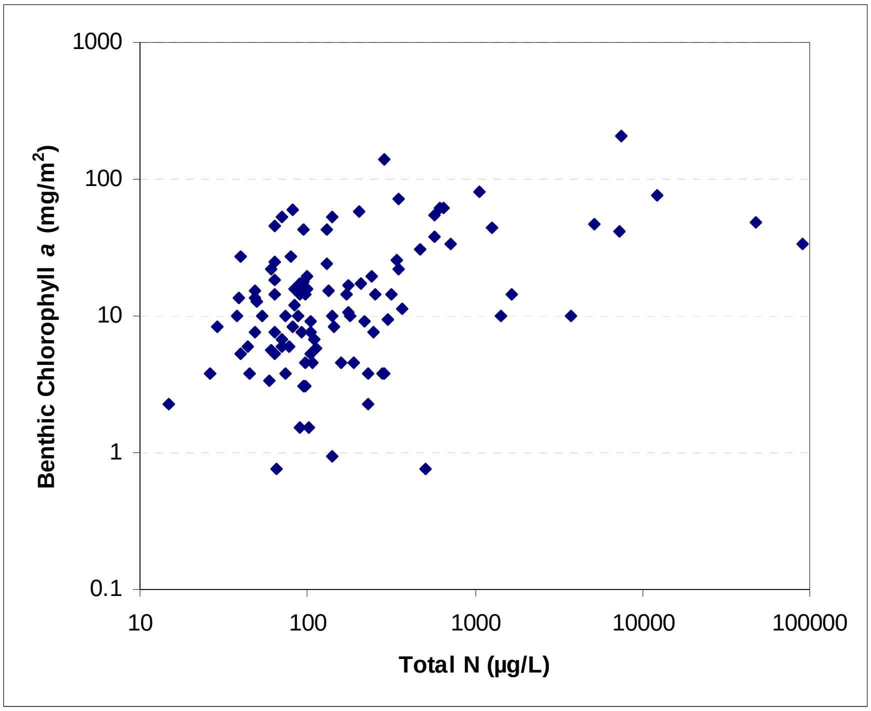

Figure 7. EMAP California Benthic Chlorophyll *a* Observations vs. Total Nitrogen

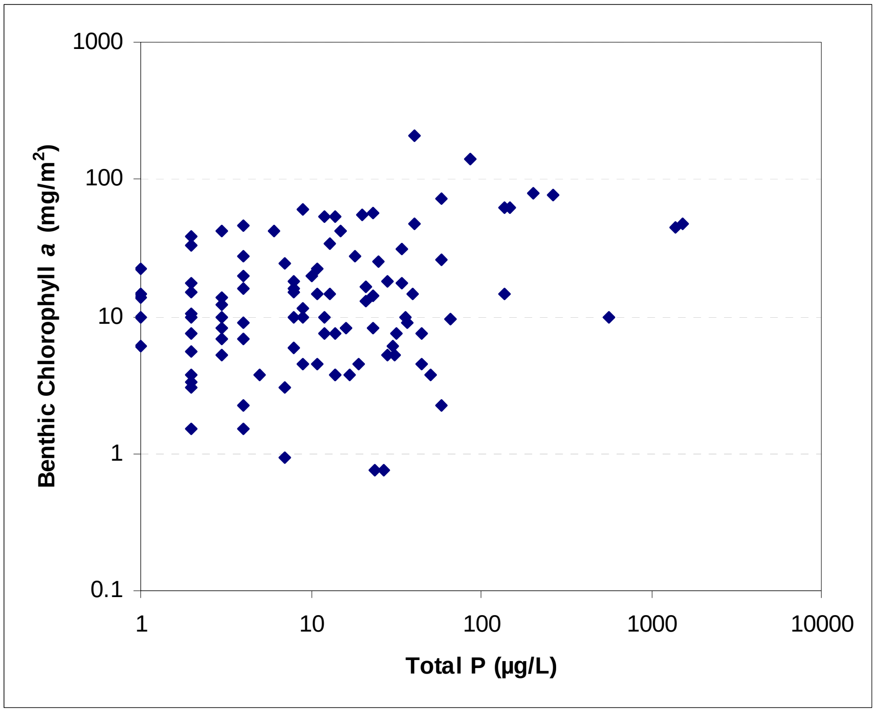

Figure 8. EMAP California Benthic Chlorophyll *a* Observations vs. Total Phosphorus

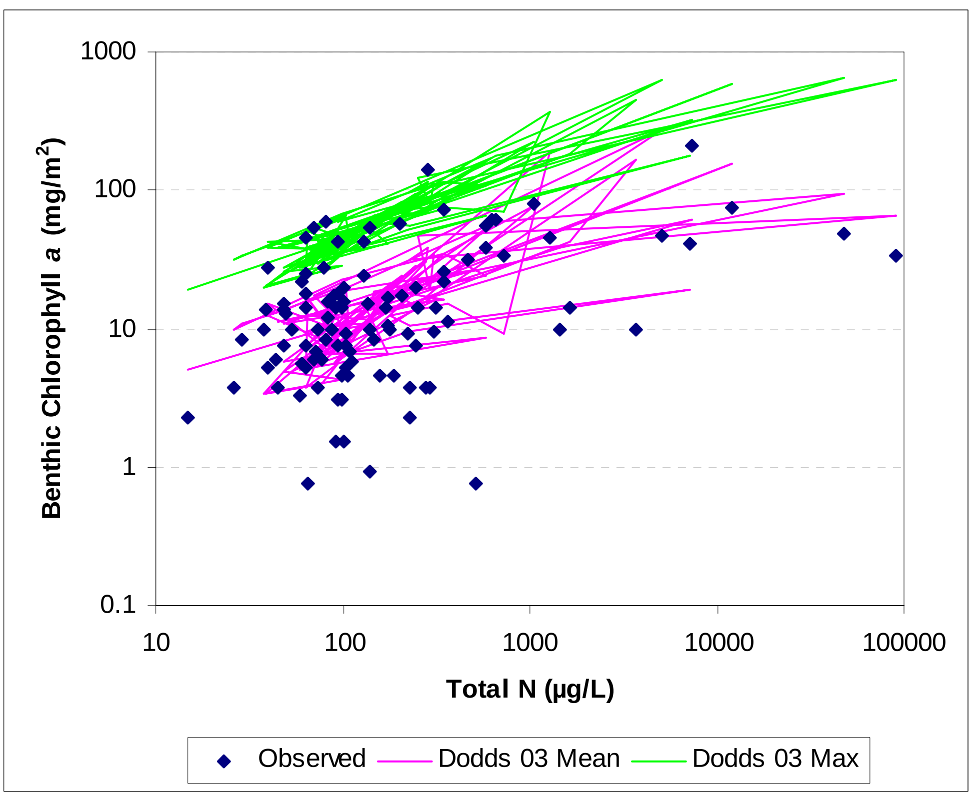

Figure 9. EMAP California Periphyton Chlorophyll *a* Compared to Various Regression Equations

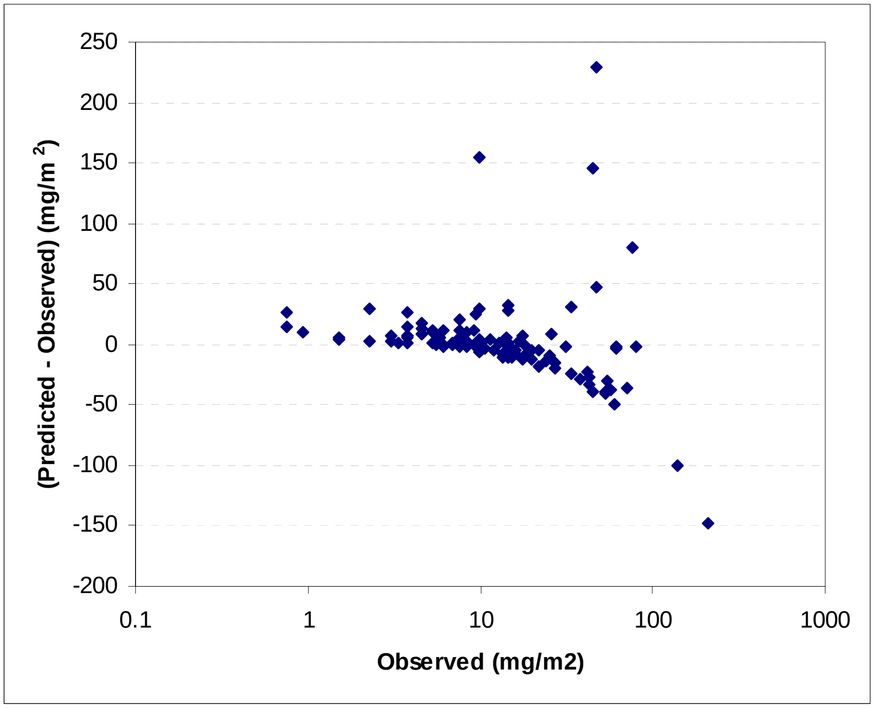

Figure 10. Deviations of EMAP California Benthic Chlorophyll *a* Data (mg/m^2) from Mean Predictions using Dodds et al. (2002)

1.3 Discussion

Comparison to the RWQCB 6 and EMAP data suggests that the equations proposed by Dodds et al. 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 40 to an R^2 greater than 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.

2 Simulation Modeling of Periphyton in Streams

Simulation modeling provides one line of evidence for the estimation of benthic algal or periphyton growth potential in streams. While a variety of models have been developed to simulate periphyton, the majority are too complex or too site-specific to be useful for initial scoping. 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 simple parametric representation can be adapted to provide initial estimates of benthic algal responses to availability of light and nutrients, and can be adjusted to achieve general agreement with the empirical relationships developed by Dodds et al. (1997, 2002). More sophisticated models may be needed to achieve accurate predictions of benthic algal density, but the simple method incorporated into QUAL2K shows promise as a scoping tool. The following sections lay out the details and proposed fine-tuning of this approach.

2.1 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).

2.1.1 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:

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

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

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

2.1.2 Respiration and Death

Benthic algae respiration is a first-order rate defined as

$$K_r = B \cdot k_{rb},$$

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

$$K_d = B \cdot k_{db},$$

where B is benthic algae (g/m^2) and k_{db} is the temperature-dependent bottom algae respiration rate (d^{-1}). The prediction of biomass uses only the sum of respiration and death, so model fitting exercises cannot distinguish these components independently.

2.1.3 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

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

2.1.4 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 below. Accounting for grazing within the “natural” death rate is problematic in general, as grazers may remove from 6 to 97 percent of algal biomass (Welch and Jacoby, 2004), depending on grazer density, types of algae and grazer(s), and so on. This problem is largely avoided when the intention is to predict the maximum concentration that would be present under minimal grazer pressure. For those conditions, 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. 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 2.

Table 2. 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} (inorganic nitrogen half-saturation constant)	0.015	mg-N/L
k_{sPb} (inorganic phosphorus half-saturation constant)	0.002	mg-P/L
K_{Lb} (light half-saturation constant)	50	cal/cm^2/d

2.1.5 Relationship to Biomass as Chlorophyll *a*

The QUAL2K model predicts as a state variable 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 biomass 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.

The 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). Numbers in this range are not representative of benthic algae. Typical stoichiometry of algae would be a carbon to AFDW ratio of about 0.45 and a carbon to chlorophyll *a* ratio typically in the range of 25 to 100 but possibly as high as 275 (Bowie et al., 1985). The typical range implies that the ratio of chlorophyll *a* density (mg/m^2) to ash free dry weight (g/m^2) for autotrophic benthic algae should be at least 4.5 and possibly as high as 18, while the highest reported value (for dinoflagellates) would lead to a ratio of 1.64. The low ratios at the EMAP sites suggest that the periphyton communities were likely dominated by heterotrophic fungi and bacteria whose growth is based on allochthonous carbon sources (e.g., terrestrial detritus or wastewater) rather than photosynthetic production. Heterotrophs usually dominate the shaded, fast-flowing, shallow first- to third-order streams in forests (Welch and Jacoby, 2004).

The QUAL2K approach directly predicts the accumulation of phototrophic algal biomass only. However, it is the total periphyton biomass that leads to nuisance conditions and the impairment of uses. Further complications arise because (1) some algae exhibit mixotrophy, in which they are able to assimilate energy from fixed carbon compounds as well as by photosynthesis, and (2) exudates of benthic phototrophic algae may support bacterial and fungal heterotrophic populations, thus tying the heterotroph density indirectly to photosynthetic production.

For comparison to the Dodds method, an appropriate ratio of chlorophyll *a* to AFDW must be selected. (The choice of a ratio does not, however, affect the quality of an optimized fit of the QUAL2K model to the Dodds results as both the ratio and $K_{p,max}$ are linear multipliers on the predicted chlorophyll *a* density; thus, an over-estimate of one value may be compensated for by a downward adjustment in the optimized value of the other parameter.) As noted above, a wide range of values are available, and the model can be fit with any reasonable value. Various authors (cited in Welch and Jacob, 2004) have noted that the autotrophic index (defined as the ratio of AFDW to chlorophyll *a*) is a useful indicator of the influence of organic wastewater on periphyton communities. Collins and Weber (1978) suggested that an autotrophic index of 400 defines the upper limit of clean water conditions, and use of this boundary has been verified by observations of Biggs (1989). Because the primary use of the model in connection with nutrient criteria is to predict periphyton and chlorophyll *a* density for "clean water" or supporting conditions, where appropriate nutrient limits can be determined, it is appropriate to use the autotrophic index value of 400 (which corresponds to a ratio of 2.5 mg/m^2 chlorophyll *a* to 1 g/m^2 AFDW) for comparison of QUAL2K to the empirical regression models of Dodds et al. (1997, 2002). Alternate, site-specific ratios may be appropriate in specific water bodies where appropriate information is available.

2.2 Maximum Benthic Algal Growth Potential – Default Parameters

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

2.2.1 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 0.5 m with an extinction coefficient of 1 m^{-1}, typical of a moderately 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 3 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 3. 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 (as N)	0.00	16.30	0.36
Nitrite (as N)	0.00	2.50	0.15
Nitrate (as N)	0.03	42.45	5.05
Total Kjeldahl Nitrogen	0.08	8.60	0.81
Phosphate (as P)	0.01	12.55	0.42
Total Phosphorus	0.10	4.42	0.29

Figure 11 and Figure 12 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 (Table 3).

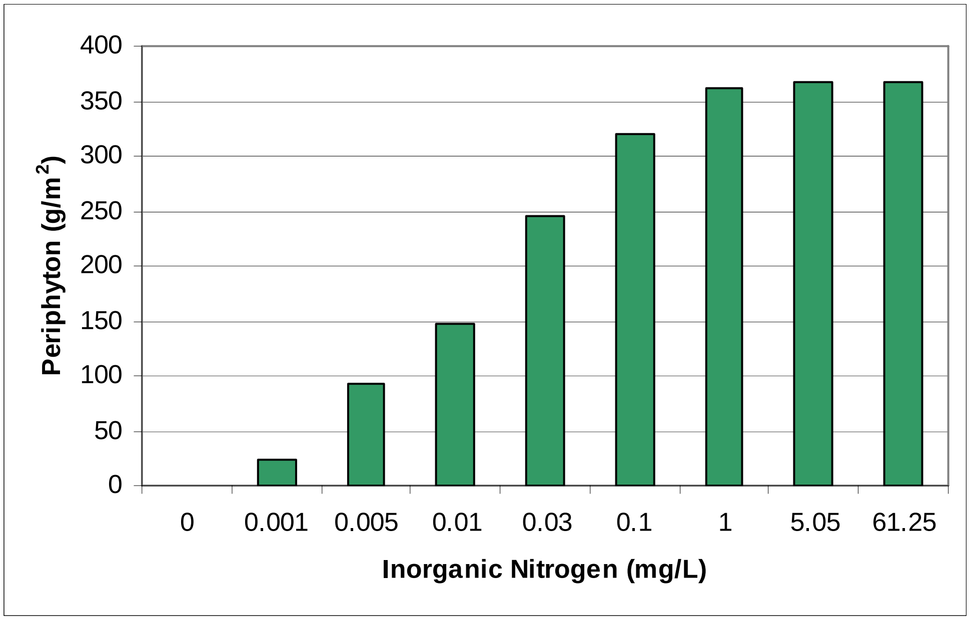

Figure 11. Predicted Steady-State Maximum Benthic Algal Biomass under Nitrogen Limitation with QUAL2K Default Parameters

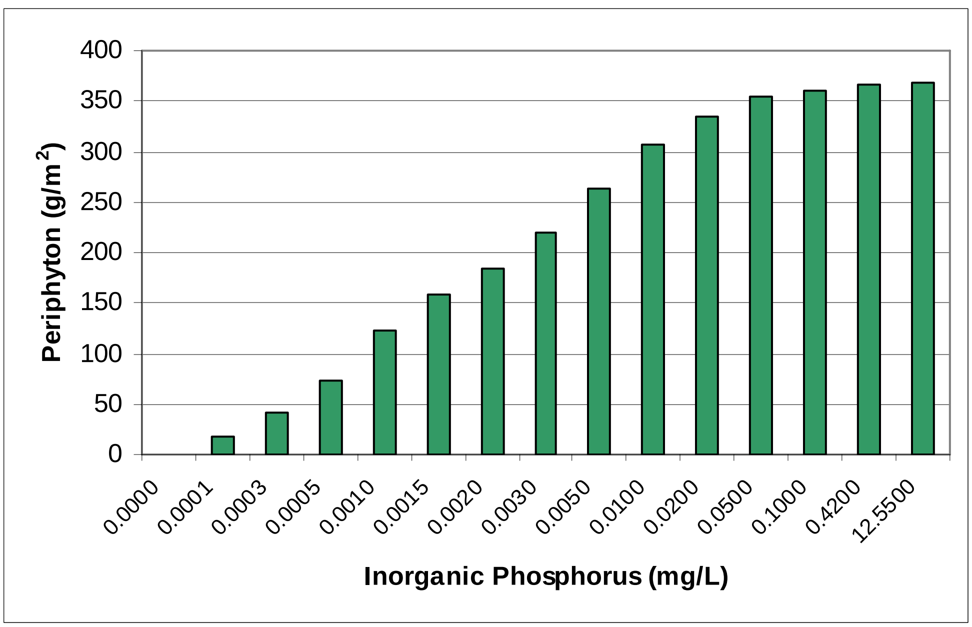

Figure 12. Predicted Steady-State Maximum Benthic Algal Biomass under Phosphorus Limitation with Default QUAL2K Parameters

The maximum predicted periphyton biomass is about 360 g/m^2 AFDW, which would be equivalent to about 900 mg/m^2 chlorophyll *a* using the ratio of 2.5. This is a biomass with no nutrient limitation, little light limitation (June clear sky insolation with relatively high water column transmission) and no losses to scour or heavy grazing.

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 13.

Two targets often cited for benthic chlorophyll *a* are 100 and 200 mg/m^2, which translate to AFDW biomass of 40 and 80 g/m^2 (using the 2.5 ratio). The results with initial default QUAL2K parameters shown in Figure 13 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 0.24 µg/L or inorganic nitrogen less than 1.8 µg/L, while to achieve the 200 mg/m^2 benthic chlorophyll *a* target it would be necessary to hold inorganic phosphorus below 0.56 µg/L or inorganic nitrogen below 4.2 µg/L. The default model thus yields extremely low target values. These values are not consistent with the targets recommended by Dodds et al. (1997), but are based on poorly constrained default parameter values that are not very meaningful for criteria in real streams. Nevertheless, the low soluble P concentrations (about 10 µg/L) that saturate periphytic biomass levels of 600 to 1,000 mg chlorophyll/m^2 in laboratory channels are consistent with the QUAL2K results (Welch and Jacoby, 2004).

As will be shown below, significantly higher criteria appear 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.

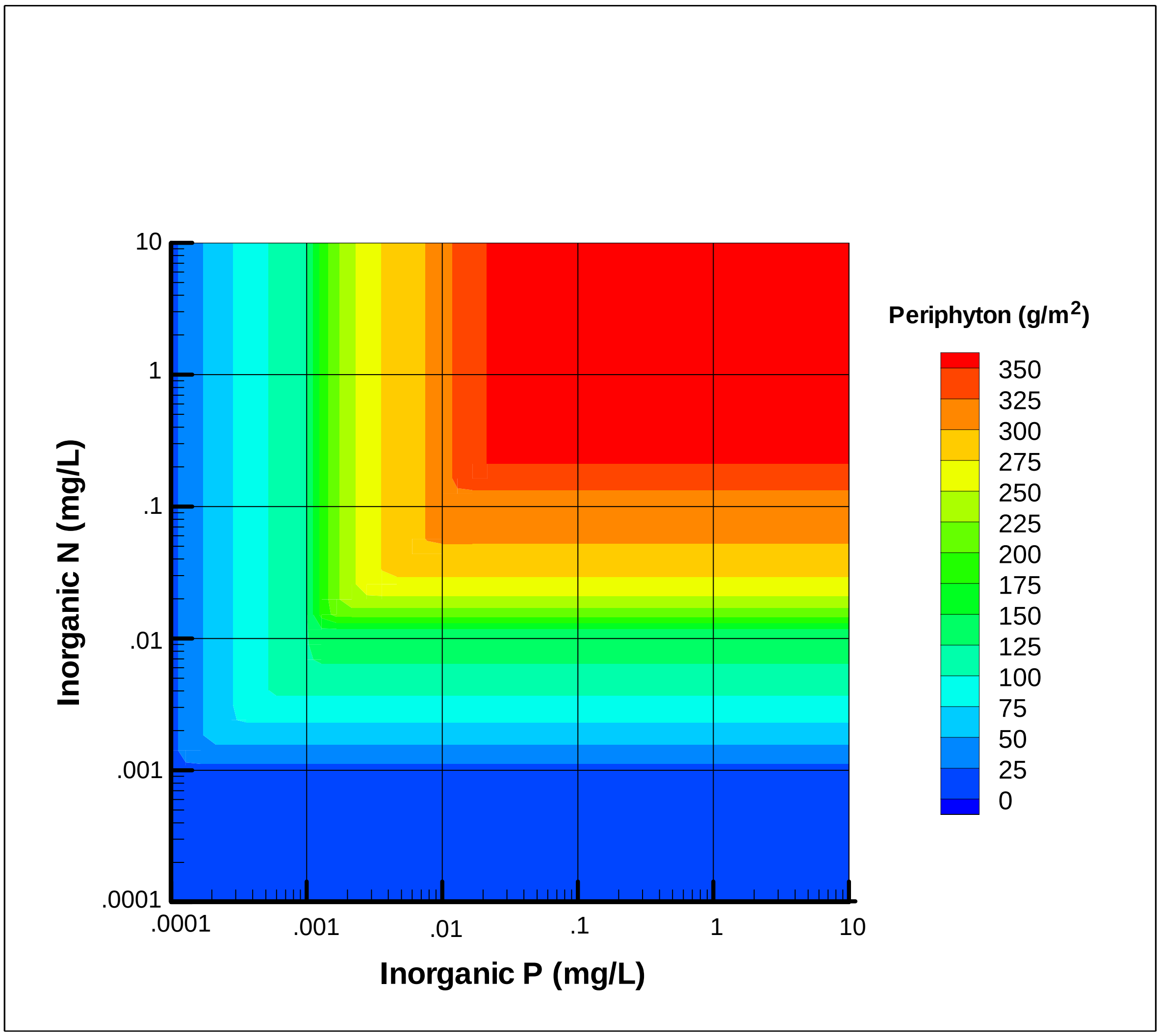

Figure 13. Response Surface for Maximum Periphyton Biomass (AFDW) versus Inorganic Nitrogen and Phosphorus Concentrations, QUAL2K Equations with Default Parameters

2.3 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 14). 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.

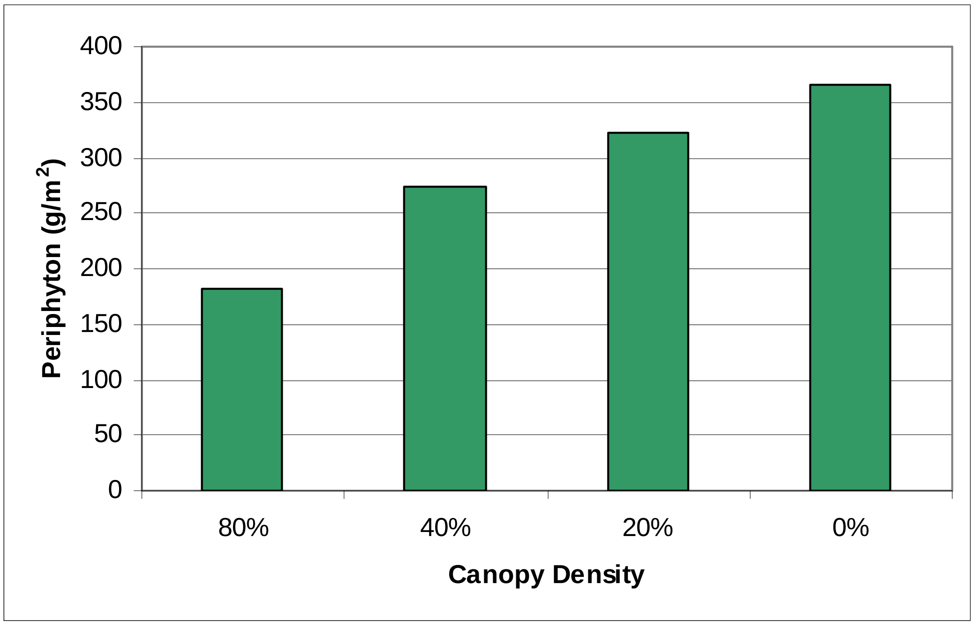

Figure 14. Steady State Benthic Algal Density Response to Canopy Closure without Nutrient Limitation (QUAL2K Default Parameters)

2.4 Reconciling QUAL2K and Dodds' Estimates – Direct Fit

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 Section 1.1, 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 QUAL2K is set up to work with soluble 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 15). 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.

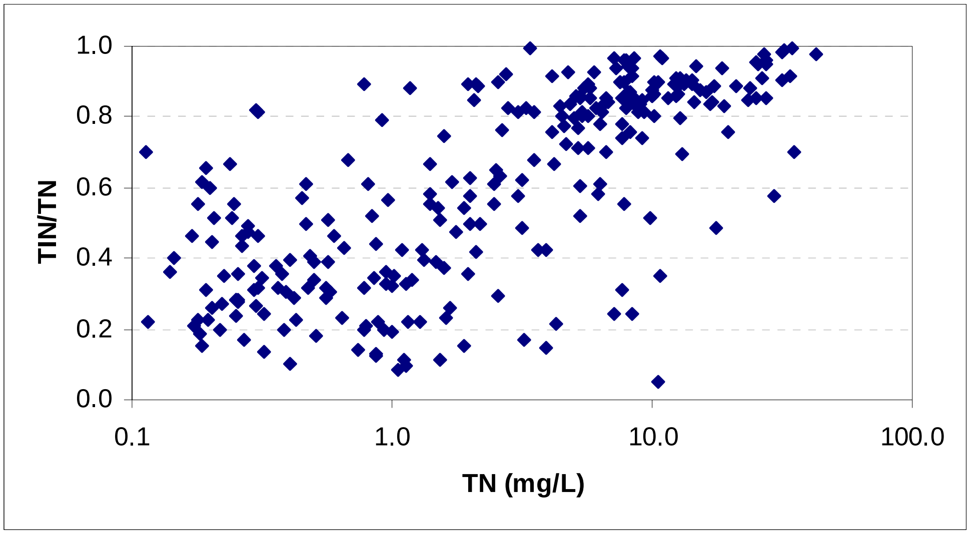

Figure 15. Inorganic Nitrogen Fraction vs. Total Nitrogen in Ecoregion 6 Streams

For phosphorus, there is no evident correlation between inorganic fraction and total phosphorus concentration (Figure 16), and observed values range from 0 to greater than 1 (not shown), reflecting dubious laboratory precision.

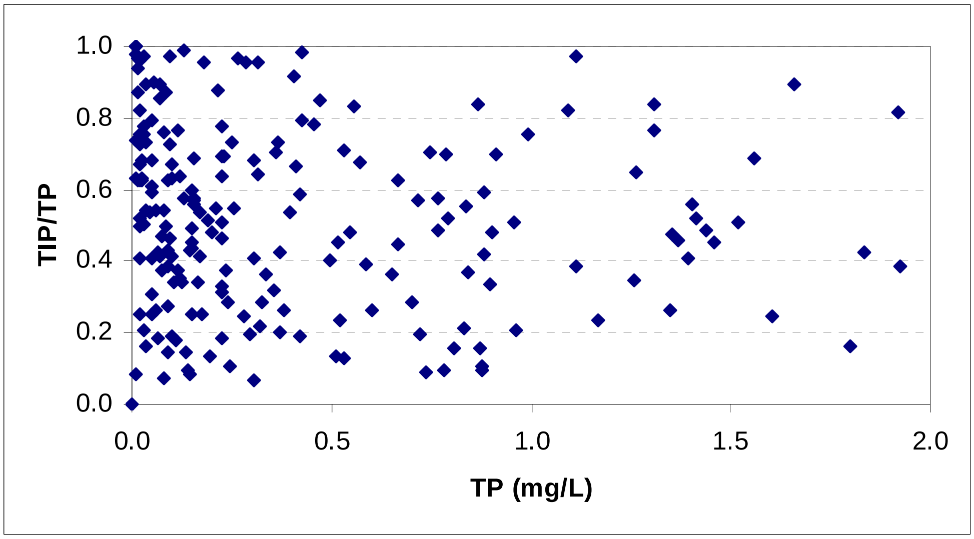

Figure 16. Inorganic Phosphorus Fraction vs. Total Phosphorus in Ecoregion 6 Streams

For the purpose of the initial 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 inorganic), 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 inorganic). 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 fractions.

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 fractions.

When these assumptions are applied, the QUAL2K predictions with default parameters deviate markedly from the Dodds et al. equation predictions. The QUAL2K predictions are considerably higher than those predicted by the Dodds model and not linearly related to those predictions.

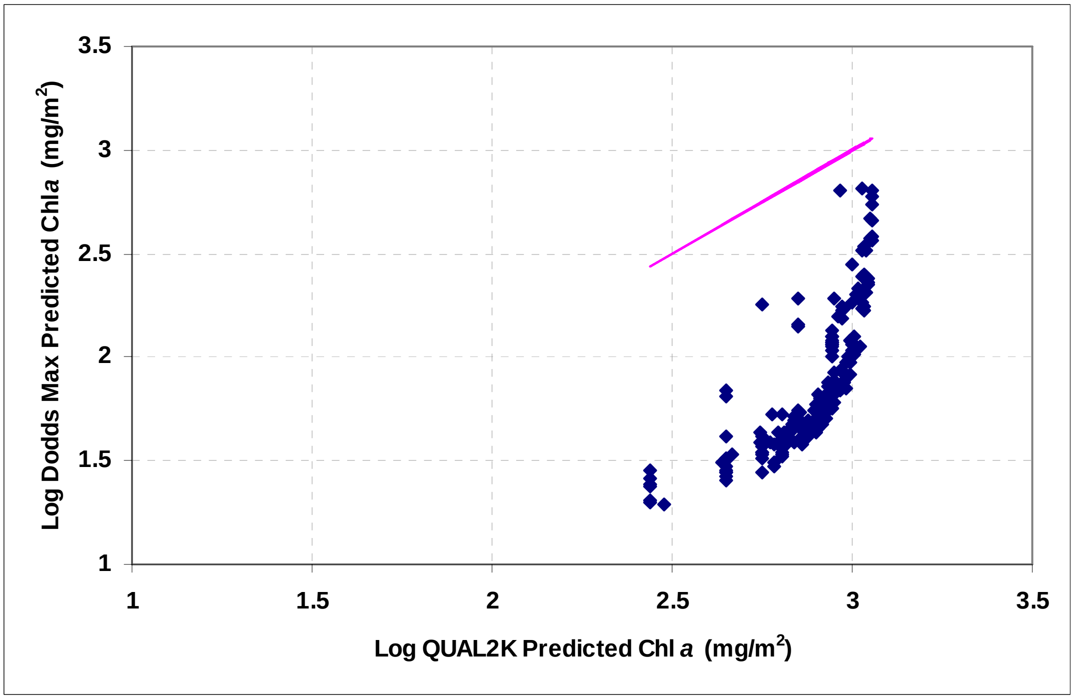

Figure 17. Comparison of QUAL2K with Default Parameters to Dodds (2002) Equation for Maximum Periphyton Chlorophyll *a*

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 resulting fit is shown in Figure 18 and the corresponding parameter values in Table 4.

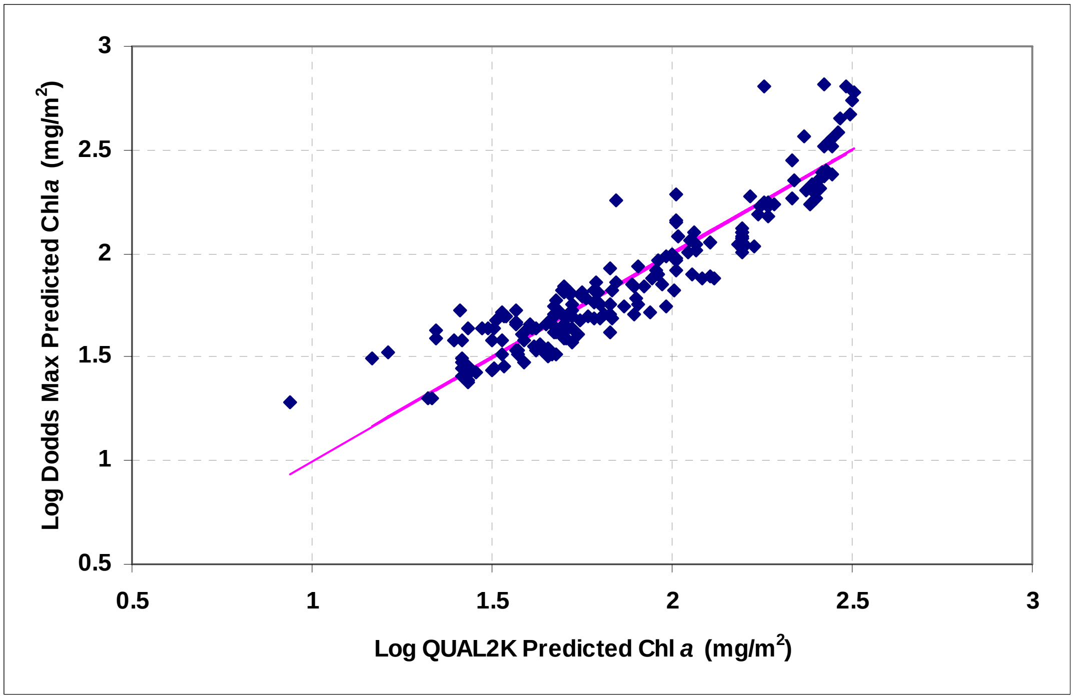

Figure 18. Optimized Reconciliation of QUAL2K Steady-State Predictions to Dodds (2002) Equation for Maximum Periphyton Chlorophyll *a*

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

Table 4. QUAL2K Kinetic Parameters for Benthic Algae Adjusted to Dodds' (2002) Results

Parameter	Default Value	Optimized Value	Units
$K_{p\,\max}$ (maximum photosynthetic rate)	60	60.06	g/m^2-d
k_{rb} (respiration rate)	0.125	0.125	1/d
k_{db} (natural death rate)	0.02	0.270	1/d
k_{sNb} (inorganic nitrogen half-saturation constant)	0.015	0.206	mg-N/L
k_{sPb} (inorganic phosphorus half-saturation constant)	0.002	0.00853	mg-P/L

Fitting the model requires an elevated death rate, which appears unrealistic. The estimated inorganic nutrient half-saturation constants are higher in the adjusted model than specified in the default parameters. This likely reflects the model assumption that the water column is completely mixed, whereas 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 until higher concentrations are reached.

The model fit yields a maximum density of about 140 g/m^2 AFDW (350 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 900 mg/m^2 chlorophyll *a*) because of the increase in the natural death rate relative to the default parameters, and lower than values observed in many streams. The QUAL2K and Dodds predictions diverge for higher concentrations, with the Dodds equation predicting much greater biomass for the same nutrient concentrations above about 250 g/m^2 AFDW. Thus, the direct fit of QUAL2K to the Dodds equation appears unsuitable for high nutrient concentrations.

Using the chlorophyll *a* to AFDW ratio of 2.5 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. Two sets of numbers are given for inorganic phosphorus. The first is obtained directly from the model, while the second represents a Redfield ratio interpretation of the total inorganic nitrogen target. The Dodds model has a high degree of uncertainty, and many of these streams appear to be nitrogen limited. Assuming that the fit to nitrogen is thus more reliable than the fit to phosphorus and following the arguments of Dodds et al. (1997), it may be more appropriate to select joint nitrogen and phosphorus targets consistent with the Redfield ratio. The resulting targets for both TN and TP are lower than those proposed in Dodds et al. (1997). However, the Dodds recommendations are based on the logarithm of the arithmetic mean concentration, while the QUAL2K fit more properly reflects the geometric mean or median concentration, which is typically lower than the mean for log-normally distributed environmental variables. The P targets are in line with the findings of Sosiak (2002) on the Bow River who associated a maximum periphytic chlorophyll *a* density of 150 mg/m^2 with a median soluble reactive phosphorus concentration of 7 µg/L, and with Biggs (2000), who found that 30 days of accrual at 10 µg/L soluble reactive phosphorus produced a biomass of 200 mg/m^2 chlorophyll *a* in a group of New Zealand streams. It should be cautioned that the nutrient targets derived in this way ignore the availability of nutrients from sediment and the role of luxury storage and nutrient recycling in the periphyton mat. They also depend on the assumption of chlorophyll *a* to AFDW ratio, and use of a lower ratio would yield higher targets. Finally, it 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. Nutrient Concentrations to Achieve Maximum Benthic Chlorophyll *a* Targets Estimated by Optimized QUAL2K Equations for Ecoregion 6

Maximum Chlorophyll *a* (mg/m^2)	Corresponding Algal Biomass (g/m^2)	Total Inorganic N (µg/L)	Total Inorganic P (µg/L)	Total Inorganic P (µg/L) – Redfield Ratio
100	40	82	3.4	11.3
150	60	154	6.4	21.2
200	80	273	11.3	37.7

2.5 Reconciling QUAL2K and Dodds' Estimates – Revised Model

While a general agreement can be obtained between Dodds' equation for maximum chlorophyll *a* and predictions using the QUAL2K equations by optimizing parameter values, the two solutions diverge at high concentrations, with the optimized QUAL2K underestimating the results of the Dodds solution. Somewhat less obvious, the linear fit for QUAL2K tends to overestimate the Dodds results in the mid range. The combined effect is to underestimate the nutrient concentrations required to meet maximum chlorophyll *a* targets less than 200 mg/m^2, while at the same time potentially underestimating the impacts that may occur in connection with higher nutrient concentrations.

These observations led to a reconsideration and revision of the approach used in QUAL2K. What causes the observed lack of fit in QUAL2K? The model represents benthic algal growth in a manner analogous to planktonic algal growth using a Monod formulation. This assumes that limitations on growth are directly determined by ambient water quality concentrations (C) of nutrients, characterized by a Michaelis-Menten half-saturation constant (K), such that the fraction of potential growth is given by $C/(C+K)$.

The Monod formulation rests on the assumption that measured ambient concentrations are representative of actual nutrient availability to growing cells, and also that K is a constant. Unlike planktonic algae, benthic algae are not fully exposed to the water column. The assumption may be reasonable at low periphyton densities, but becomes less appropriate at higher densities, resulting in a divergence from the Dodds solution. Assuming that the empirical relationship developed by Dodds is correct, the divergence could be explained by a combination of some of the following factors: (1) inappropriate assumptions regarding inorganic nutrient fractions, particularly at higher periphyton densities where the periphyton may deplete inorganic nutrients from the water column, (2) development of a concentration gradient in the water column under conditions of high periphyton nutrient demand and low mixing intensity such that ambient water column samples are not representative of the concentrations available at the stream bottom, (3) "luxury" nutrient storage, either directly by algae or by heterotrophs in the periphyton mat, and (4) diffusion limitations on nutrient penetration into the periphyton mat.

Difficulties in obtaining reliable measurements of the inorganic nutrient fraction available to periphyton suggest that it may be advantageous to base the model on total nutrient concentrations. Similarly, Dodds et al. (1997, 2002) found that empirical models based on TN and TP provided better fits than models based on inorganic N and P.

Shifting to a total nutrient basis alone does not solve the discrepancy between the methods. Indeed, the model optimization in the previous section was developed using constant inorganic fractions, and so is directly proportional to a fit to total nutrients. A much better agreement between the QUAL2K and Dodds results can be obtained if nutrient availability (as a fraction of total nutrient concentration) is assumed to vary as a function of periphyton density. Mathematically, this is equivalent to assuming that the Michaelis-Menten half-saturation constant (on total nutrient concentration) varies with density, with a factor equal to the inverse of the availability factor.

Somewhat counter-intuitively, a much improved match between the two predictions across the entire range of concentrations can be obtained if nutrient availability is assumed to decline with increasing concentration (or the half-saturation constant increases with increasing nutrient concentration). Here, nutrient concentration is a surrogate for the thickness of the periphyton mat. As mat thickness increases, higher rates of detrital sloughing may be anticipated, resulting in higher particulate nutrient concentrations in the water column. Decrease in availability with increasing concentration (such as might occur due to diffusion limitation) means that the QUAL2K model can be fit with a lower death rate, resulting in higher predicted benthic algal densities at high nutrient concentrations, while maintaining or improving fit at lower concentrations.

We investigated several adjustments to obtain better agreement between QUAL2K and Dodds, and eventually settled on a scaled logistic model to represent the fraction of nutrient concentration that is not available. The logistic model is an approximation of the cumulative normal distribution and exhibits a characteristic s-curve shape. Availability is expressed as the complement of the unavailable fraction, resulting in the following representation:

$$Availability\ Fraction = 1 - \frac{\gamma}{1 + \exp(\alpha - \beta \log_{10} C)},$$

where γ is a shift parameter that establishes a non-zero lower bound for availability and α and β are scaling parameters. This representation ensures that as C goes toward zero availability goes to 1, and as C becomes large availability goes to $1 - \gamma$. The typical shape of this representation is shown in Figure 19.

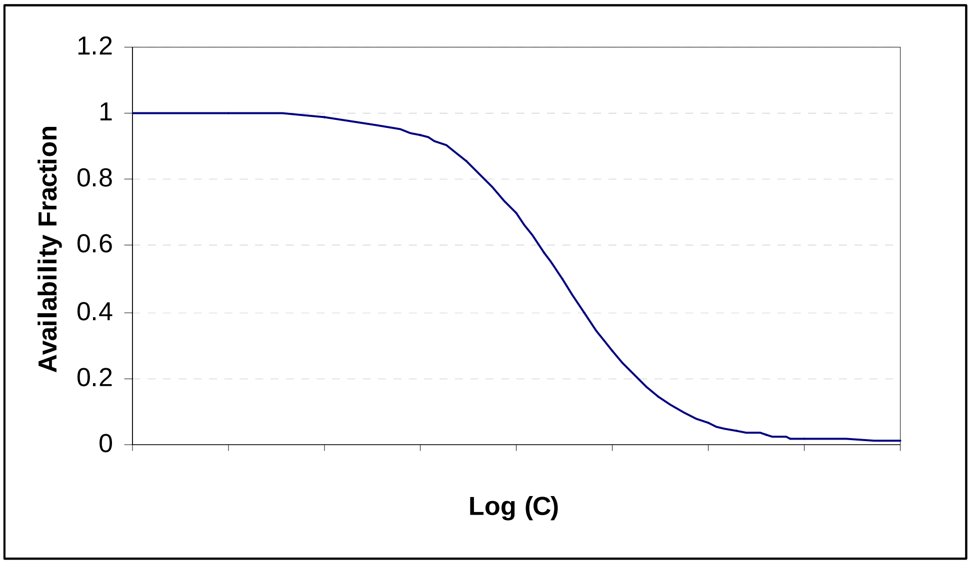

Figure 19. Logistic Representation of Total Nutrient Availability Fraction

Using this representation, an approximately unbiased agreement between the revised QUAL2K and Dodds estimates can be obtained across the entire range of data (Figure 20). The maximum potential biomass predicted by the revised model under the assumed light conditions (280 g/m^2 AFDW or 700 mg/m^2 chlorophyll *a*) is similar to that predicted by the QUAL2K model in Section 2.4); however, the relationship to the Dodds predictions is in much better agreement.

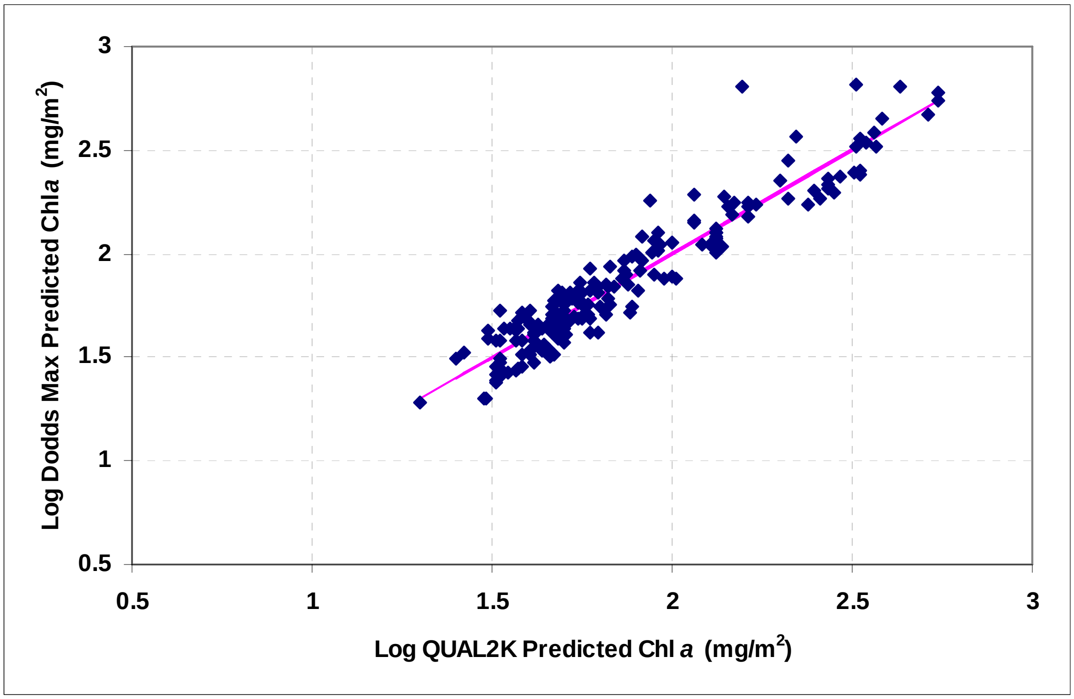

Figure 20. Optimized Reconciliation of Revised QUAL2K Steady-State Predictions to Dodds (2002) Equation for Maximum Periphyton Chlorophyll *a*

Optimized parameter values for the revised QUAL2K model are shown in Table 6. The maximum photosynthetic rate and natural death rate are similar to the previous model fit. Half-saturation constants are not directly comparable to the previous model – first because they are in terms of total nutrient concentration rather than the inorganic fraction, and second because they may be thought of as varying as a function of concentration (as the inverse of availability). Values are shown at zero and at the average concentration of each nutrient. The optimized logistic model parameters result in a wide variability in availability (or in effective half-saturation constant) for TN, with a much smaller range of variation for TP.

Table 6. Kinetic Parameters for Revised QUAL2K Model of Benthic Algae Adjusted to Dodds' (2002) Results

Parameter	Optimized Value		Units
$K_{p\max}$ (maximum photosynthetic rate)	54.95		g/m^2-d
k_{rb} (respiration rate)	0.125		1/d
k_{db} (natural death rate)	0.0563		1/d
k_{sNb} (total nitrogen half-saturation constant)	0.0260 at 0 mg/L TN 2.83 at 1.622 mg/L TN		mg-N/L
k_{sPb} (total phosphorus half-saturation constant)	0.0205 at 0 mg/L TP 0.0470 at 0.046 mg/L TP		mg-P/L
Logistic Availability Model	**Total N**	**Total P**	
α	-1.20	7.91	for concentrations in µg/L
β	1.60	10.36	
γ	0.993	0.564	

Concentrations predicted to achieve various target levels by control of one nutrient only are shown in Table 7. The phosphorus targets remain quite low. However, in this model, potential nitrogen co-limitation persists to a much higher concentration, due to the assumption of nitrogen availability varying with concentration, and the total phosphorus targets calculated from the Redfield ratio to total N are more consistent with the targets derived from Dodds et al. (2002) of 304 µg/L TN and 42 µg/L TP to achieve a maximum periphytic algal biomass of 100 mg/m^2 chlorophyll *a*, which also relied on the Redfield ratio. The directly calculated targets for total phosphorus are, however, in line with the findings of Sosiak (2002), studying the effects of removal of phosphorus and nitrogen from sewage effluent in the Bow River, Calgary, Alberta, who developed a regression equation predicting a maximum periphytic chlorophyll *a* density of 150 mg/m^2 in response to a median concentration of 18 µg/L total phosphorus.

Table 7. Nutrient Concentrations to Achieve Maximum Benthic Chlorophyll *a* Targets Estimated from Revised QUAL2K Model for Ecoregion 6

Maximum Chlorophyll *a* (mg/m^2)	Corresponding Algal Biomass (g/m^2)	Total N (µg/L)	Total P (µg/L)	Total P – Redfield Ratio (µg/L)
100	40	347	3.6	48
150	60	665	12.3	92
200	80	1065	18.7	147

2.6 EFFECTS OF GRAZING

As noted above, grazing likely accounts for much of the variation in observed relationships between nutrient concentrations and periphyton biomass (Walton et al., 1995; Anderson et al., 1999). Effects of grazing, however, depend on the density of grazers (which may vary dramatically over time) and the type of algae that are present. In some cases, grazers may accomplish nearly complete removal of diatoms, while there is generally little or no effect on *Cladophora*. Where the approach to nutrient criteria is based on maximum potential periphyton density, it may generally be appropriate to ignore the influence of grazers. It is possible to include the effect of grazing through the death rate of benthic algae in the QUAL2K model. However, assigning realistic rates will require site-specific information.

2.7 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. Unlike grazing, hydraulic scour can be considered as a relatively consistent feature of many streams that is less dependent on biology. Therefore, it may be appropriate to include scour effects as a co-factor in the analysis of likely maximum benthic algal density in streams.

Flow volume is a useful surrogate for velocity, and changes in flow volume correlate with changes in velocity. Sudden increase in velocity (e.g., by a factor of two to three) can result in the scour of a constant-velocity adapted biomass. Periphyton can adapt to growth in high velocity conditions (up to about 70 cm/s), but periphyton mats that develop under such conditions are still susceptible to scour when velocity increases (Horner et al., 1990). Therefore, the frequency of sudden increases in velocity (or flow) is a useful indicator of the role of scour impacts.

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_a), which he defined as the average time between flood events greater than 3 times the median flow. Note that days of accrual are calculated as [1/(mean frequency of events per year >3x median flow) x 365 d], and the measure is not the same as an actual count 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_a are similar for regressions on accrual only and on accrual and soluble reactive phosphorus. This has the form:

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

Figure 3 in Biggs (2000) shows that the response to d_a 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, computing the ratio of B at a specified value of d_a 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:

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

This results in a factor on the maximum potential periphytic algal biomass as a function of days of accrual as shown in Figure 21.

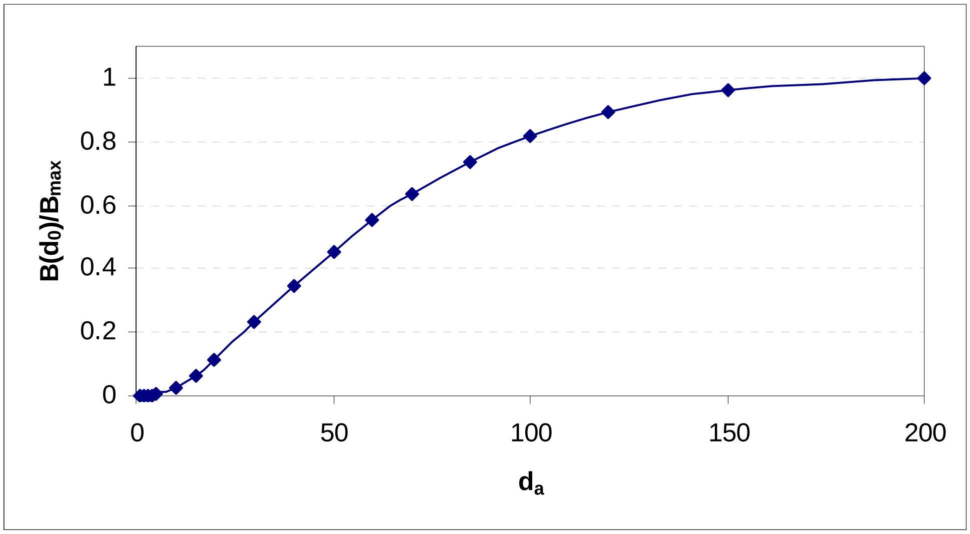

Figure 21. Fraction of Potential Maximum Periphyton Biomass Expected as a Function of Days of Accrual

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_a should be large for streams in this area, and the effects on biomass small. This is not, however, necessarily the case, as d_a 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_a 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 22). 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.

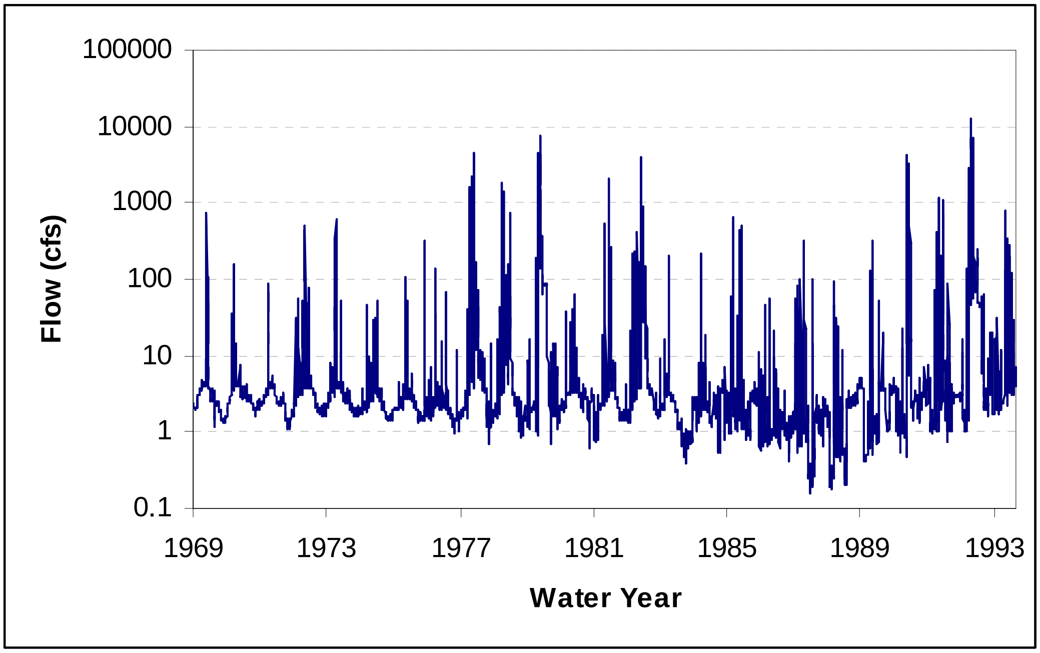

Figure 22. Daily Flows in the Santa Margarita River at Temecula, CA

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_a 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 exceeded 7.5 cfs. The value of d_a 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_a 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_a, 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_a 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.

2.8 DO Deficit Contribution of Benthic Algae

In addition to direct impacts, extensive growths of benthic algae are of concern due to their ability to cause significant diurnal fluctuations in dissolved oxygen concentration. To evaluate this component as an endpoint for analysis, the portion of the DO balance due to benthic algae can be separated from the overall DO balance.

The components of DO deficit are superposable (additive). Consider the portion due only to fixed (benthic) algae. Respiration and daily average photosynthesis by fixed algae can be treated like SOD and described by a net, distributed source-sink term:

$$S_d = P_a - R \ ,$$

where S_d is the net source-sink term (M/L^3-T), P_a is the daily average oxygen production rate due to photosynthesis, and R is the respiration rate.

The DO equation for these sources is described in Thomann and Mueller (1987, p. 320):

$$U\frac{dc_d}{dx} = K_a(c_s - c_d) + S_d \, ,$$

where U is the water velocity, x is the travel distance, K_a is the reaeration rate, c_s is the saturation concentration, and c_d is the DO concentration. The DO deficit equation (D_d) for benthic algae is then:

$$U\frac{dD_d}{dx} = -K_a D_d - S_d \, .$$

This is a first-order, ordinary-linear, non-homogeneous differential equation with solution

$$D_d = -\frac{S_d}{K_a}[1 - \exp(-K_a \frac{x}{U})] \, .$$

As $K_a \frac{x}{U}$ becomes larger, $D_d \Rightarrow -\frac{S_d}{K_a}$, which gives the maximum possible daily average deficit associated with benthic algae. The maximum is reached more quickly in streams that are shallow and rapidly moving, as opposed to deeper, slower moving streams.

Work of DiToro (1975a) and Erdmann (1979), as cited in Thomann and Mueller (1987, p. 290), shows that the diurnal range in deficit can be approximated by $D_d \pm \frac{\Delta c}{2}$, where

$$\Delta c \approx 0.5 P_a \text{ for } K_a(\text{day}^{-1}) < 2 \text{, and}$$

$$\Delta c \approx 0.3 P_a \text{ for } 2 \le K_a(\text{day}^{-1}) \le 10 \, .$$

For an approximate analysis in flowing streams, use the O'Connor and Dobbins (1958) method for reaeration:

$$K_a = \frac{12.9 U^{1/2}}{H^{2/3}} \, ,$$

where K_a is the reaeration rate (day^{-1}), U is the velocity in ft/s, and H is the depth in ft (note the non-SI units).

Outputs of the QUAL2K equations, bottom algae photosynthesis (K_p) and respiration (K_r) are in g DW/m^2-d (g DW = grams dry weight) as a daily average. From stoichiometry, photosynthesis (using NH_4) creates 32 mg O_2 per 12 mg C (creates more when NO_3 is used, so use NH_4 for conservative estimate). Respiration similarly uses 32 mg O_2 per 12 mg C. In addition, assume that DW = 40% C (QUAL2K default)

Given K_p and K_r from QUAL2K equations, along with water depth (H),

$$P_a(\frac{\text{mg - O}_2}{\text{L - d}}) = K_p(\frac{\text{g DW}}{\text{m}^2\text{ - d}}) \cdot \frac{32}{12} \cdot 1000 \frac{\text{mg O}_2}{\text{g C}} \cdot \frac{4\ g\,C}{100\ g\,DW} \cdot \frac{1}{H\text{ (m)}} \cdot \frac{1\,\text{m}^3}{1000\,\text{L}} \, ,$$

$$P_a = K_p \cdot 1.067 \cdot \frac{1}{H}.$$

Similarly, direct respiration is estimated as $K_r \cdot 1.067 \cdot \frac{1}{H}$. Assume that algal death also represents a direct oxygen demand and can be included with respiration, then

$$R = (K_r + K_d) \cdot 1.067 \cdot \frac{1}{H}.$$

Combining, the daily average DO deficit impact of benthic algae is estimated as

$$S_d = K_p \cdot 1.067 \cdot \frac{1}{H} - (K_r + K_d) \cdot 1.067 \cdot \frac{1}{H} = \frac{K_p - K_r - K_d}{H} \cdot 1.067\ ,$$

$$D_d = {}^{-S_d}\!/_{K_a}\ ,$$

At steady-state conditions (as in the benthic biomass calculation), (K_p-K_r-K_d) is, by definition, equal to zero. Deleterious impacts of benthic algae on the DO balance occur, however, under unsteady conditions of a biomass die-off, when production is less than respiration plus death. In the benthic biomass predictor, a reasonable estimate of relative risk can be obtained by evaluating K_r and K_d at the expected maximum benthic algal concentration associated with ambient conditions, while evaluating K_p at the user-specified minima of daily average insolation and nutrient concentrations to establish the maximum expected daily average deficit.

The diurnal range in DO deficit is $\pm \frac{\Delta c}{2}$ as given above, and is added to the daily average deficit to estimate the maximum instantaneous DO deficit. This approach is incorporated into the benthic biomass predictor spreadsheet to provide scoping analyses of the maximum DO deficit, or the range of DO fluctuation, that is expected from nutrient-induced growth of benthic algae.

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3 Using the Benthic Biomass Spreadsheet Tool

This memo provides background and instructions for using the California Benthic Biomass Tool. The tool is a Microsoft Excel spreadsheet, and is intended to be a simple but effective tool for predicting in-stream benthic algal density and other metrics in response to a number of inputs. The tool calculates algal density as ash free dry weight (g/m^2).and benthic chlorophyll *a* (mg/m^2). Both are estimated using a variety of methods as described in Sections 0 and 2 of this appendix:

- Dodds 1997 method (both mean and maximum)
- Dodds 2002 method (both mean and maximum)
- Standard QUAL2K Model method (maximum)
- Revised QUAL2K Model method (maximum)
- Revised QUAL2K, with adjustment for days of biomass accrual (maximum).

The maximum algal contribution to dissolved oxygen deficit is also calculated, using the Revised QUAL2K Model method. Lastly, the tool allows the user to supply a target (either algal density or benthic chlorophyll *a*), select a calculation method, and the tool will display a graph of allowable TN and TP to meet the target.

To run the Benthic Biomass Predictor spreadsheet, you must have macros enabled in Excel. If you have difficultly using the spreadsheet (e.g., the drop-down box for target selection is disabled), please refer to the last page of this memo for instructions about enabling macros in Excel.

The tool has two sheets for user input and viewing results. The majority of user input fields and model results are located on the **Main** sheet. The **Sensitivity** sheet shows how algal density varies when nitrogen, phosphorus, or unshaded solar radiation are varied. The tool also has a **Params** sheet, where the user can adjust less frequently used model parameters, and a **Calcs** and a **Target** sheet, which are used for internal model calculations and have no input fields or final output information. The **Main** sheet and the **Sensitivity** sheet are configured to automatically print on one and two pages each, respectively.

The **Main** sheet has a **USER INPUTS** section with the following inputs:

USER INPUTS

Nutrient Concentrations (mg/L)

	Average	Minimum	Maximum
Ammonia-N	0.03	0.02	0.05
Nitrite-N	0.001	0.001	0.001
Nitrate-N	0.14	0.05	0.2
Organic N	0.32	0.1	1
Total N (calc)	0.491	0.171	1.251
Inorganic P	0.0062	0.003	0.01
Organic P	0.0036	0.002	0.01
Total P (calc)	0.0098	0.005	0.02

Unshaded Solar Radiation (cal/cm2/d)	Average	Minimum	Maximum
	658	400	700

Stream Inputs

Stream Depth (m)	1
Stream Velocity (m/s)	0.3
Water Temperature (^oC)	20.0
Days of Accrual (optional)	80
Canopy Closure	○ 0% ◉ 20% ○ 40% ○ 80%

Method & Target Selection

Select Method:	Revised QUAL2K, max algal density
Target Max Algal Density (g/m^2 AFDW)	60
Corresponding Benthic Chl a (mg/m^2)	150

California Benthic Biomass Tool, v11 (3-21-06)

The Target Selection allows for 14 variations of methods, which are available from the drop-down box:

- Standard QUAL2K, max algal density
- Standard QUAL2K, benthic chl *a*
- Revised QUAL2K, max algal density
- Revised QUAL2K, benthic chl *a*
- Revised QUAL2K+accrual adj, max algal density
- Revised QUAL2K+accrual adj, benthic chl *a*
- Dodds '97, mean algal density
- Dodds '97, mean benthic chl *a*
- Dodds '97, max algal density
- Dodds '97, max benthic chl *a*
- Dodds '02, mean algal density
- Dodds '02, mean benthic chl *a*
- Dodds '02, max algal density
- Dodds '02, max benthic chl *a*

Each method allows for the target itself to be specified either as algal density or benthic chlorophyll *a*. The units displayed next to the target entry cell reflect the type of target. The corresponding target, calculated from the ratio of chlorophyll *a* density (mg/m^2) to ash-free dry weight (AFDW, g/m^2) is shown below the specified target. The previous figure shows an algal density target; if a benthic chlorophyll *a* target is selected, the description and units are updated to reflect the selected method.

Method & Target Selection

Select Method:	Standard QUAL2K, benthic chl a
Target Benthic Chl a (mg/m^2)	**100**
Corresponding Algal Density (g/m^2 AFDW)	**40**

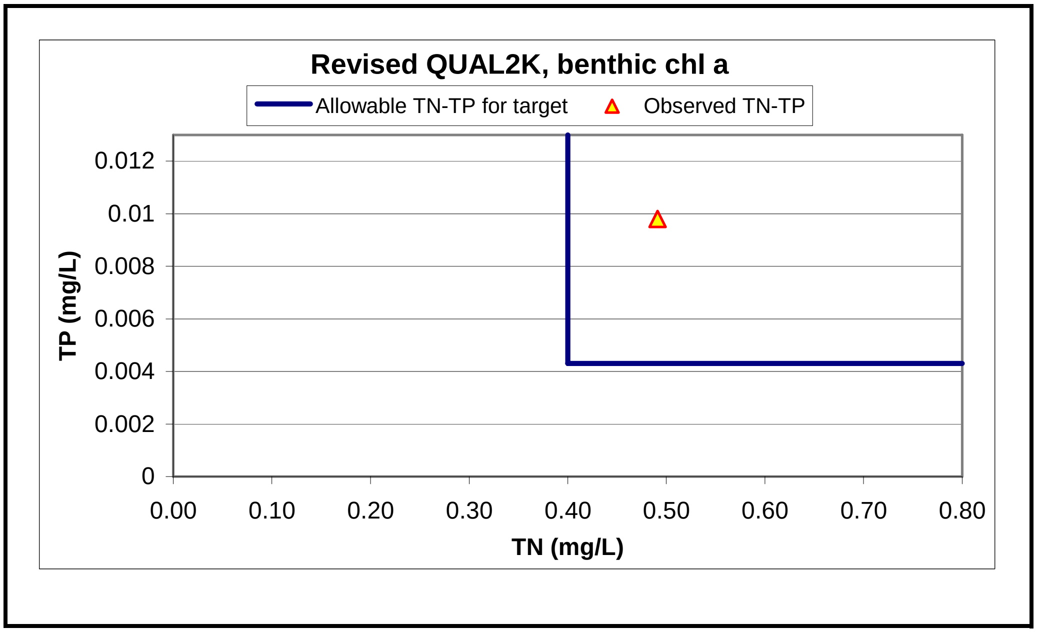
The RESULTS are divided into two sections. The first section is a table showing estimated algal density and benthic chlorophyll *a* under several variations of the methods. The maximum algal contribution to dissolved oxygen deficit (calculated using the Revised QUAL2K method) is also shown:

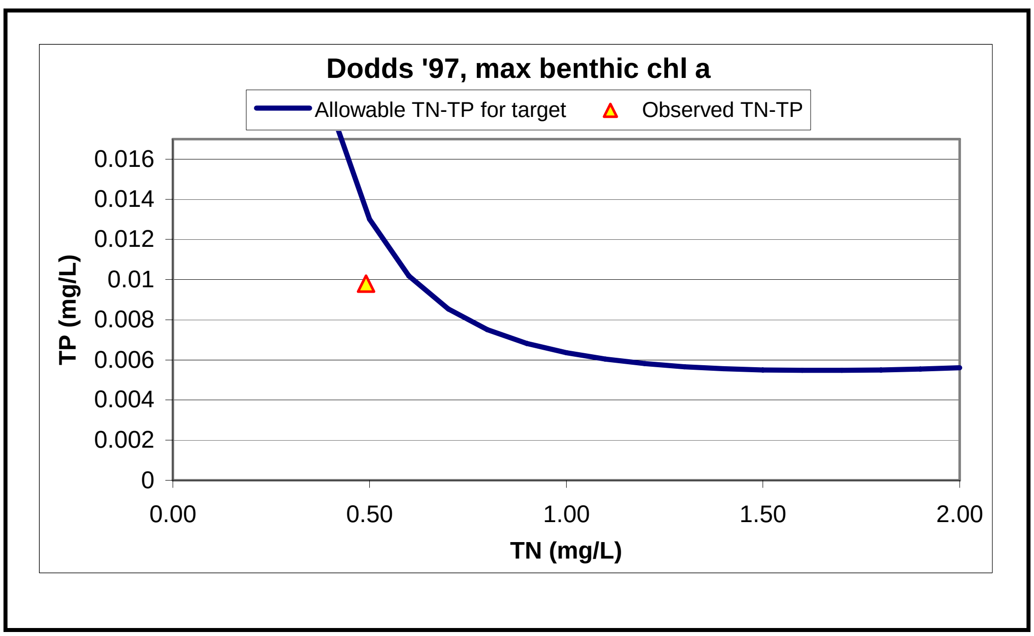
RESULTS

Method	Max algal density, ave conditions (g/m^2 AFDW)	Benthic chlorophyll a estimate (mg/m^2)
Standard QUAL2K	**54**	**135**
Revised QUAL2K	**45**	**113**
Revised QUAL2K with accrual adj	**32**	**80**
Dodds '97, mean Chl a	**13**	**32**
Dodds '97, max Chl a	**36**	**91**
Dodds '02, mean Chl a	**7**	**17**
Dodds '02, max Chl a	**35**	**86**

Max algal contribution to DO deficit (mg/L)	**1.42**

The second section shows the graph of maximum allowable TN and TP according to the selected numeric target and estimation method. The blue line shows the threshold above which the combination of TN and TP is estimated to result in a violation of the target. The current TN and TP from the USER INPUTS section are shown on the graph for reference. Two example graphs are shown here. The first graph was calculated using a QUAL2K method; note that QUAL2K calculates fixed thresholds, reflected by the linear relationships shown in the graph. The second graph was calculated using a Dodds method, which provides a non-linear relationship.



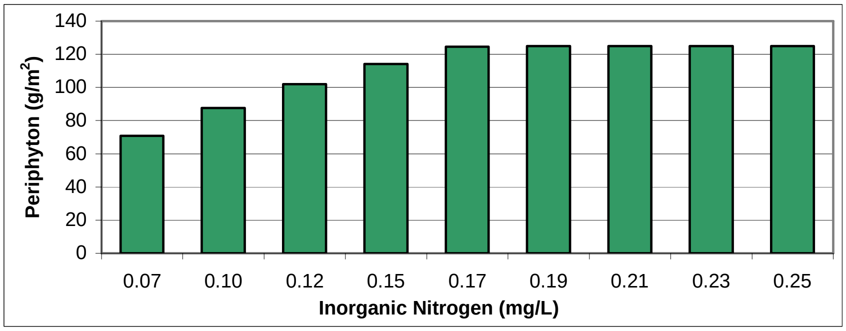


The Sensitivity sheet shows the effect of varying inputs across the chosen minimum and maximum ranges, for both the Standard and Revised QUAL2K Methods:

SENSITIVITY ANALYSIS, STANDARD QUAL2K METHOD

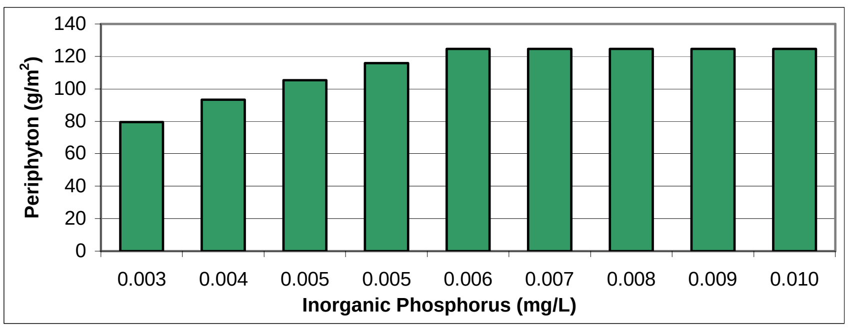
Vary inorganic nitrogen, keep inorganic phosphorus and solar radiation at mean

Inorganic N (mg/L)	Algae (g/m^2)
0.07	70.9
0.10	87.8
0.12	102.0
0.15	114.2
0.17	124.7
0.19	125.0
0.21	125.0
0.23	125.0
0.25	125.0



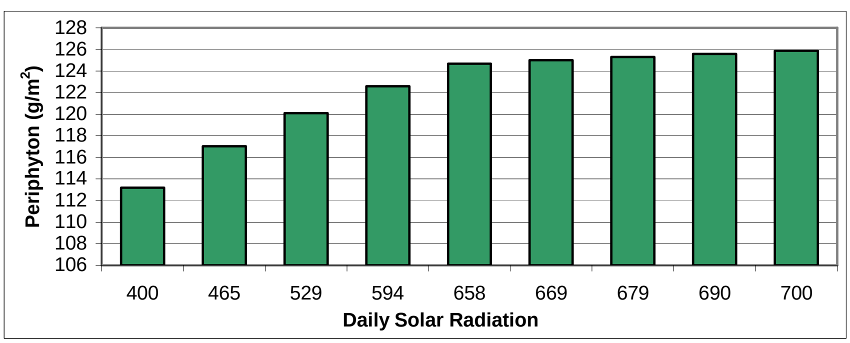
Vary inorganic phosphorus, keep inorganic nitrogen and solar radiation at mean

Inorganic P (mg/L)	Algae (g/m^2)
0.003	79.5
0.004	93.4
0.005	105.4
0.005	115.8
0.006	124.7
0.007	124.7
0.008	124.7
0.009	124.7
0.010	124.7



Vary Solar Radiation, keep nitrogen and phosphorus at mean

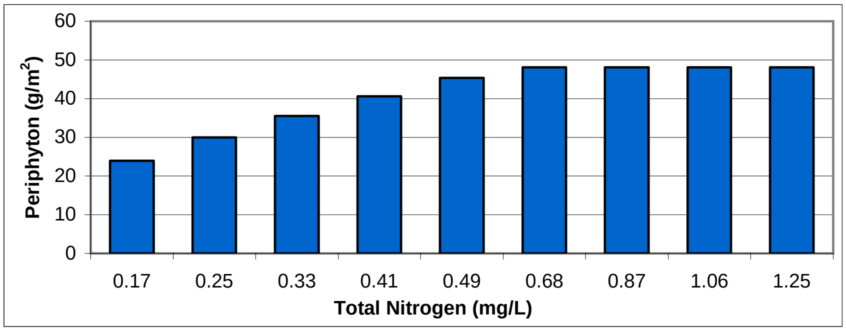
Unshaded Solar Radiation	Algae (g/m^2)
400	113.2
465	117.0
529	120.1
594	122.6
658	124.7
669	125.0
679	125.3
690	125.6
700	125.9



SENSITIVITY ANALYSIS, REVISED QUAL2K METHOD

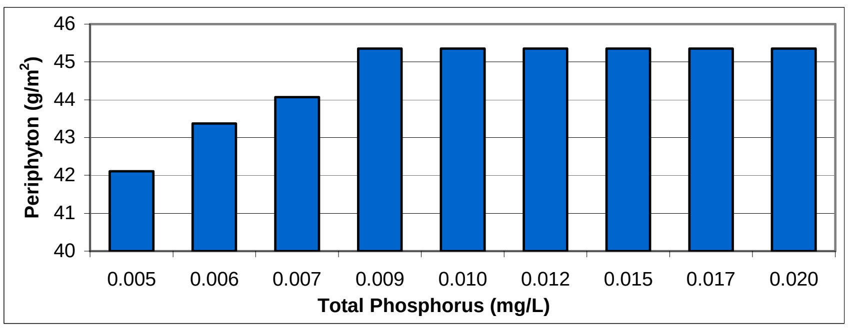
Vary total nitrogen, keep total phosphorus and solar radiation at mean

TN (mg/L)	Algae (g/m^2)
0.17	23.9
0.25	30.0
0.33	35.5
0.41	40.6
0.49	45.3
0.68	48.1
0.87	48.1
1.06	48.1
1.25	48.1



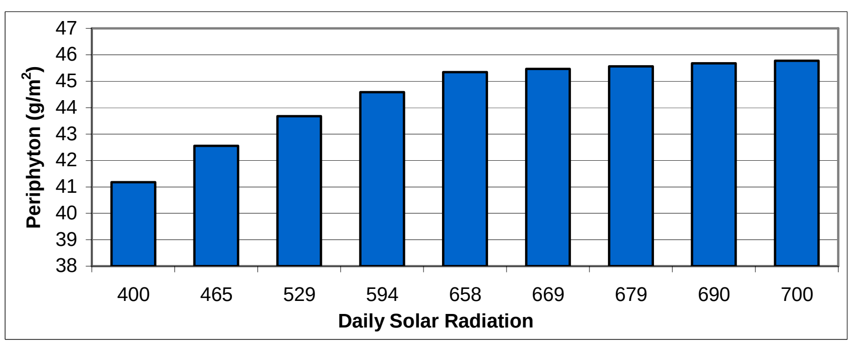
Vary total phosphorus, keep total nitrogen and solar radiation at mean

TP (mg/L)	Algae (g/m^2)
0.005	42.1
0.006	43.4
0.007	44.1
0.009	45.3
0.010	45.3
0.012	45.3
0.015	45.3
0.017	45.3
0.020	45.3



Vary Solar Radiation, keep total nitrogen and total phosphorus at mean

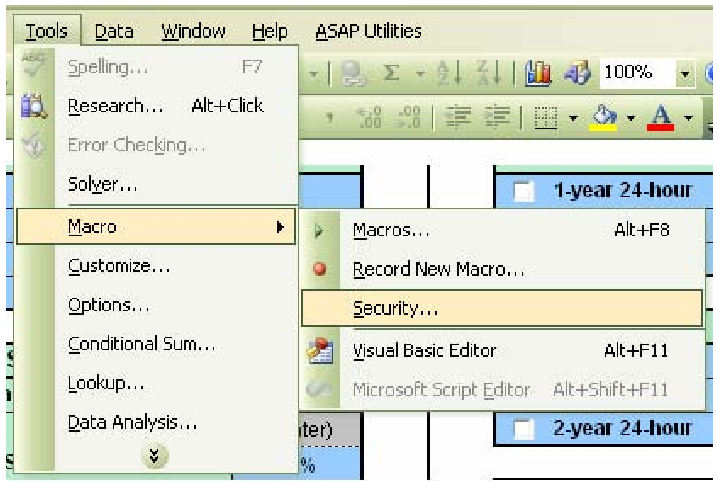
Unshaded Solar Radiation	Algae (g/m^2)
400	41.2
465	42.6
529	43.7
594	44.6
658	45.3
669	45.5
679	45.6
690	45.7
700	45.8



Enabling Macros in Excel

This tool makes extensive use of Microsoft Visual Basic for Applications (VBA) script, both for navigation and for more complicated internal calculations. The use of these “macros” is essential for operating this tool. However, one of these settings (frequently enabled by default in Excel) prevents all macros from running, and also may not warn the user that macros are disabled.

To enable this tool to use its VBA Script, please make the following changes to your Excel security settings. On the menu, select **Tools**, choose **Macro**, and then choose **Security…** In the window that opens, select the **Security Level** tab.



Select the button next to **Medium**, and click the **OK** button. Now when you open the tool, you should be given a choice to enable macros. You must enable macros for the tool to operate properly.

The **Security Level** will remain set to **Medium** even after you close the spreadsheet – this setting applies to Excel as a whole, not just the tool. You may change the **Security Level** to a higher setting after you have finished, but you would need to reset it to **Medium** whenever you use the tool.

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APPENDIX 4
BATHTUB SPREADSHEET TOOL USER GUIDE AND DOCUMENTATION

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Appendix 4 BATHTUB Spreadsheet Tool User Guide and Documentation

This appendix provides background and instructions for using the California BATHTUB Lake Model Tool. The tool is a Microsoft Excel spreadsheet, and is intended to be a simple but effective tool for predicting growing season chlorophyll *a* lake response to a number of inputs. The tool also allows the user to specify a chlorophyll *a* target, and predicts the probability that current conditions will exceed the target, as well as showing allowable N and P loading combinations necessary to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in Secchi depth.

The lake response model is based on the Army Corps of Engineers BATHTUB model.

The following sections are included:

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Table of Contents

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1 BATHTUB Model Background

The objective of the BATHTUB model spreadsheet tool application is to establish screening level nutrient loading targets for lakes and reservoirs by estimating algal response to nutrients while accounting for hydraulic residence time, light availability, and other key variables.

The U.S. Army Corps of Engineers' BATHTUB model (Walker, 1987, 1996) is used to analyze the water quality response in lakes and reservoirs to different nutrient loading scenarios. BATHTUB is designed to facilitate application of empirical eutrophication models to reservoirs and was modified for use in a spreadsheet application. The program performs water and nutrient balance calculations in a steady-state, spatially-segmented hydraulic network that accounts for advective transport, diffusion, and nutrient sedimentation. Eutrophication-related water quality conditions are expressed in terms of total phosphorus, total nitrogen, chlorophyll *a*, transparency, inorganic nitrogen, ortho-phosphorus, and hypolimnetic oxygen depletion rate. These conditions are predicted using semi-empirical relationships developed and tested on a wide range of reservoirs.

Mass balances are computed in BATHTUB at steady state over an appropriate averaging period. Steady-state approximation means that only seasonal or annual average loads and conditions are simulated, although the loads and conditions may change from year to year. In other words, the model does not represent day-to-day changes in flow, loads, or nutrient concentrations. Although this approach represents a compromise, it has proven effective in practice: short-term variations in lake conditions reflect variations in flow, including wind and weather effects, which require complex and labor-intensive models; such effects tend to average out, however, over longer time frames. Thus, annual or seasonal average conditions can be successfully predicted using data that are insufficient for simulating day-to-day variability.

BATHTUB provides a variety of options for simulating nutrient sedimentation, including several first- and second-order representations proposed in the literature, as well as methods developed explicitly for BATHTUB. Also available are five candidate sub-models for chlorophyll *a*, which depend variously on nitrogen, phosphorus, light and flushing rate limitations, and three candidate models relating Secchi depth (transparency) to chlorophyll *a*, turbidity, and nutrient concentrations. BATHTUB thus provides a highly flexible tool for developing a semi-empirical, annual-average analysis of nutrient concentrations and eutrophication. The model also includes extensive diagnostics and capabilities for error analysis.

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, the multiple segment option is not implemented in the spreadsheet tool, which is most appropriate for smaller lakes without highly complex morphometry.

Once the BATHTUB application is set up, lake responses to variations in other parameters can then be analyzed in a sensitivity analysis. The 2003-2004 Progress Report (Tetra Tech, 2004) used BATHTUB 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 can then be defined based upon whether the receiving waters exceed target criteria for planktonic chlorophyll *a* density.

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2 Calculation of Chlorophyll *a* Exceedance Probabilities

Selection of a target summer mean chlorophyll *a* concentration also has implications for the frequency of severe bloom conditions (defined as concentrations greater than 30 µg/L). In work on USACE reservoirs, Walker (1985, 1987) determined that the distribution of chlorophyll *a* concentrations in an impoundment could generally be described as lognormal. An estimate of the frequency of time that concentrations are greater than 30 µg/L can then be made from the arithmetic mean target concentration and a coefficient of variation on the log-transformed values (CV; standard deviation divided by the mean), using the algorithm found in Walker (1985).

Results of the analysis depend on the selection of an appropriate CV value. Walker (1987) states that the temporal CV for chlorophyll *a* concentrations in ACOE reservoirs was 0.62; however, the accompanying computer program defaults to 0.26. In an apparent later reanalysis, Figure 7.6 in Welch and Jacob, (2004) appears to have been calculated with a CV of 0.17, citing personal communication from Walker "for calibration to Corps of Engineers reservoirs." Temporal CVs will likely differ for other datasets.

0**}** shows the frequency of severe bloom conditions (concentrations greater than 30 µg/L) for different summer mean chlorophyll *a* targets and various assumptions regarding CV. Based on this analysis, setting a summer mean target of 5 µg/L means that blooms will almost never occur, while a target of 10 µg/L implies that such blooms will be rare. A target of 20 µg/L suggests blooms will occur about 15-20 percent of the time, which is suggested as the maximum allowable level consistent with full support of contact recreation use. A target mean concentration of 25 corresponds to blooms about one quarter of the time.

Table 1
Frequency of Chlorophyll *a* Concentrations Greater than 30 µg/L Using the Method of Walker (1985)

Summer Mean Chlorophyll *a* (µg/L)	CV = 0.62	CV = 0.26	CV = 0.17
5	0.4 %	0.0 %	0.0 %
10	5.9 %	1.2 %	0.1 %
20	16.7 %	17.7 %	14.1 %
25	20.4 %	26.8 %	27.1 %

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3 Calculation of Chlorophyll *a* Target Based on Water Clarity

In addition to the chlorophyll *a* target, another endpoint of potential interest in lakes and slow moving rivers is the effect of algal density on water clarity, especially light transmission. This has important consequences for both ecological uses (e.g., support of submerged aquatic vegetation) and aesthetic uses (appearance of the water column). Light transmission is measured in various ways, including the extinction coefficient (rate at which light is attenuated in the water column), the depth to 1 percent light, turbidity, and Secchi depth (the depth to which a disk of specific characteristics can be seen). Turbidity is actually a measure of light scattering, rather than light transmission, and is thus not an optimal measure for evaluating water clarity in lakes and reservoirs. The extinction rate is most satisfying from a technical perspective; however, Secchi disk depth is the most commonly used and most easily understood measure.

These various measures are related to one another. The depth to 1 percent light penetration, z, is related to the extinction coefficient, K_e, through (Thomann and Mueller, 1987):

$$z = 4.61 / K_e \, ,$$

while the extinction coefficient is also approximately related to Secchi disk depth (SD, m) through (Sverdrup et al., 1942; Becton, 1956; cited in Thomann and Mueller, 1987):

$$K_e \approx 1.8 / SD \, .$$

The extinction coefficient in turn can be partitioned into portions due to algal and non-algal components. While several relationships have been proposed, Thomann and Mueller recommend use of the equation of Riley (1956):

$$K_e' = 0.0088\,B + 0.054\,B^{2/3},$$

where K_e' is the contribution to the extinction coefficient (m^{-1}) due to algae, and B is the chlorophyll *a* concentration (µg/L).

We assume that targets for water clarity changes due to algal growth are most likely to be expressed as an allowable decline in Secchi depth (e.g., a decline of 0.5 m is the maximum acceptable decline). From the equations above, the contribution of algae to a decline in Secchi depth is a nonlinear function such that determination of an acceptable change in algal density to meet a target specified as a change in Secchi depth is dependent on the baseline conditions. Let the baseline (optimal target) value of Secchi depth be SD_0 at chlorophyll *a* concentration B_0, with corresponding contribution to extinction coefficient of $K_e'_0$. Then, the allowable change in extinction coefficient, ΔK_e, in response to the allowable change in Secchi depth, ΔSD is given by

$$\Delta K_e = \frac{1.8}{SD_0 + \Delta SD} - \frac{1.8}{SD_0} \, ,$$

and the target chlorophyll *a* criterion can be obtained by solving for B in

$$K_e'_0 + \Delta K_e = 0.0088\,B + 0.054\,B^{2/3} \, .$$

Rather than solving the equation directly for B, the BATHTUB spreadsheet tool uses an optimization routine to find the value of B for a given ΔK_e and $K_e'_0$.

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4 Using the BATHTUB Spreadsheet Tool

This section provides instructions for using the California BATHTUB Lake Model Tool. The tool is a Microsoft Excel spreadsheet, and is intended to be a simple but effective tool for predicting growing season chlorophyll *a* lake response to a number of inputs. The tool also allows the user to specify a chlorophyll *a* target, and predicts the probability that current conditions will exceed the target, as well as showing allowable N and P loading combinations necessary to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in Secchi depth.

The tool has two sheets for user input and viewing results and two sheets for calculations. The majority of the user input fields and model results are located on the **Main** sheet. The user input section has the following inputs:

USER INPUTS

Lake Volume	0.6	10^6 m3
Surface area	72050	m2
Average Depth (calc)	8.33	m
Mixed depth	2	m
Net Evap-Precip rate	10	in/per
Secchi depth at typical Chl a	1.5	m
Typical Chl-a	10	µg/L

BATHTUB Calibration Factors

Phosphorus (Kp)	1
Nitrogen (Kn)	1
Chlorophyll a (Kc)	1
Secchi Depth (Ks)	1
Oxygen Depletion (Khod)	1

Delivered Loads for Period of Interest

P Load	640.00	kg
N Load	6405.60	kg
Ortho P	609.73	kg
Inorg N	6048.29	kg
Inflow	1.36	hm3

Target Value for Chlorophyll-a

Chl-a target	25.0	µg/L

Calculate Target from water clarity: Click Here

Other Parameters

Initial Concentration of Dissolved Oxygen

(affects the calculation of oxygen demand)

Initial DO	10	mg/L

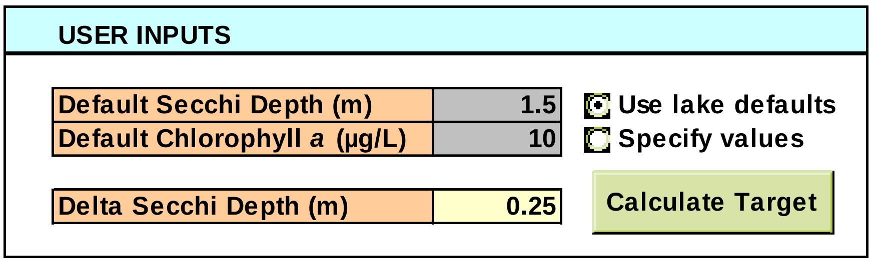
Covariance of Natural Log of Chl-a

(affects the calculation of exceedance probabilities)

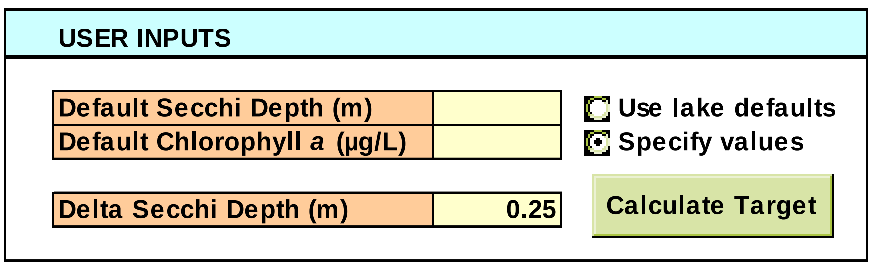
CV(InChla)	0.42

Instead of specifying a chlorophyll *a* target directly, the user might want to calculate the target based on an allowable reduction in Secchi depth. To do this, the user must click the button in the ***Target Value for Chlorophyll a*** section:

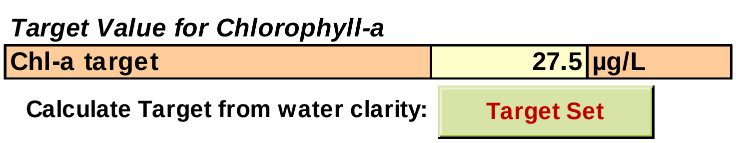
This transfers the user to the **Target** sheet:



By default, the Secchi depth and chlorophyll *a* entered on the **Main** sheet is transferred to the **Target** sheet. The user may also test a different Secchi depth and chlorophyll *a* combination by selecting **Specify Values**:



Once values have been chosen, the user clicks the **Calculate Target** button. The user is returned to the **Main** sheet, and the calculated chlorophyll *a* target is transferred to the **Chl-a target** entry:



The button now reads **Target Set** to indicate that the selected target is based on the water clarity calculation.

Results are displayed in three section on the Main sheet:

- **Summary Results**.
- **Chl-a Exceedance Probabilities** – the probability that daily chlorophyll *a* concentrations will exceed specific levels.
- **N-P Frontier** – those combinations of N and P loads that just meet the chlorophyll *a* target.

Examples of each follow:

The Summary Results section gives the BATHTUB predictions of average concentrations. The user can adjust the calibration factors to achieve a better agreement with observations.

Summary Results

Growing Season Average Chl a	53.4	µg/L
Predicted Median Secchi Depth	0.6	m
Growing Season P Conc.	0.15	mg/L
Growing Season N Conc.	1.85	mg/L
Oxygen Depletion, hypolimnion	11.4	d
Oxygen Depletion, metalimnion	21.9	d

California BATHTUB Lake Model Tool, v11 (3-21-06)

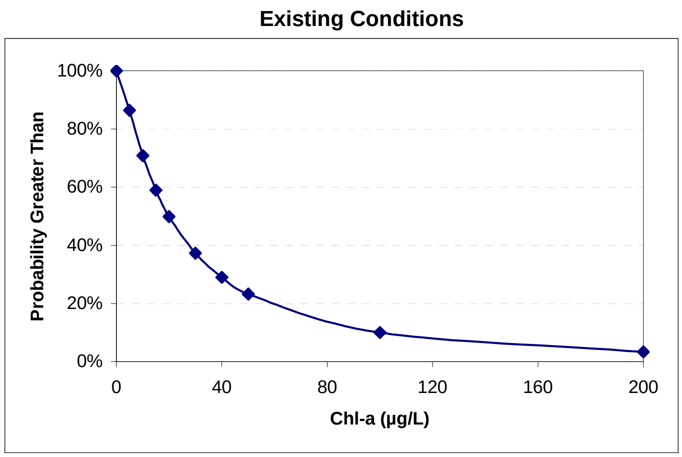
The Results section provides the bloom probabilities, while the N-P Frontier sections shows the allowable load combinations to meet the target.

RESULTS

Chl-a Exceedance Probabilities

Concentration (µg/L)	*Probability Greater Than*
0	1
5	86.44%
10	70.84%
15	58.95%
20	49.88%
30	37.24%
40	28.96%
50	23.21%
100	9.96%
200	3.32%

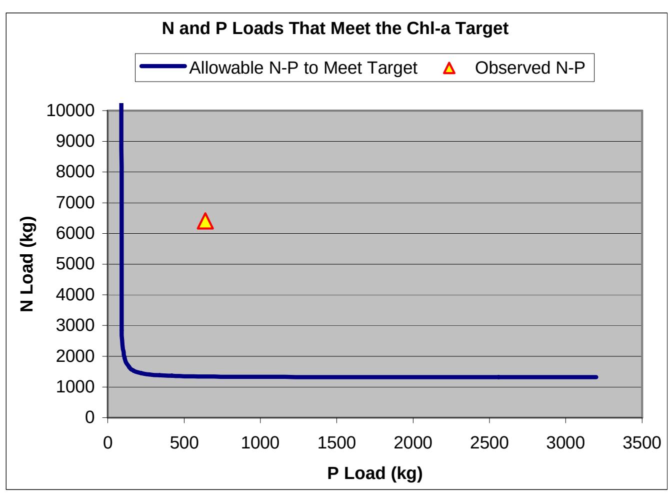
Target (µg/L)	*Probability Greater Than*
25	42.83%



N-P Frontier (Allowable N-P Loading to Meet Chl-a Target)

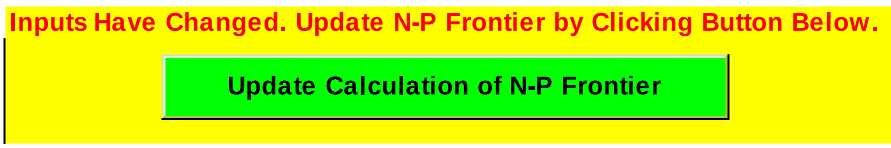
P Load (kg)	*N load (kg)*
70	##############
90	2,838
110	1,958
140	1,644
170	1,530
220	1,447
270	1,409
340	1,381
420	1,364
530	1,350
660	1,342
830	1,335
1,040	1,330
1,300	1,326
1,630	1,323
2,040	1,321
2,560	1,320
3,200	1,318

Update Calculation of N-P Frontier



The N-P Frontier calculations cannot be performed on-the-fly in Excel and require the use of a back-end Visual Basic routine embedded in the Excel workbook, executed by clicking the following button:

Whenever the user adds or changes input fields affecting the frontier calculation, the tool alerts the user to click the button and run the routine to obtain new results:



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