White Paper (a)

A Risk-based Approach to Development of Nutrient Criteria and TMDLs

J. B. Butcher

Tetra Tech, Inc.

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1 Introduction

1.1 EPA National Approach to Nutrient Criteria

The process for developing nutrient criteria for the region started in 1998 with the publication of the *National Strategy for the Development of Regional Nutrient Criteria* (EPA 1998). EPA then proceeded to develop national criteria recommendations on the basis of aggregated Level III ecoregions. Data sets from Legacy STORET, NASQAN, NAWQA and EPA Region10 were used to assess nutrient conditions from 1990 to 1998. Reference conditions were then developed based on 25th percentiles of all nutrient data including a comparison of reference condition for the aggregate ecoregion versus the subecoregions. These 25th percentile values were characterized as criteria recommendations that could be used to protect waters against nutrient over-enrichment (USEPA, 2000). However, EPA also noted that States and Tribes may "need to identify with greater precision the nutrient levels that protect aquatic life and recreational uses. This can be achieved through development of criteria modified to reflect conditions at a smaller geographic scale than an ecoregion such as a subecoregion, the State or Tribe level, or specific class of waterbodies." EPA also encouraged that States and Tribes "critically evaluate this information in light of the specific designated uses that need to be protected."

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

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

1.2 Proposed California Approach

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

This white paper sets out a theoretical justification for the proposed California approach, as well as practical suggestions for its implementation. Specifically, the development of nutrient criteria and nutrient TMDLs is evaluated in terms of the *risk* of impairment of designated uses. Several concepts from EPA's (1998) *Guidelines for Ecological Risk Assessment* – in particular, the use of conceptual models and surrogate measures (or indicators) – are particularly useful for the development of nutrient criteria and the estimation of nutrient TMDLs. The risk-based approach is described in Section 2. Section 3 discusses potential indicators and targets for nutrient assessment, while Section 4 shows how these targets can be incorporated into a tiered approach to nutrient criteria. Section 5 provides an overview of potential tools available for evaluating targets. Finally, Section 6 demonstrates the connection between a risk-based approach to criteria and the development of nutrient TMDLs.

2 Uses and Impairments – A Risk-Based Approach

The approach taken for the California pilot study is to propose nutrient criteria based on an evaluation of risk relative to designated beneficial uses. Essentially, the objective is to control excess nutrient loads/concentrations to levels such that the risk or probability of impairing the 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 thus 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, waterbodies 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 wildlife. Elevated nutrients, while most likely not posing a toxicological threat, could stimulate primary productivity and result in increased food supplies or shelter for aquatic life. Excessive primary productivity, however, could result in depletion of oxygen supplies and impact aquatic life.

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

Wildlife Habitat: 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 (SPAWN), and Warm Freshwater Habitat (WARM).

2.2 RISK-BASED APPROACH

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

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

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

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

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

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

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

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

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

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

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

2.3 Conceptual Models of Nutrient Impairment

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

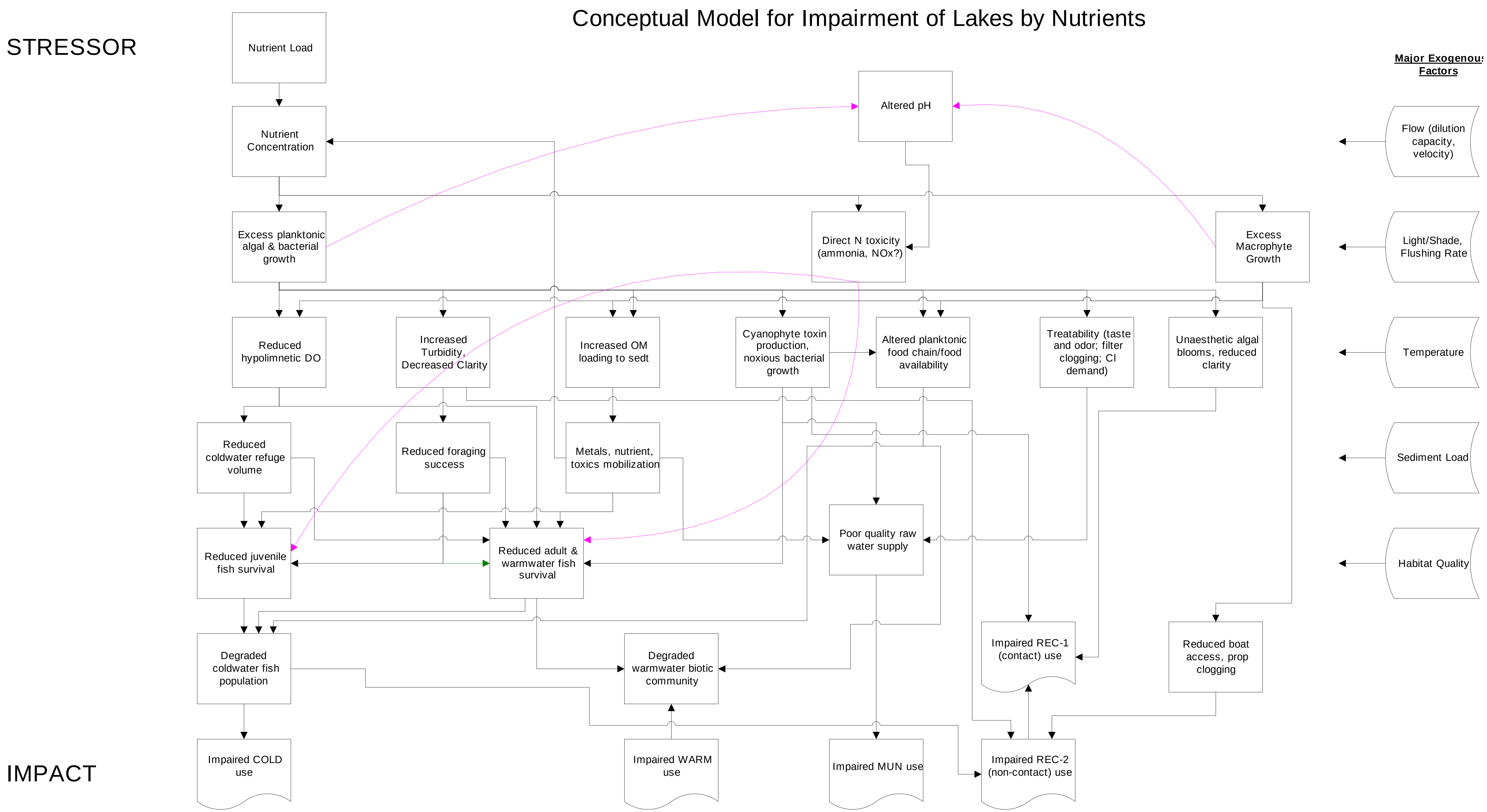

Figure 1. Conceptual Model for Lakes

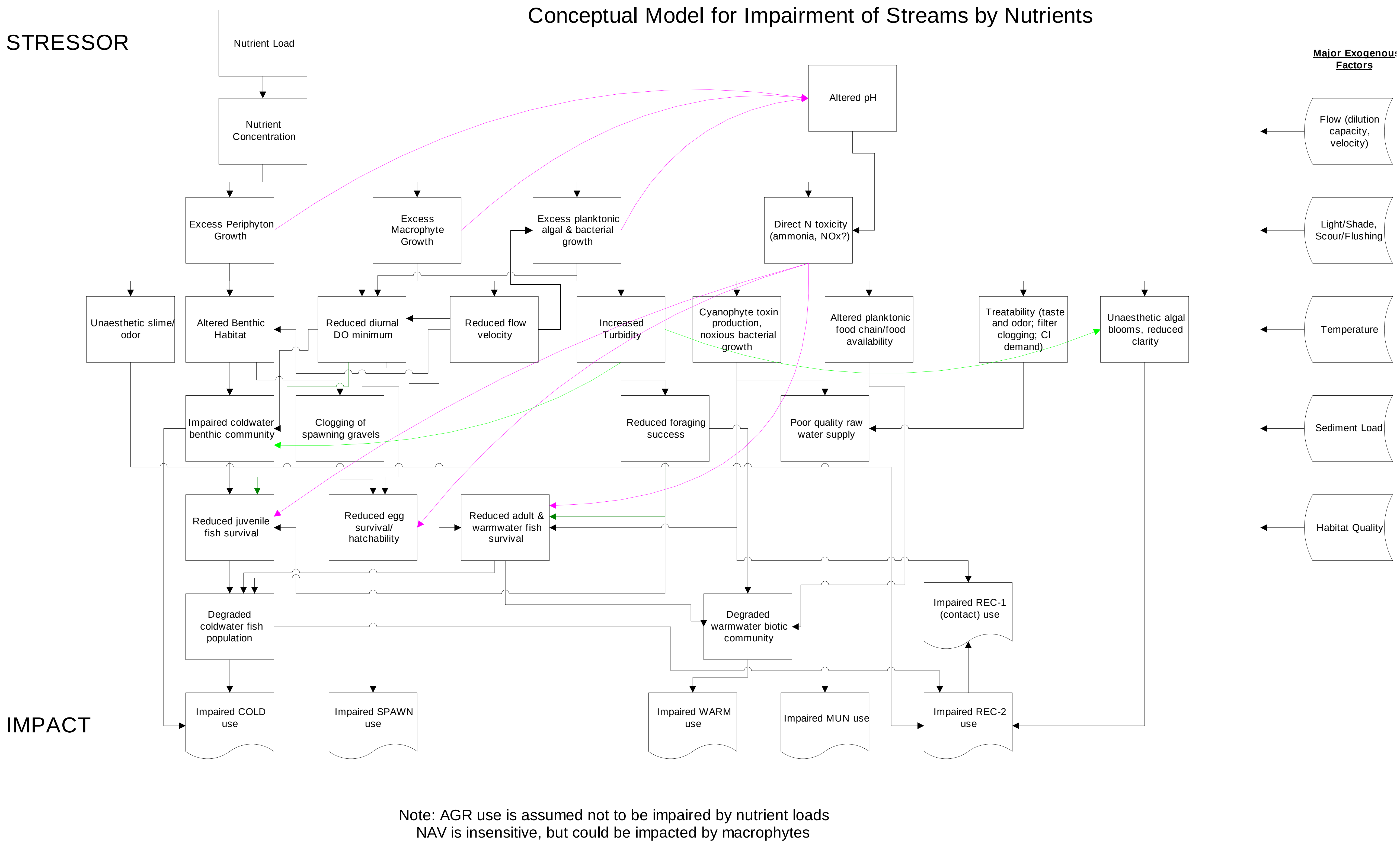

Figure 2. Conceptual Model for Streams

2.4 RISK HYPOTHESES

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

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

The complex conceptual models may first be reduced to a table showing the relationship of key uses to major stressor response factors that can be key causes of impairment of use, as shown in Table 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:

<u>Lakes/Reservoirs</u>

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

<u>Rivers/Streams</u>

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

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

Table 1. Stressor-Response Factors

Use	Key Stressor-Response Factors								Seconday Factors
	Reduced hypolimnetic DO	Increased Turbidity	Cyanophyte toxins	Altered food chain	Toxic metal, NH4 cycling	Taste & Odor	Unaesthetic Blooms	Excess Macrophytes	
COLD	X	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

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 (U.S. EPA, 1998). The TMDL and Watershed Approach literature tends to refer to these measures as “indicators”.

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

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 2.4 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 waterbodies. Other states have focused on intermediate measures. For instance, Georgia and Alabama assign chlorophyll *a* criteria to lakes and, if impairment is assessed, allocate nutrient loads on a site-specific basis to meet these criteria.

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

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

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

3.2 Indicators and Targets for Lakes and Reservoirs

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

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

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

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

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

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 the direct supply of nutrients from the sediment.

Linkages can, however, be built successfully between nutrient concentrations and potential maximum periphytic algal growth in the absence of other limiting factors, as summarized in Section 5.3. As with lakes and reservoirs, the linkage analysis should include a representation of the most significant co-factors to set site specific requirements. However, because of the uncertainty inherent in predicting nutrient response in streams it may be preferable to use multiple lines of evidence to set stream nutrient criteria.

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

Downstream Requirements

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

4 A Tiered Approach to Criteria

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

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

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

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

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

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

> ***Tier I***. Impacts unlikely (use is supported).
>
> ***Tier II***. Probably sustaining (but potentially threatened).
>
> ***Tier III***. Impacts likely (use is not supported or highly threatened).

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

> ***Tier I/II Criterion***: Conceived foremost as a concentration or load below which impacts are unlikely. However, the Tier I/II Criterion should also take into account the natural

background concentration likely to be present, and should not be set less than the natural background. If natural background results in an impairment of a designated use, then the use designation is inappropriate and should be refined.

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

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

Many individual waterbody segments will fall into the "gray area" of Tier II, in which impairment is a reasonable possibility, but not a certainty. These are the waterbodies for which a more detailed and site-specific analysis is warranted. The first step in processing candidate Tier II waterbodies would be to develop site-specific targets (as discussed in Section 3), using linkage analysis tools (e.g., models) that help account for site characteristics (as briefly summarized in Section 5). If such a site-specific target is exceeded, the waterbody could then be moved to the Tier III category (likely nonsupporting). Where the site-specific target is met, the waterbody could remain in the Tier II (probably sustaining) category, but an antidegradation analysis would be warranted in connection with any proposed changes in permitted discharges.

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

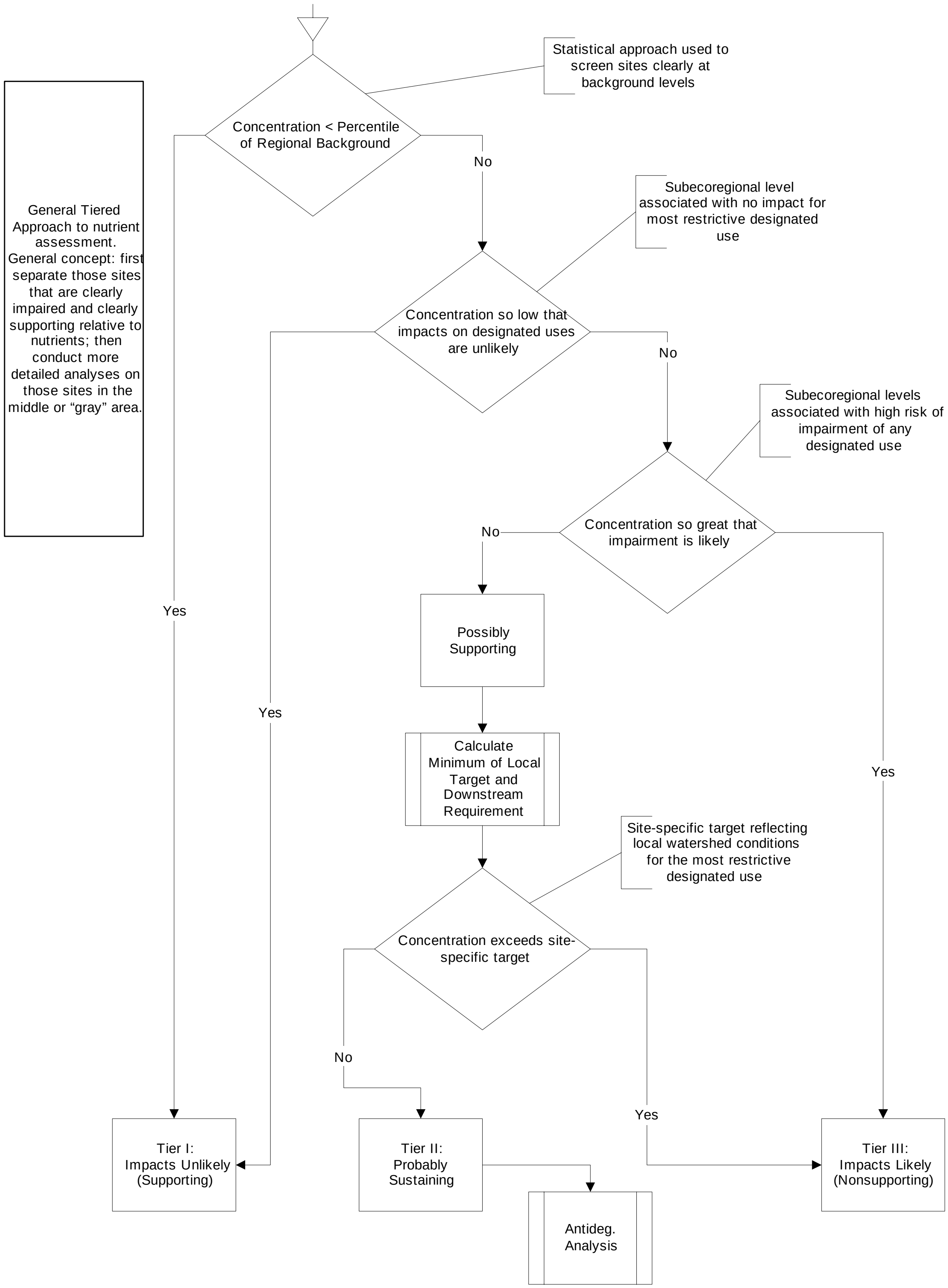

Figure 3. Tiered Approach to Nutrient Criteria Implementation

5 Methods for Evaluating Targets

Methods for evaluating nutrient targets relative to designated uses are needed to set the Tier I/II and Tier II/III boundaries, and, most importantly, to evaluate more site-specific targets relative to waterbodies that initially fall within Tier II. Detailed development of methods is not the subject of this white paper, and will be discussed elsewhere. However, some brief and general notes relative to methods for evaluating targets are presented here.

5.1 TYPES OF APPROACHES

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

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

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

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

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

- Models can provide a process-based interpretation of observed data, thus helping to sort out multiple causes.

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

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

5.2 METHODS FOR LAKES AND RESERVOIRS

5.2.1 Statistical and Reference Site Approaches

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

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

5.2.2 Empirical Approaches

A number of empirical approaches to predicting lake response have been developed. In most cases these predict average chlorophyll *a* concentrations as a function of phosphorus (or phosphorus and nitrogen) loading and hydraulic residence time. An excellent summary is provided in Reckhow and Chapra (1983). While expected to hold in general, these relationships have not, to our knowledge, been fully validated on California lakes.

5.2.3 Modeling Approaches

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

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

Extensive work with BATHTUB was reported in Section 4.4 of Tetra Tech's 2003 Progress Report on the Ecoregion 6 Pilot Study. This work demonstrated the calculation of normalized loading-response curves, such as the example shown here as Figure 4. Once a target level of chlorophyll *a* is assigned to the support a specific use (as an intermediate measure), these relationships can be used to derive loading criteria.

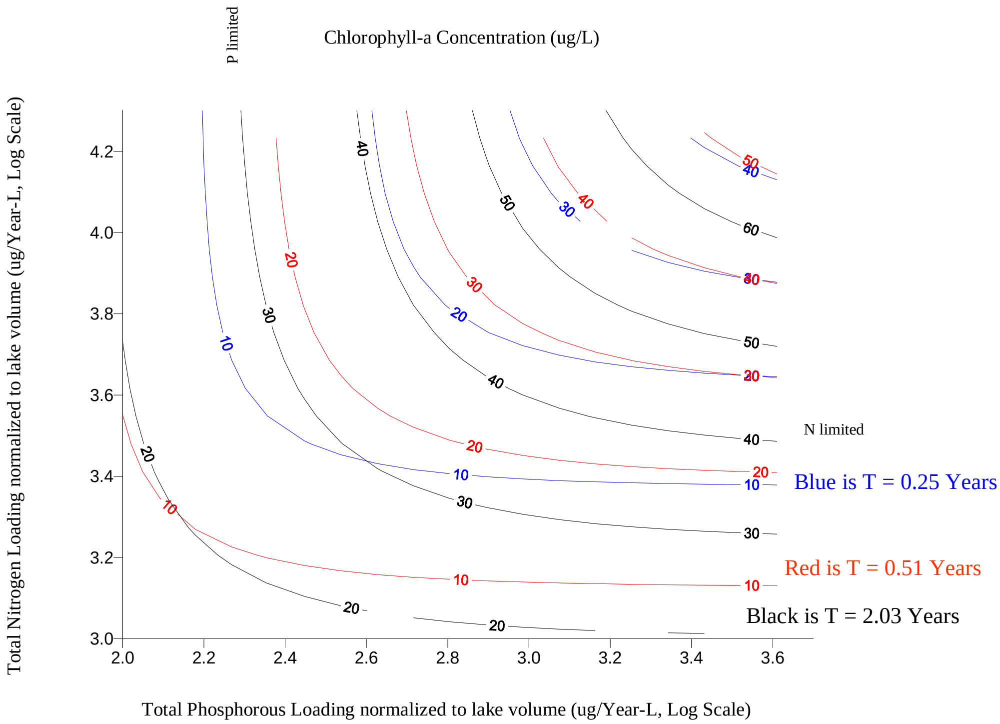

Figure 4. Example use of BATHTUB as a method for establishing lake targets (from Tetra Tech, 2003) . Lake Chlorophyll-a concentration versus phosphorus and nitrogen loading (normalized to lake volume) for residence times of 0.25, 0.51, and 2.03 years, and non-algal turbidity of 1.25 m.

5.3 Rivers and Streams

5.3.1 Statistical and Reference Site Approaches

Use of reference sites is more promising for streams than reservoirs in most of California, if only because unimpacted reference streams are somewhat easier to find. The approach is still subject to the general limitation of not providing an exact criterion. For developing the Tier I/II criterion, the generalized reference site or statistical method proposed by EPA is of some relevance, but only when the analyses are conducted at an appropriate sub-ecoregional level. Preliminary results of this type of analysis as applied to Ecoregion 6 are reported in Tetra Tech (2003).

5.3.2 Empirical Approaches

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

5.3.3 Modeling Approaches

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

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

5.4 Natural Background Nutrient Loading

Setting the Tier I/II criterion boundary also requires estimation of natural background concentrations or loads. As with the target development, this can proceed via a weight-of-evidence approach using several different sources. The different approaches are presented in Tetra Tech (2003) and summarized below.

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

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

6 Application to TMDLs

Total Maximum Daily Load (TMDL) development for nutrients is in many ways the inverse of criteria development, particularly when the criteria are developed on a risk basis, as is proposed here. Criteria development is focused on the determination of target values that will minimize the risk of impairment of uses. A TMDL begins with an assessment of impairment, and then determines allocations that will meet a target that is estimated to result in attainment of uses (plus a margin of safety). Where the criteria are risk-based targets that vary by site they should be closely related to the targets that will be used in a nutrient TMDL.

The TMDL process is established under §303(d) of the Clean Water Act, which requires States and tribes to define water quality goals for a water body by designating the uses to be made of the water bodies and setting criteria necessary to protect those uses. The designated uses and criteria together constitute water quality standards, which should, wherever attainable, achieve a level of water quality that provides for the protection and propagation of fish, shellfish, and wildlife, and other uses as applicable. Water bodies not meeting these standards are then placed on the States §303(d) list of impaired waters.

TMDLs are used to provide for more stringent water quality-based controls when technology-based controls on point and nonpoint sources, are inadequate to achieve state water quality standards (EPA 1991a).

In recent years, USEPA (1999a, 1999b, 2001) has published a series of protocols containing recommendations for States and tribes to follow when developing TMDLs. The protocols identify a recommended set of six components (plus an external component for Implementation), which are shown in Figure 5 and summarized below.

1. *Problem Identification* lists the key factors and the nature of the impairment and context for the TMDL. This step is a guiding factor in the remainder of the TMDL development.
2. *Identification of Water Quality Indicators and Targets*. The protocols recommend identifying numeric or measurable indicators and target values that can be used to evaluate attainment of standards. Where impairment is related to a narrative water quality standard, the narrative standard is interpreted to develop a quantifiable target value to measure attainment.
3. *Source Assessment* identifies the type, magnitude, and location of stressor sources
4. *Linkage Analysis* defines the cause-and-effect relationship between the pollutant of concern and the stressor sources, usually accomplished through modeling.
5. *Allocations* are based on the linkage analysis, and specify the assignment of stressor source loadings that are estimated to be consistent with attaining the standards. A Margin of Safety is usually included in the allocations.
6. *Follow-up Monitoring and Evaluation* is recommended for all TMDLs to determine whether water quality standards are attained.

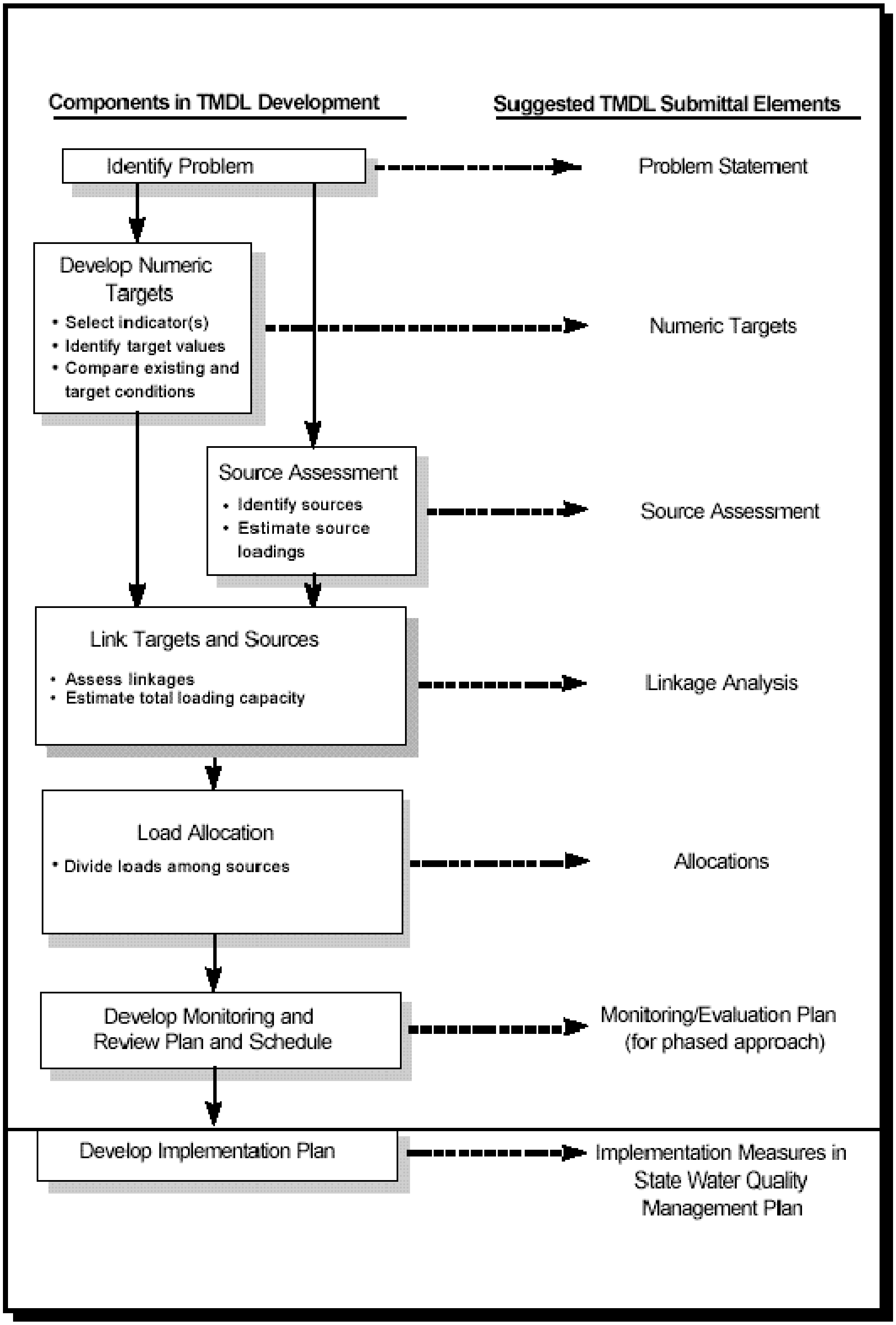

Figure 5. General Components of TMDL Development in the *Protocols* (USEPA, 1999a)

Steps 2, 3, and 4 in the recommended TMDL process are closely related to the nutrient criteria development process outlined in preceding sections of this white paper. TMDL indicators and targets are the same type of objects as were discussed in Section 3 for criteria development. Source assessment in a TMDL is by definition site specific; however, the key determination of the proportion of source loads that is attributable to natural backgrounds is also an issue for criteria development, as was discussed in Section 5.4. Finally, the Linkage Analysis for the TMDL may make use of the same tools as were proposed for evaluating criterion targets (Section 5).

TMDL development for a listed segment is likely to involve more detailed, site-specific analyses than are proposed for criteria development; however, the general processes and many of the specific tools used will be similar. Where the criteria are explicitly tied to an analysis of the risk of not attaining uses, the estimation of a TMDL (which is a tool for attaining uses) should be fully consistent with the criteria development process.

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