White Paper

The Development of Nutrient Criteria

For Ecoregions Within:
California, Arizona, and Nevada

Prepared for:
US EPA Region IX Regional Technical Advisory Group
&
CA SWRCB State Regional Board Advisory Group

Prepared by:
Tetra Tech, Inc.
3746 Mt. Diablo Boulevard. Suite 300
Lafayette, CA 94549-3681
(925) 283-3771
EPA Contract No. 68-C-99-249

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1.0 INTRODUCTION

Development of regional nutrient criteria is one component of a larger strategy to address water quality problems associated with nutrient overenrichment and culturally-induced accelerated rates of eutrophication of waterbodies in the U.S. The U.S. Environmental Protection Agency (EPA)and Department of Agriculture have several active program initiatives to address the nutrient overenrichment problem. These programs address point and nonpoint sources of pollution, evaluate public health impacts from animal feeding operations, conduct research and monitoring to provide data and assessment techniques to better characterize the problem, and have offer nutrient management policies to provide practical support to agricultural operations to reduce the export of nutrients from their lands. The purpose of the regional nutrient criteria development process is to provide numeric targets for the various nutrient management programs that are regionally appropriate by reflecting geographic variations of waterbody response to nutrients.

The current nutrient criteria development process for California and EPA Region IX (California, Nevada, Arizona, and Hawaii) started with the publication of the National Strategy for the Development of Regional Nutrient Criteria (EPA 1998) in June 1998.

1.1 PURPOSE AND SCOPE

The RTAG /STRTAG is a diverse stakeholder group that has undertaken the task of developing nutrient criteria that protect designated uses using scientifically defensible methods and appropriate water quality data. This white paper is an important milestone and tool in that process. The white paper was specifically proposed as a pre-draft work plan that would allow the RTAG and STRTAG stakeholders to develop a common understanding of the language, concepts, options, procedures needed to develop nutrient criteria. The white paper will be the primary topic of discussion at a two day RTAG and STRTAG workshop scheduled for March 14 and 15 in San Diego, California. This document outlines the recommended process, goals, strategies, options for regionalization categories, and causal and response parameters to be considered as nutrient criteria. The choices or preferences developed during the two-day workshop will form the basis of the draft work plan for the region.

There are several features of the nutrient criteria process that present unusual opportunities not typically associated with criteria development. The ecoregional approach is attuned to natural rather than political boundaries. In addition, the involvement of regional teams that can choose to lead the process for their region is also precedent setting. The current RTAG and STRTAG work group framework has successfully responded to these opportunities and will continue to address the challenges associated with development of nutrient criteria for the region. A high level of collaboration between states and regional boards in the affected areas is the most logical response since ecoregion boundaries are not consistent with political boundaries. There is not much sense in having several different entities developing nutrient criteria for the same ecoregion. The RTAG and STRTAG forum also provides an effective forum for coordinating the use of resources for various facets of the process (e.g., monitoring, public and stakeholder involvement). The overarching goal of this initiative is the development through a science-based process of common sense nutrient criteria that serve as a numeric target for nutrient management decisions that protect water quality.

1.2 OVERVIEW OF NUTRIENT CRITERIA DEVELOPMENT PROCESS

The process to develop nutrient criteria for the region started in 1998 with the publication of the *National Strategy for the Development of Regional Nutrient Criteria* (EPA 1998). The process described in this section is illustrated in Figure 1-1. EPA Region IX made an early commitment to the regional team concept 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 th e criteria document that had been completed for rivers and streams. The comparison tables for total phosphorus and total nitrogen are included as Table 1-1. The tables 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. This decision committed the RTAG and STRTAG to the development of a work plan for the region.

Figure 1-1.

Key Milestones and Draft Schedule for Nutrient Criteria Development

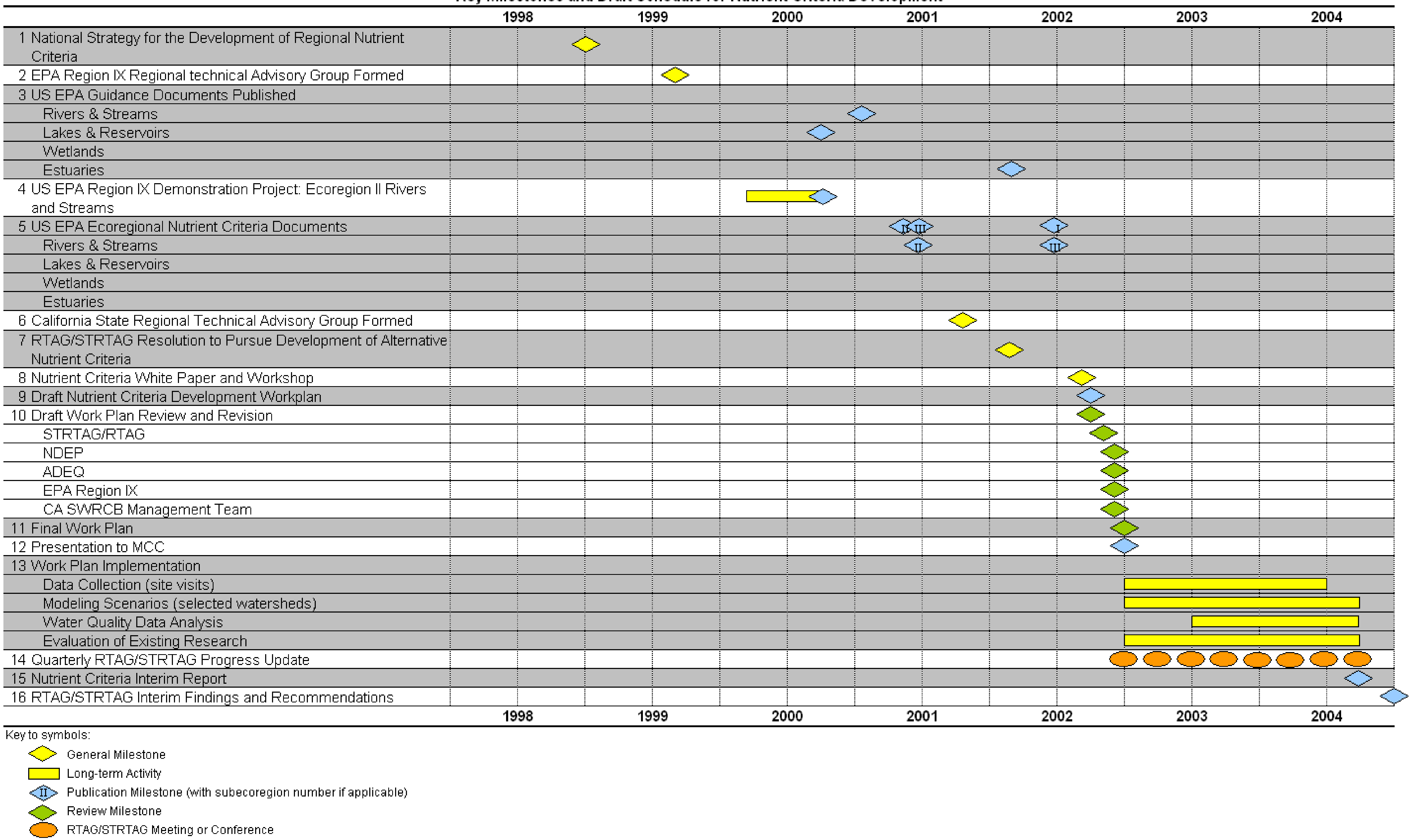


Table 1-1. Nutrient Data Plot Summary

Ecoregion*	Total Phosphorus (approx. mg/L)					Total Nitrogen (approx. mg/L)				
	304(a) Criterion	Reference 75%	% > 304(a)	STORET 25%	% > 304(a)	304(a) Criterion	Reference 75%	% > 304(a)	STORET 25%	% > 304(a)
1	0.010	0.03	70	0.01	70	0.13	na	na	0.17	85
5	0.015	0.04	85	0.02	85	0.29	0.36	33	0.22	62
6	0.030	na	na	0.06	88	0.50	na	na	0.40	69
8	0.011	na	na	0.002	44	0.52	na	na	0.10	17
9	0.030	0.13	67	na	na	0.15	0.40	97	na	na
14	0.010	0.03	47	0.03	80	0.67	0.25	0	0.55	66
22	0.015	0.07	62	0.02	97	0.23	0.48	60	0.18	47
23	0.011	0.06	85	0.005	85	0.28	0.48	58	0.13	47
24	0.018	0.07	56	na	na	0.62	0.32	12	na	na
78	0.032	0.05	28	0.12	98	0.53	0.58	25	na	na

***Ecoregion Key:**

1 Coastal Range
5 Sierra Nevada
6 Southern and Central California Chaparral and Oak Woodlands
8 Southern California Mountains
9 Eastern Cascades Slopes & Foothills
14 Southern Basin & Range
22 Arizona/New Mexico Plateau
23 Arizona/New Mexico Mountains
24 Southern Deserts
78 Klamath Mountains

This white paper is the first key milestone towards the development of the work plan. The RTAG and STRTAG will use the March workshop to select the nutrient criteria development options to include in the draft work plan. The draft work plan will be ready for RTAG and STRTAG review during May 2002. Each participating organization will also conduct their review of the work plan. The final review by the California SWRCB Management Team is significant because with their approval and presentation to the SWRCB Management Coordinating Committee (MCC) the contract project team will be working in close association with the Regional Boards to implement the plan for the development of nutrient criteria for California. Presentation to the MCC is anticipated for June 2002.

Implementation of the work plan will commence as soon as it has received the approval of the California SWRCB. This will include members of the contract project support team making site visits to Regional Boards and other potential sources of water quality data. In addition, the project team will begin laying the groundwork for the modeling scenarios and the review of site-specific studies. Progress will be reported to the RTAG and STRTAG on a regular (quarterly) basis. The final key milestones for this phase of this work plan will be the development of the interim report on nutrient criteria for the region and its presentation for review by the RTAG and STRTAG. The interim report will include recommendations for as many of the regionalization units as possible. Following response to RTAG and STRTAG review comments the final report will be submitted to the Regional Boards to begin the necessary administrative steps for adopting the approved nutrient criteria into their Basin Plans.

1.3 PROPOSED APPROACH

The proposed approach for developing nutrient criteria in the region is to supplement the EPA criteria document by enhancing the water quality reference database and to evaluate information related to an effects-based approach. There are three components of strategy:

1. Conduct multivariate empirical data analysis to better define regionalization units; enhance regional distribution datasets for subecoregions, and to evaluate the relationship between nutrient inputs and water quality endpoints.
2. Develop modeling scenarios to supplement empirical empirical nutrient distribution data, and to evaluate relationships between various parameters.
3. Compile a synthesis of existing site-specific studies to evaluate the performance of selected waterbodies under various nutrient conditions (i.e., are designated uses supported?).

Level III ecoregions described by Omernik (1987) and other physical classification criteria serve as the basis for the approach. Figure 1-2 is an example decision diagram that illustrates the relationship of the primary decision components for lakes and reservoirs. It is important to note that the illustrated decision components for physical classification have not yet been determined and those included in the diagram are hypothetical. The same caveat applies to Figure 1-3 for rivers and streams.

Figure 1-2. Decision Diagram for Lakes and Reservoirs

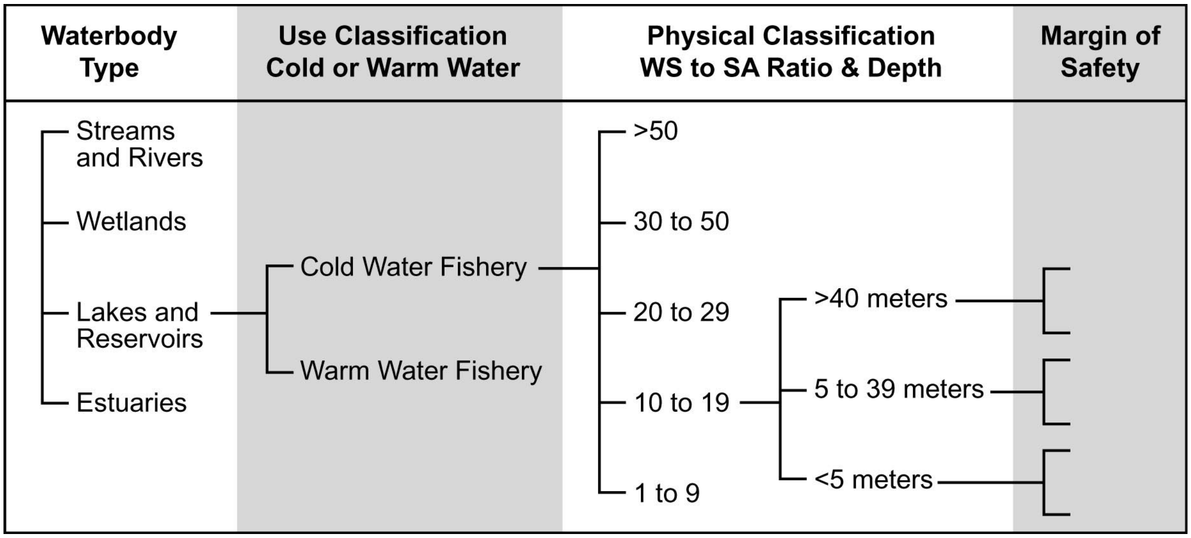
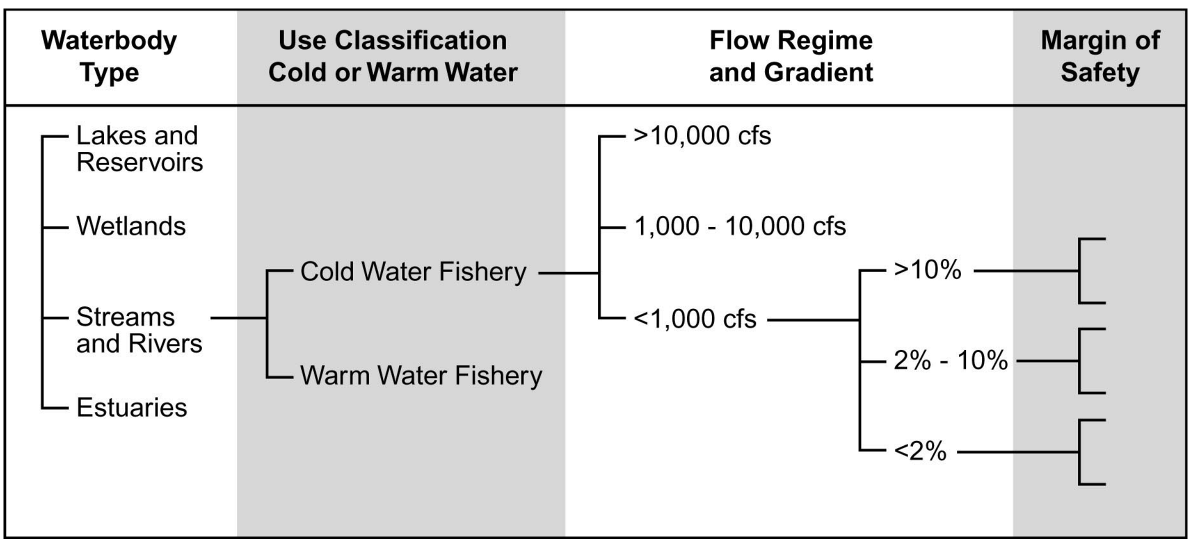


Figure 1-3. Decision Diagram for Lakes and Reservoirs



Implicit in the figure is that there will be a similar decision diagram each of the 16 Level III ecoregions within EPA Region IX. Determining the type of waterbody is the first decision point. Thus far the approach is based on the waterbody types addressed in EPA's criteria documents. Typically waterbodies are assigned more than one Designated Uses by the state or Regional Board. The objective of this decision component is to identify the Designated Use that is most sensitive nutrient inputs. The next decision component involves assigning a waterbody to its appropriate physical classification category. The physical classification decision component may include more than one classfication criteria. The framework eventually branches to the final decision point where a protective range of the criteria parameter(s) is assigned.

The approach described in this white paper that is being proposed for use in the region to develop nutrient criteria is consistent with recent draft guidance that was distributed at a recent meeting of EPA Regional Nutrient Coordinators (Appendix A). Appendix A describes a hypothetical state decision process to determine whether or not to proceed with development of alternate nutrient criteria. Appendix A also includes a decision pathway for development and implementation of a nutrient criteria development work plan. The process depicted in Appendix A is consistent with the process that has been developed by the RTAG and STRTAG that is described in Figure 1. EPA has issued additional draft guidance that provides an outline and format for a nutrient criteria development plan. This guidance has been included as Appendix B. One purpose of the white paper and the associated workshop is to develop a work plan that is consistent with the example in Appendix B. EPA has also prepared additional information in response to State question on issues related to nutrient criteria development. These additional issues are included in Appendix C. Most of the issues in Appendix C have been raised, considered and addressed by the RTAG and STRTAG and will be documented in the work plan to be prepared for the region. The RTAG and STRTAG will evaluate several options that are include in the white paper as responses to the issues included in Attachment 3. In addition to the draft EPA guidance that has been provided in Appendices 1 through 3, the Arizona Department of Environmental Quality (ADEQ) has allowed us to include a draft of their proposed strategy for development of nutrient criteria (Appendix D). The approach being proposed in the white paper is also consistent with the draft strategy proposed by ADEQ. Since the participating states also share several ecoregions the high level of consistency in work plans should allow for collaboration in the nutrient criteria development process.

1.4 ORGANIZATION OF THIS WHITE PAPER

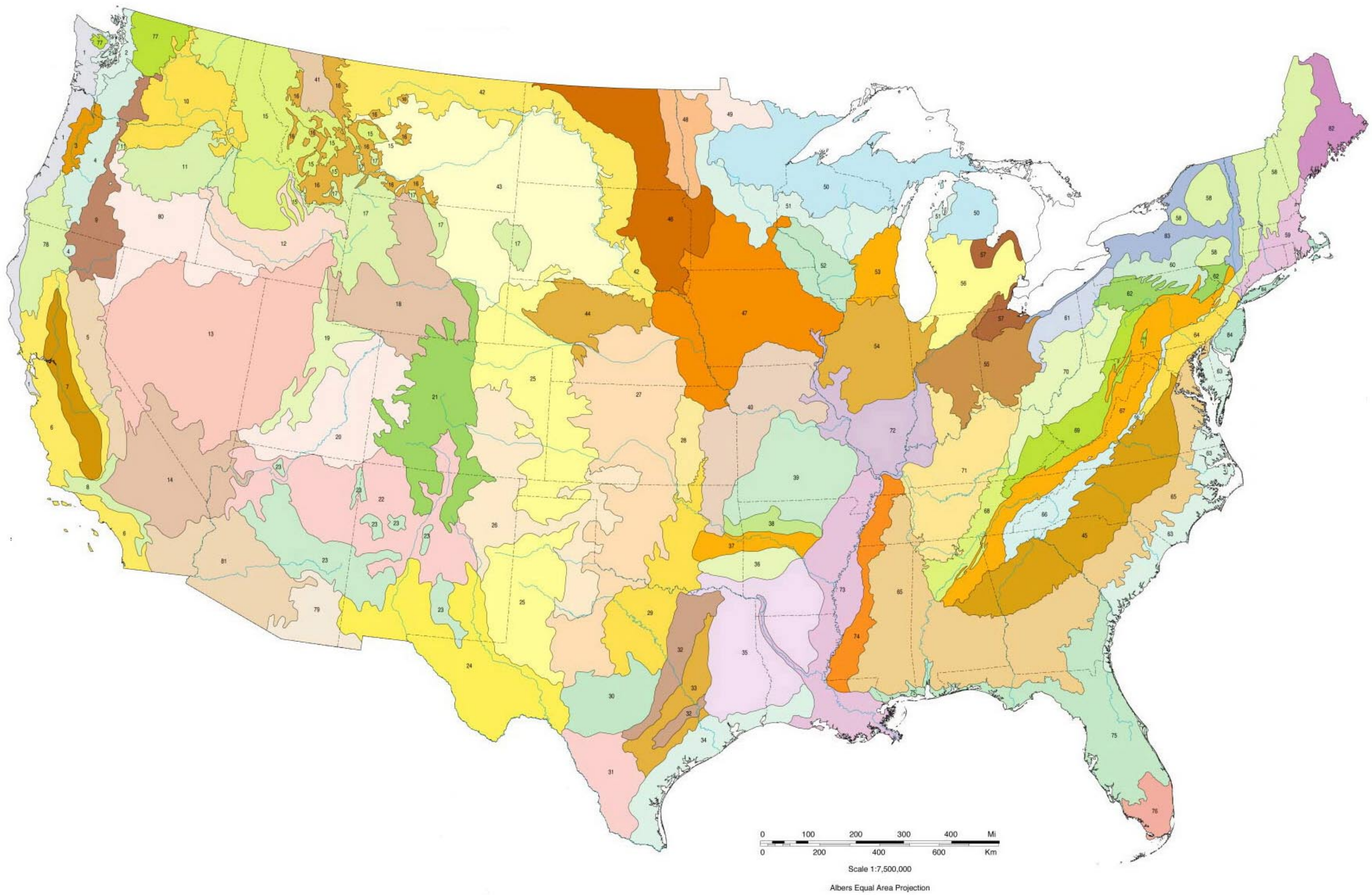
Section 1 of this document provides an overview of the nutrient criteria process including both past and future milestones. Section 2 explains the basis of the regional approach and identifies potential confounding factors that will need to be addressed outside the nutrient criteria framework Section 3 discusses various issues related to the form of the standard, such as how to incorporate temporal and spatial variability into the regulatory framework. Section 4 discusses various options for establishing regional classifications for waterbodies to ensure that waterbodies within a category respond to similar inputs of nutrients in a similar manner. Section 5 evaluates a wide-range of parameters that could be used in the criteria. Section 6 describes potential next steps and outlines various task areas that will be addressed in the work plan.

2.0 REGIONALIZATION UNITS

Landscape- and local-scale factors influence the expression of waterbody habitats. Landscape-scale factors, such as climate, geology, and vegetation operate over large areas, are stable over long time periods (hundreds to thousands of years) and act to shape the overall character and attainable condition within drainage networks. Local-scale factors are a function of ultimate factors and refer to local conditions of geology, landform, and biotic processes that operate over smaller areas (e.g., stream reach scales) and over shorter time spans (years to decades). A hierarchical classification system that integrates both landscape-scale factors and local-scale factors provides the organizational framework necessary to address the spatial variability inherent in aquatic habitats.

The EPA National Strategy for the Development of Regional Nutrient Criteria (EPA 1998) identified use of geographic regions as one of the five primary elements of their proposed approach. The initial recommendation for this element was to divide the nation into aggregated ecoregions based on Omernik's (1987) original Level III ecoregions (Figure 2-1) (84 ecoregions cover the continental U.S.). The RTAG pilot study that evaluated the use of the aggregated ecoregions indicated that the aggregated ecoregions were too course to capture the variability in inherent nutrient levels and nutrient responses throughout the region. Therefore the RTAG / STRTAG has adopted the original Level III ecoregions described in Omernik (1987). The ecoregions within EPA Region IX (excluding Hawaii and territories) are briefly described in section 2.1 below.

Figure 2-1. Omernik's Level III Ecoregions for the Continental U.S. (1987)



It is necessary to further stratify the ecoregions into more refined regional units in order to achieve the RTAG and STRTAG goal of grouping waterbodies into categories that respond in a similar manner to similar levels of nutrient inputs. Section 2.2 lists landscape variables that will be evaluated to determine which classification categories minimize variability in regional waterbody conditions and response. It is possible that ecoregions will use different sets of stratification criteria to achieve the reduced variability objective. However, it is important to keep the number of classification categories to a minimum because nutrient criteria recommendations must be developed for each category that is established. The decision framework algorithm is:

> # of Ecoregions (16) * # of waterbody types (2) * # of classification categories (number unknown) = the total number of nutrient criteria to be developed.

The work plan will attempt to achieve a balance between defining criteria too coarsely and requiring the development of site-specific criteria for each waterbody. Section 4 provides a more detailed description of each landscape variable to be considered.

The development of more refined regional categories through the use of stratification criteria will not eliminate the need for exceptions to address waterbodies not consistent with criteria that have been developed for other surrounding or adjacent waterbodies. There are several confounding factors that may require the development of site-specific nutrient criteria or that they may be exempted from the numeric criteria approach altogether. Several confounding factors that affect waterbodies within the region are briefly discussed in Section 2.3.

2.1 ECOREGIONS

There are 16 Level III ecoregions in EPA Region IX. Twelve ecoregions are represented in California, six in Arizona and four in Nevada. A description for each of the 16 ecoregions in the region is included in Table 2-1. EPA has published preliminary nutrient criteria for the rivers/streams and lakes/reservoirs located in these subecoregions (Tables 2-2 and 2-3). The preliminary criteria presented in these tables demonstrate the significant variations among subecoregions with respect to ambient nutrient levels.

Table 2-1. Region IX Level III Ecoregion Descriptions

No.	Ecoregion Name	CA	NV	AZ	Description
1	Coast Range	✓			Highly productive, rain-drenched coniferous forests cover the low mountains of the Coast Range. Sitka spruce and coastal redwood forests originally dominated the fog-shrouded coast, while a mosaic of western red cedar, western hemlock, and seral Douglas fir blanketed inland areas. Today Douglas fir plantations are prevalent on the intensively logged and managed landscape.
78	**Klamath Mountains**	✓			The ecoregion is physically and biologically diverse. Highly dissected, folded mountains, foothills, terraces, and floodplains occur and are underlain by igneous, sedimentary, and some metamorphic rock. The mild, subhumid climate of the Klamath Mountains is characterized by a lengthy summer drought. It supports a vegetal mix of northern Californian and Pacific Northwest conifers.
4	**Cascades**	✓			This mountainous ecoregion is underlain by Cenozoic volcanics and has been affected by alpine glaciations. It is characterized by broad, easterly-trending valleys, steep ridges in the west, a high plateau in the east, and both active and dormant volcanoes. Elevations range upward to 4,390 meters (14,400 feet). Its moist, temperate climate supports an extensive and highly productive coniferous forest. Subalpine meadows occur at high elevations.
9	**Eastern Cascades, Slopes and Foothills**				The Eastern Cascade Slopes and Foothills are in the rainshadow of the Cascade Mountains. Its climate exhibits greater temperature extremes and less precipitation than ecoregions to the west. Open forests of ponderosa pine and some lodgepole pine distinguish this region from the higher ecoregions to the west where spruce fir forests are common, and the lower dryer ecoregions to the east where shrubs and grasslands are predominant. The vegetation is adapted to the prevailing dry continental climate and is highly susceptible to wildfire. Volcanic cones and buttes are common in much of the region.
5	**Sierra Nevada**	✓	✓		The Sierra Nevada is a deeply dissected block fault that rises sharply from the arid basin and range ecoregions on the east and slopes gently toward the Central California Valley to the west. The eastern portion has been strongly glaciated and generally contains higher mountains than are found in the Klamath Mountains to the northwest. Much of the central and southern parts of the region are underlain by granite as compared to the mostly sedimentary formations of the Klamath Mountain s and volcanic rocks of the Cascades. The higher elevations of this region are largely federally-owned and include several national parks. The vegetation grades from mostly ponderosa pine at the lower elevations on the west side and lodgepole pine on the east side, to fir and spruce at the higher elevations. Alpine conditions exist at the highest elevations.

No.	Ecoregion Name	CA	NV	AZ	Description
80	**Northern Basin and Range**	✓	✓		This ecoregion consists of arid tablelands, intermontane basins, dissected lava plains, and widely scattered low mountains. The bulk of the region is covered by sagebrush steppe vegetation. The ecoregion is drier and less suitable for agriculture than the Columbia Plateau, is higher and cooler than the Snake River Basin to the east, and contains a lower density of mountain ranges than the adjacent Central Basin and Range ecoregion to the south. Much of the region is used as rangeland.
13	**Central Basin and Range**	✓	✓		The Central Basin and Range ecoregion is characterized by a mosaic of xeric basins, scattered low and high mountains, and salt flats. Compared with the Snake River Basin and Northern Basin and Range regions to the north the region is hotter and contains higher and dense mountains that have perennial streams and ponderosa pine forests at higher elevations. Also, there is less grassland and more shrub land, and the soils are mostly Aridisols rather than dry Mollisols. The region is not as hot as the Mojave and Sonoran Basin and Range ecoregions and it has a greater percentage of grazed land.
14	**Mojave Basin and Range**	✓	✓		This ecoregion contains scattered mountains that are generally lower than those of the Central Basin and Range. Potential natural vegetation in this region is predominantly creosote bush, compared with the mostly saltbush-greasewood and Great Basin sagebrush of the ecoregion to the north, and creosote bush-bur sage with large patches of palo verde-cactus shrub and saguaro cactus in the Sonoran Basin and Range to the south. Most of this region is federally owned and there is relatively little grazing activity because of the lack of water and forage for livestock. Heavy use of off-road vehicles and motorcycles in some areas has caused severe wind and water erosion problems.
7	**Central California Valley**	✓			Flat, intensively farmed plains with long, hot dry summers and cool wet winters distinguish the Central California Valley from its neighboring ecoregions which are either hilly or mountainous, forest or shrub covered, and generally nonagricultural. Nearly half of the region is in cropland, about three-fourths of which is irrigated. Environmental concerns in the region include salinity due to evaporation of irrigation water, groundwater contamination from heavy use of agricultural chemicals, wildlife habitat loss, and urban sprawl.
6	**Southern and Central California Chaparral and Oak Woodlands**	✓			The primary distinguishing characteristic of this ecoregion is its Mediterranean climate of hot, dry summers and cool, moist winters, and associated vegetative cover comprising mainly chaparral and oak woodlands; grasslands occur in some lower elevations and patches of pine are found at higher elevations. Most of the region consists of open low mountains or foothills, but these are areas of irregular plains in the south and near the border of the adjacent Central California Valley Ecoregion. Much of this region is grazed by domestic livestock; very little land has been cultivated.

No.	Ecoregion Name	CA	NV	AZ	Description
8	**Southern California Mountains**	✓			Like the other ecoregions in central and southern California, the Southern California Mountains has a Mediterranean climate of hot dry summers and moist cool winters. Although Mediterranean types of vegetation such as chaparral and oak woodlands predominate, the elevations are considerably higher in this region, the summers are slightly cooler, and precipitation amounts are greater, causing the landscape to be more densely vegetated and stands of ponderosa pine to be larger and more numerous than in the adjacent regions. Severe erosion problems are common where the vegetation cover has been destroyed by fire or overgrazing.
81	**Sonoran Basin and Range**	✓		✓	Similar to the Mojave Basin and Range to the north, this ecoregion contains scattered low mountains and has large tracts of federally owned land, most of which is used for military training. However, the Sonoran Basin and Range is slightly hotter than the Mojave and contains large areas of palo verde-cactus shrub and giant saguaro cactus, whereas the potential natural vegetation in the Mojave is largely creosote bush
79	**Madrean Archipelago**			✓	Also known as the Sky Islands in the U.S., this is a region of basins and ranges with medium to high local relief, typically 1,000 to 1,500 meters (3,280 to 4,921 feet). Native vegetation in the region is mostly grama-tobosa shrubsteppe in the basins and oak-juniper woodlands on the ranges, except at higher elevations where ponderosa pine is predominant. The region has ecological significance as both a barrier and a bridge between two major cordilleras of North America, the Rocky Mountains and the Sierra Madre Occidental.
23	**Arizona/New Mexico Mountains**			✓	The Arizona/New Mexico Mountains are distinguished from neighboring mountainous ecoregions by their lower elevations and an associated vegetation indicative of drier, warmer environments, which is also due in part to the region's more southerly location. Forests of spruce, fir, and Douglas fir, that are common in the Southern Rockies and the Uinta and Wasatch Mountains, are only found in a few high elevation parts of this region. Chaparral is common on the lower elevations, pinyon-juniper and oak woodlands are found on lower and middle elevations, and the higher elevations are mostly coved with open to dense ponderosa pine forests.
22	**Arizona/New Mexico Plateau**			✓	The Arizona/New Mexico Plateau represents a large transitional region between semiarid grasslands and low-relief tablelands of the Southwestern Tablelands ecoregion in the east, the drier shrub lands and woodland-covered higher relief tablelands of the Colorado Plateau in the north, and the lower, hotter, less vegetated Mojave Basin and Range in the west and Chihuahuan Deserts in the south. Higher, more forest-covered, mountainous ecoregions border the region on the northeast and southwest. Local relief in the region varies from a few meters on plains and mesa tops to well over 300 meters (984 feet) along tableland and side slopes.

No.	Ecoregion Name	**CA**	**NV**	**AZ**	**Description**
24	**Chihuahuan Deserts**			✓	This desert ecoregion extends from the Madrean Archipelago in southeastern Arizona to the Edwards Plateau in south-central Texas. The region comprises broad basins and valleys bordered by sloping alluvial fans and terraces. Isolated mesas and mountains are located in the central and western parts of the region. Vegetative cover is predominantly arid grass and shrub land, except on the higher mountains where oak-juniper woodlands occur.

Table 2-2. Ambient Water Quality Criteria Recommendations for Rivers and Streams

No.	Ecoregion Name	Total Phosphorus (µg/L)			Total Nitrogen (µg/L)			Chlorophyll α (µg/L)			Turbidity (FTU)		
		Min	Max	25th	Min	Max	25th	Min	Max	25th	Min	Max	25th
1	Coast Range	0.63	522.5	10.2	0.05	1.88	0.13	1.99	14.23	2.53	0.25	72.5	1.5
78	Klamath Mountains	5.63	455	32.5	0.53	0.53	0.53*	0.75	6.3	1.15	0.68	33.81	1.5
4	Cascades	0	242.5	9.1	0	0.37	0	0.58	12.75	1.01	0.68	13.5	1.75
9	Eastern Cascades, Slopes and Foothills	4.38	752.5	30	0.11	3.1	0.15	0.43	53	2.95	0.33	66.5	1.61
5	Sierra Nevada	2.5	485	15	0.20	0.91	0.29	--	--	--	0.38	26.25	0.62
80	Northern Basin and Range	10	333.7	55	0.42	1.7	0.48	0.6	4.3	2.85	0	28.5	2
13	Central Basin and Range	2.5	2,150	28.8	0.23	5.55	0.42	2.1	60.35	3.26	0.35	102	1.92
14	Mojave Basin and Range	5	515	10	0.57	1.21	0.67	--	--	--	0.88	37.56	3.92
7	Central California Valley	11	1,900	77	0.35	2.26	0.35*	0.9	15.3	1.6	3.23	21	7.13
6	Southern and Central CA Chapparal and Oak	2.5	3,212	30	0.22	9.95	0.5	2.39	2.39	2.39*	1	35.9	1.9
8	Southern CA Mountains	10.9	10.9	10.9*	0.52	0.52	0.52*	--	--	--	1.05	1.05	1.0*
81	Sonoran Basin and Range	0	1,485	25	0.3	10.3	0.61	--	--	--	1.7	116.2	2.4
79	Madrean Archipeligo	7.5	675	10	0.34	0.48	0.35	--	--	--	4.1	27	4.1*
22	Arizona/New Mexico Plateau	0	12,787	15	0.04	4.44	0.23	--	--	--	3.4	25.8	5.1
23	Arizona/New Mexico Mountains	0	357.5	11.2	0.08	0.89	0.28	--	--	--	0.9	26	2.0
24	Chihuahuan Deserts	2.5	1462	17.5	0.43	3.22	0.62	0.25	10.53	0.25	0.48	37.8	2.1

Total number of sub-ecoregions:
EPA Region IX = 16
California = 12
Arizona = 6
Nevada = 4

*If fewer than 4 streams were used in developing a seasonal quartile and or all-seasons median, the entry is flagged.
NA = no lakes are found in the ecoregion.

ND = criteria recommendations document does not exist.

Table 2-3. Ambient Water Quality Criteria Recommendations for Lakes and Reservoirs

No.	Ecoregion Name	Total Phosphorus (µg/L)			Total Nitrogen (µg/L)			Chlorophyll □ (µg/L)			Secchi (m)		
		Min	Max	25^{th}	Min	Max	25^{th}	Min	Max	25^{th}	Min	Max	25^{th}
1	Coast Range	5.0	35.4	7.10	0.19	0.19	0.19*	1.8	7.6	2.3	1.0	6.8	5.1
78	Klamath Mountains	40	160	40*	--	--	--	--	--	--	2.0	2.2	2.2*
4	Cascades	1.5	98.0	6.25	0.00	0.34	0.00	0.3	41.4	0.9	0.0	10.5	5.6
9	Eastern Cascades, Slopes and Foothills	65	191.2	68.80	1.16	2.15	1.16	4.7	44.5	4.7*	3.5	6.1	4.4
5	Sierra Nevada	15	100	15	0.25	0.25	0.25	--	--	--	--	--	--
80	Northern Basin and Range	86	90	86	--	--	--	4.4	5.2	4.4	1.7	3.7	2.8
13	Central Basin and Range	11	742	30	0.50	2.37	0.51	1.8	46.2	3.5	0.1	4.9	2.3
14	Mojave Basin and Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
7	Central California Valley	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
6	Southern and Central CA Chapparal and Oak	55	309	172	--	--	--	24.6	24.6	24.6*	0.9	1.9	1.9*
8	Southern CA Mountains	--	--	--	--	--	--	--	--	--	--	--	--
81	Sonoran Basin and Range	20	20	20*	--	--	--	--	--	--	1.7	1.7	1.7*
79	Madrean Archipeligo	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
22	Arizona/New Mexico Plateau	2	135	15	0.23	1.51	0.31	1.1	4.4	2	0.7	4	2.9
23	Arizona/New Mexico Mountains	9.06	107.5	12.5	0.44	3.07	0.88	2.4	21.5	6.1	0.4	1.9	1.8
24	Chihuahuan Deserts	13	67	22	0.45	1.30	0.57	0.6	30.3	3.3	0.4	2.9	1.5

Total number of sub-ecoregions:
EPA Region IX = 16
California = 12
Arizona = 6
Nevada = 4

*If fewer than 4 lakes used in developing a seasonal quartile and or all-seasons median, the entry is flagged.
NA = no lakes are found in the ecoregion.
ND = criteria recommendations document does not exist.

2.2 STRATIFICATION CRITERIA

Because nutrients, unlike toxic pollutants, are naturally present in all water bodies at greater or lower levels depending on their inherent characteristics (e.g., slope, underlying geology, watershed area, etc.), the numeric nutrient criterion cannot be a single number that applies nationally, or even to a state. We must divide water bodies in a geographic region into different groups, with typical natural nutrient levels for each group. Nutrients in a water body are considered to be a source of pollution when they exceed levels natural of that type of water body. For the purpose of this white paper, we term this grouping of water bodies into different categories based on ecoregion and physical characteristics as stratification. Table 2-4 includes a list of the stratification criteria that are discussed in greater detail in Section 4 of this report.

Table 2-4. Potential Stratification Criteria used to Classify Waterbodies

Stratification Criteria	Rivers and Streams	Lakes and Reservoirs
Ecoregion	✓	✓
Beneficial Uses	✓	✓
Land Use/Watershed Characteristics	✓	✓
Underlying Geology	✓	
Stream Order	✓	
Size/Shape	✓	✓
Downstream Waterbody	✓	
Flow	✓	✓
Downstream Loading	✓	
Stream Gradient (slope)	✓	
Width/Depth Ratio	✓	
Entrenchment Ratio	✓	
Sinuosity	✓	
Channel Materials	✓	
Location		✓
Lake Type		✓
Water Quality		✓
Stratification		✓
Lake Origin		✓
Age		✓
Dam Operation		✓
Fish Community		✓
Other Biological Characteristics		✓

2.3 CONFOUNDING FACTORS

A completely comprehensive nutrient criteria development strategy is not a feasible goal: There will always be waterbodies that are outliers or exceptions which will require site-specific consideration. This section identifies and briefly discusses some of the more common confounding factors that will result in a site-specific approach.

2.3.1 Water Transfers

The arid West has been the host region for some of the largest water diversion, storage, and transfer projects on earth. This causes a problem for the ecoregions-based nutrient framework when large amounts of water are transferred from one ecoregion to another. The transferred water can overwhelm or significantly alter the ecoregional characteristics of the receiving waterbody. An example is the transfer of water from the Colorado River basin into the Santa Margarita River in Southern California. At a minimum the total dissolved solids present in the water transferred from the Colorado River water has significantly altered the characteristics and quality of the ground water of the Santa Margarita watershed. Waterbodies that have had a significant portion of their flow contributed from other ecoregions need to undergo an evaluation to determine if a site-specific criterion is called for.

2.3.2 Effluent Dominated Waterbodies

Effluent dominated waterbodies (EDWs) are, by definition, exceptions to other waterbodies within an ecoregion. Arizona has developed a designated use classification for EDWs. EDWs represent a category that will include a large number of waterbodies. California will soon be undertaking the development of an approach for evaluating EDW issues and their relationship to Designated Uses and water quality standards. A primary concern with EDWs is that in many locations they are not isolated waterbodies, but rather they often flow into more sensitive receiving waterbodies.

2.3.3 Highly Engineered Waterbodies

The term highly engineered waterbodies (HEWs) refers to waters that have been created or significantly modified through engineering, to the extent that they no longer reflect ecoregion conditions. Examples of HEWs include aqueducts, concrete lined reservoirs, and agricultural drainage tiles. It is possible that natural waterbodies have been so extensively modified that they also cannot reflect or exhibit ecoregion conditions, such as portions of the Los Angeles River.

2.3.4 Waterbodies That Cross Ecoregional Boundaries

Ecoregions are delineated on maps with distinct boundaries. There are many examples of rivers and streams that may have their origin in one ecoregion and pass through or into another downstream. The characteristics of a river passing from one ecoregion into another will not change at the ecoregion boundary. Waterbodies that cross ecoregion boundaries require evaluation to determine where ecoregion criteria would apply. The work plan should identify the rivers that fall into this category and attempt to delineate river reaches to appropriate ecoregions.

3.0 FORM OF THE STANDARD

This section addresses the parameters that will be measured as well as how often and in what location they will be measured to determine compliance. Although it is generally understood that nutrient criteria will be defined in terms of chemical concentrations, it need not necessarily be so. For the purpose of defining nutrient criteria, we broaden the potential metrics to include (in addition to various chemical species of nitrogen and phosphorus), dissolved oxygen, turbidity, chlorophyll a, and indices of biological integrity. All of these metrics will exhibit gradients in water bodies due to uptake and cycling by biota, and also due to transport and dilution; these gradients may be both spatial and temporal. Examples of gradients with depth for phosphorus and nitrogen species, and dissolved oxygen are shown in Figures 1 and 2. Figure 3 shows the spatial and temporal trends of nitrogen species with depth and time. Because algae and macrophytes in surface waters take up nutrients as they grow, they exert an influence on measured concentrations. There are two temporal cycles of interest: diurnal and seasonal. The definition of the standard needs to consider both cycles.

Figure 3-1. Generalized vertical distribution of soluble (PS) and total (PT) phosphorus in stratified lakes of very low (oligotrophic) and very high (eutrophic) productivity (Source: Wetzel 1983)

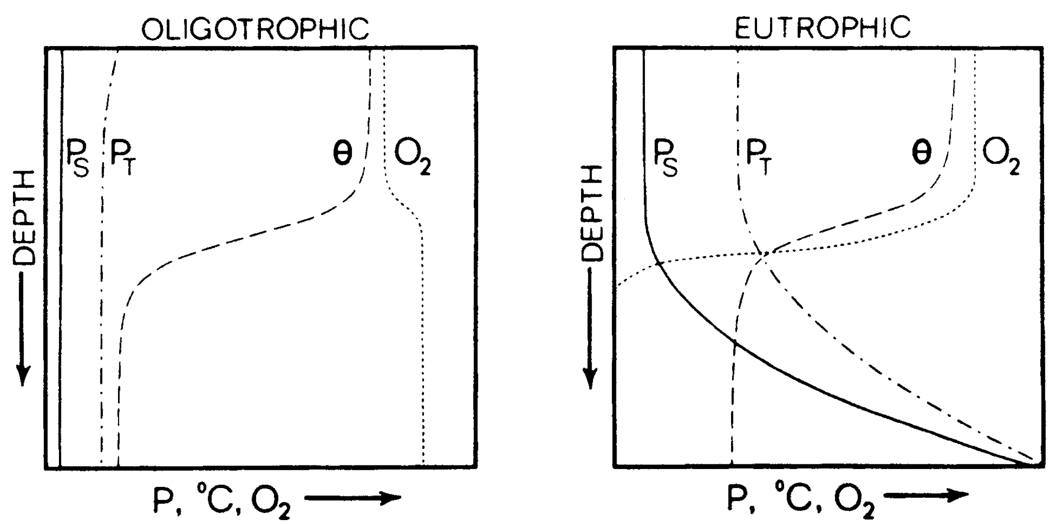


Figure 3-2. Generalized vertical distribution of ammonia and nitrate-nitrogen in stratified lakes of very low (oligotrophic) and very high (eutrophic) productivity (Source: Wetzel 1983)

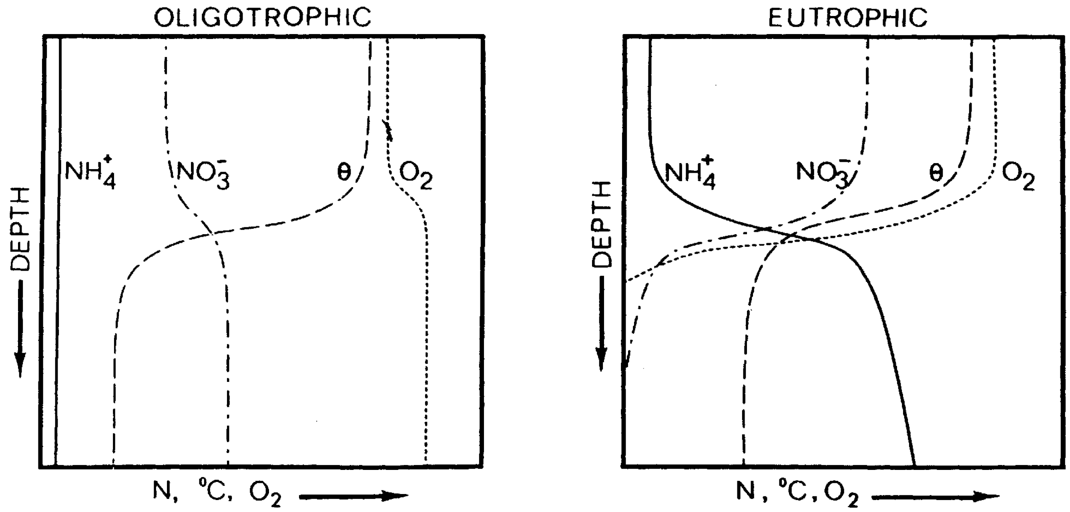
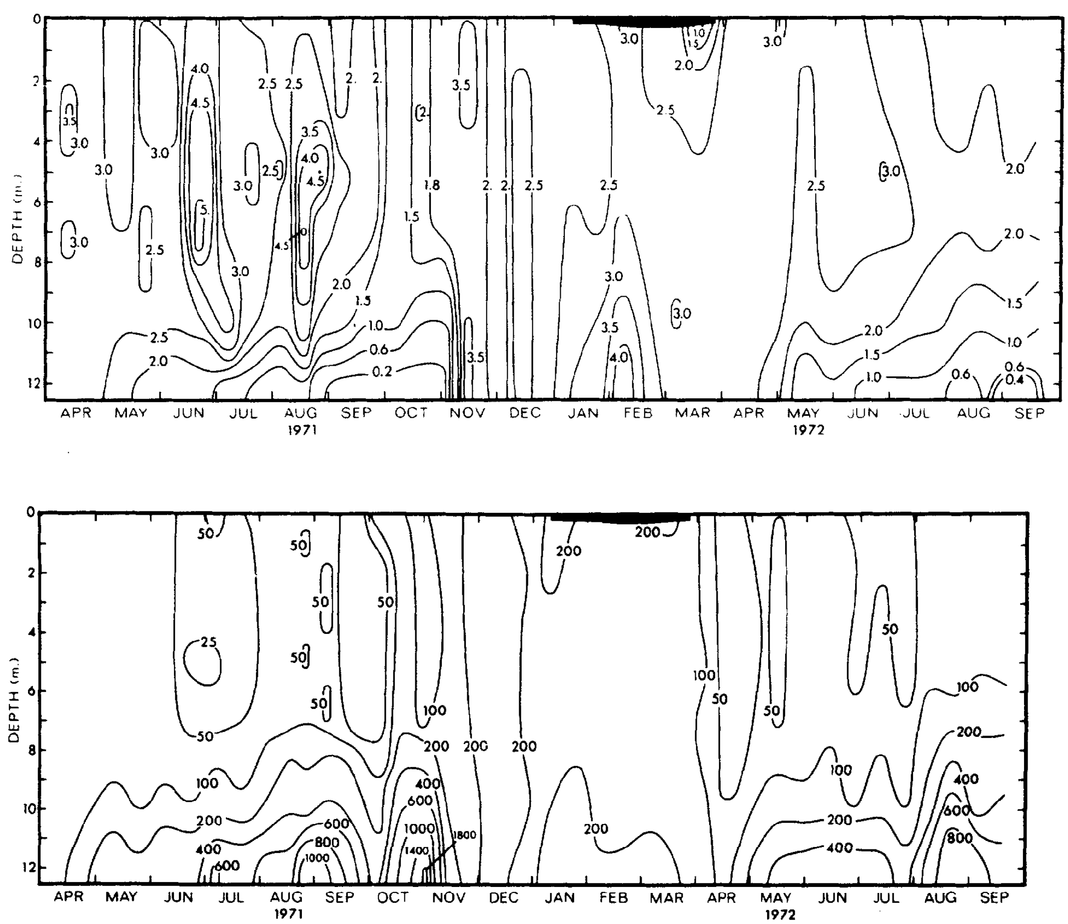


Figure 3-3. Depth-time diagrams of seasonal concentrations of NO3-N + NO2-N in mg/l (upper) and NH4-N in mg/l (lower) in Lawrence Lake, Michigan, 1971-72. Opaque areas represent ice cover (Source: Wetzel 1983)



3.1 TYPICAL NUTRIENT CONCENTRATIONS FOR WATER BODIES

In the absence of any specific information, some general comments can be made on the ranges of nutrient concentrations commonly observed in nature. Note that unlike toxic pollutants, nutrients are a natural component of the biogeochemical cycle; different water bodies may naturally have higher or lower levels of nutrients. Thus, some water bodies with elevated nutrient levels, may be naturally eutrophic (highly productive), and some may be naturally mesotrophic (less prodictive) or oligotrophic (very low productivity). Characterization of natural levels of nutrients has been performed most extensively for lakes. Typical levels of nutrients and chlorophyll for different lake classifications is presented in Table 3-1 (from Wetzel 1983).

Table 3-1. General Trophic Classification of Lakes and Reservoirs in Relation to Phosphorus and Nitrogen

Parameter (Annual Mean Values)	**Oligotrophic**	Mesotrophic	Eutrophic	Hypereutrophic
Total phosphorus (mg m-3)				
Mean	8.0	26.7	84.4	--
Range	3.0-17.7	10.9-95.6	16-386	750-1200
N	21	19	71	2
Total nitrogen (mg m-3)				
Mean	661	753	1875	--
Range	307-1630	361-1387	393-6100	--
N	11	8	37	--
Chlorophyll □ (mg m-3) of phytoplankton				
Mean	1.7	4.7	14.3	--
Range	0.3-4.5	3-11	3-78	100-150
N	22	16	70	2
Chlorophyll □ peaks (mg m-3) ("worst case")				
Mean	4.2	16.1	42.6	--
Range	1.3-10.6	4.9-49.5	9.5-275	--
N	16	12	46	--
Secchi Transparency (mg m-3)				
Mean	9.9	4.2	2.45	--
Range	5.4-28.3	1.5-8.1	0.8-7.0	0.4-0.5
N	13	20	70	2

Based on data of an international eutrophication program. Trophic status based on the opinions of the experienced investigators of each lake. (Modified from Vollenweider, 1979.)

3.2 NUMERIC PARAMETERS IN CRITERIA

What metric, or combination of metrics, should be used to define the numeric criteria for water bodies? These metrics might include chemical parameters such as phosphorus (soluble reactive phosphorus and total phosphorus) and nitrogen (total nitrogen, total organic nitrogen, nitrate, ammonia) species, and also direct and indirect biological parameters such as dissolved oxygen, chlorophyll a, Secchi depth, and turbidity. A locally-specified index of biological integrity might also be part of the standard, particularly where it can be more closely related to the designated use of the water body.

There are advantages and disadvantages to using each of the metrics listed here. Perhaps the most significant constraint is that much of the historical data available for water bodies is limited to chemical concentrations (i.e., the different nitrogen and phosphorus species). Thus, if we need to define a baseline reference condition from existing historical data, we are limited to the constituents that are most commonly measured. However, if we use only chemical parameters to define nutrient criteria, we may not always capture conditions that are most likely to cause impairment. As we work toward developing nutrient criteria, the RTAG must balance the advantages of using the historical database against generating a new metric (or combination of metrics) that may not have been measured in the past (but one which scientists think is a more accurate reflection of nutrient-related impairment).

3.3 SPATIAL AVERAGING

Because of the existence of spatial gradients in nutrient levels (and also other surrogate metrics) where we take measurements will influence the values we observe. Examples of gradients in nutrient concentrations are shown in Figures 1 through 3. Typically we need to obtain samples at several locations to obtain a representative picture of nutrient levels across a water body. The specification of the spatial averaging to be undertaken should be part of the standard. Some examples of ways to represent a spatial gradient are listed below:

- certain number of points per unit area (applicable to lakes and reservoirs, wetlands)
- depths at which measurements are to be made, or specify that only surface concentrations will be considered (lakes and reservoirs)
- number of measurements per river mile

This list must be supplemented by consideration of other factors that are responsible for generating spatial gradients in nutrient levels in water bodies.

3.4 TEMPORAL AVERAGING

Because of diurnal and seasonal cycles in the growth and uptake of nutrients by biota, nutrient concentrations and other surrogate parameters are influenced by the time of sampling. For example, temporal gradients in nitrogen species are shown in Figure 3. Algae, macrophytes, and the biota that feed on them grow most rapidly in the spring and summer months, and grow only slightly in fall and winter. Over the course of a day, plants consume oxygen at night and produce it during the day as they photosynthesize. Nutrient uptake is greatest during photosynthesis. As with spatial averaging described previously, specification of the time of sampling should be a part of the numerical nutrient criteria. Some examples of the inclusion of temporal averaging in the criteria are:

- Only daytime values will be used (say between the hours of 10 am and 3 pm).
- Only the growing season will be considered for compliance with criteria, or values from the entire year will be used.
- Values will be considered only when certain minimum depths (lakes/reservoirs) or minimum flows (rivers) are exceeded.

This list must be supplemented by other factors that are responsible for causing temporal gradients in nutrient levels.

3.5 HOW WILL COMPLIANCE BE DETERMINED?

Once several measurements of nutrient levels in a water body have been made, it still remains to be determined what numerical level will be defined as an exceedance of the nutrient criterion. There are several approaches to consider in determining when a water body exceeds a criteria. (If a numeric value chosen for the criterion required a minimum value rather than a maximum value, this argument would be reversed.)

Compare the annual arithmetic mean or the geometric mean to the numeric criterion, and consider any mean value greater than the criterion to be an exceedance. Selection of the arithmetic mean versus the geometric mean should be based on the distribution of data. (For log-normally distributed data, a geometric mean is preferred; for normal distributions, an arithmetic mean should be used.) This test could be performed once each year or with a rolling 12-month mean every month. It is important to note that we are more likely to observe exceedances if we sample and compare with the standard more often. Therefore, to compare all water bodies in a similar way, we need to use approximately the same frequency for sampling and comparing against the standard. In the event that different water bodies of necessity are sampled at different frequencies (e.g., to control costs in a water monitoring program), we must apply corrections to the data that account for the greater incidence of false negatives.

Allow a certain percentage of exceedances (e.g., 5% or 10%) over the criterion, that would still have the water body in compliance. Thus, if 95% or 90% of the measured values were below the criterion, it would not be in violation. This is generally appropriate for nutrients (as compared with toxins) because nutrients do not normally have acute, irreversible effects at specific threshold concentrations. This qualification excludes particular chemical species that may be toxic at specific threshold levels (e.g., ammonia) which must be treated separately, and for which more rigorous exceedance criteria may be established.

Consider the allowance of seasonal or climatic factors in the standard. For example, in rivers with low flow, or during a season with low flow, a greater level of exceedances may be permissible.

Consider a tiered approach for the criterion. Because it is likely that all water bodies cannot be monitored at the same level of intensity, due to priority ranking of water bodies or due to finite monitoring resources, a tiered approach may be used to evaluate the exceedance of nutrient criteria. This suggests that water bodies be sampled at a certain frequency to begin with and, if there are indications that there may be nutrient-related impairment in the water body, the monitoring is intensified so that a better understanding of the problem may be obtained. For the purpose of this standard, an indication of nutrient-related impairment could be one or more of the following: occasional values of nutrients in excess of the numeric criteria, although on average the water body is within compliance and observation of nutrient-related secondary impacts, such as low dissolved oxygen, or negative effects on fish and other biota, even though numeric concentrations of nutrients are within acceptable levels. Other indications of potential over-enrichment might be added to this list that would put a water body under a higher level of monitoring and study. Following this higher level of monitoring, it could be determined if the water body was genuinely out of compliance (using the tests described above).

Classify the water body before applying the criterion. The type of water body, and the chemical and physical characteristics of its surroundings (e.g., slope, climate, water body size and watershed area, geology, land use) has a fundamental effect on the levels of nutrients that may be considered natural and that may cause observable negative impacts. For this reason, it is vital to develop standards by classifying water bodies into certain major categories and calculate different nutrient criteria for different classifications. In previous studies, this classification has been performed at the ecoregion level for California; however, we need further classification of the physical characteristics of waterbodies. A major limitation of developing a finer classification is the paucity of data beyond a certain level of resolution. In the event that the RTAG has an influence on the future level of monitoring activities to be performed across California, an effort should be made to ensure that data are obtained from representative waterbodies along the principal classification criteria. These classification criteria should be communicated to the entity responsible for monitoring.

4.0 WATERBODY CLASSIFCATION CATEGORIES

The term classification suggests that sets of characteristics and observations can be organized into meaningful groups based on measures of similarity or difference. Experience suggests that each stream type possesses a set of inherent and presumably predictable attributes (e.g., channel pattern, dimensions and profile, biogeochemical signature, resistance and response to change, biotic productivity) which reflect the expressions of local climate, geology, landforms, and disturbance regimes. Basin characteristics (e.g., size, climate, geology) help define flow (water and sediment) characteristics which in turn help to shape channel characteristics within some broadly predictable ranges (Rosgen 1996, Orsborn 1990).

Understanding these inherent relationships is the key to identifying the appropriate factors for the assessment of the status and trends of aquatic systems, including the communities of organisms they support. Understanding how various geologic and climatic processes interact within a watershed gives a more thorough picture of the natural conditions (actual and potential) as well as of the direction and magnitude of possible changes triggered by natural or human disturbances.

4.1 OVERARCHING CLASSIFICATION CATEGORIES FOR ALL WATERBODY TYPES

4.1.1 Ecoregions

Ecologists and geographers, using general patterns in climate, soils, and vegetation, have classified terrestrial ecosystems into ecoregions (Bailey et al. 1994). Ecoregions are large-scale landscape units that include relatively homogeneous ecosystems and are distinguishable from other ecoregions (Omernik and Bailey 1997). Nonetheless ecoregions contain a mosaic of sites, which, at a finer scales, can be further subdivided into more detailed land units and effectively delineated by using mapable characteristics such as climate, geology, soils, and vegetation. The process of delineating such ecological units -- termed ecoregion analysis--and relating them within a hierarchical framework is increasingly viewed as a crucial step toward ecosystem management. Ecoregions are useful to river classifications as descriptors of landscapes within and among river basins (Omernik 1987, Omernik 1995, Omernik and Gallant 1990). Because the catchments of many medium-sized or larger rivers will span more than one ecoregion, ecoregion and watershed boundaries differ. Thus, neither provides a singular truth (Omernik and Bailey 1997) and both may be useful for management. Ecoregions group environmental resources and ecosystems into fairly homogeneous spatial units, while watersheds define contributions to the quantity and quality of the water at a particular point. Within a smaller region, the specific protocols used by U.S. management agencies (e.g., Meador et al. 1993, Rankin 1995, Barbour et al. 1998) commonly rely upon the segment-reach-habitat hierarchy described in Frissell et al. (1986).

Within ecoregions, ranges of expected values for habitat quality indicators can be developed empirically from data representing reference conditions. Reference conditions should provide us with a better understanding of the range of concentrations exhibited by both causal and response variables in minimally impacted waterbodies and, by inference, reflect the potential stream habitat for impacted streams having similar watershed characteristics.

This approach, however, has certain limitations. First, there is little agreement currently on what constitutes reference areas to cover Level III ecoregions (Bauer and Ralph 1999). Identification and use of reference areas is an ongoing effort at the state and regional level. In California, the Surface Water Ambient Monitoring Program (SWAMP) and California Bioassessment Workgroup (CABW) are currently working on identifying reference conditions for the state's waterbodies. Second, the currently available databases are generally not robust enough to provide statistically reliable values. This was observed in a demonstration pilot study conducted by Tetra Tech (2000) on Ecoregion II rivers and streams. Third, there are some ecoregions or regional areas, such as grass/shrub lands, where land management has been so pervasive that it has eliminated the potential for reference conditions (e.g., California's Central Valley). Regardless of these current limitations, it remains useful to outline an approach and then search for appropriate datasets or encourage the collection of appropriate data. In the interim, we may need to rely on the published data sets available, using them with appropriate caution.

4.1.2 Beneficial Uses

Since the ultimate objective of the Nutrient Criteria Development Program is to establish and promulgate numeric water quality standards for nutrients, it would be remiss to not include Beneficial Uses as one of the stratification criteria. 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 (Table ? in Appendix X). This table provides a starting point in understanding the relationships between causal and response variables and beneficial uses. The table lists the concentration of nitrogen, phosphorus, chlorophyll a, or turbidity/secchi depth that, when exceeded, caused some impairment of the designated beneficial use for a specific waterbody. The authors of the table note that these values must be used with discretion since certain details presented in the original study report were not included in the summary table. However, it does provide us with a linkage between nutrient enrichment and potential beneficial 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. Water used to support agricultural supply would be expected to be characterized by elevated concentrations of nutrients as a result of fertilizer application to agricultural lands.

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 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.

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. 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.

Freshwater Replenishment

Uses of water for natural or artificial maintenance of surface water quantity or quality. 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.

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, 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.

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 could stimulate primary productivity and result 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 could stimulate primary productivity and result clogged intake pipes.

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 clogged intake pipes. Additionally, elevated concentrations of nitrate (>10 mg/l) are toxic to human infants.

Navigation

Uses of water for shipping, travel, or other transportation by private, military, or commercial vessels. Excessive primary productivity could result in nuisance periphyton 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 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 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.

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, 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, impact aquatic life, impact anglers, and have negative aesthetic value.

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, 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, impact aquatic life, and have negative aesthetic value.

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.

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 aquatic life. Excessive primary productivity, however, could result in depletion of oxygen supplies and impact aquatic life.

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, 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.

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, 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.

Ecoregions provide a first tier of organization and are stratified on the basis of ultimate factors: climate, geology, and vegetation. At the ecoregion scale, the ranges of expected values for habitat quality indicators can be developed empirically from data representing reference conditions. Beneficial Use classifications provide a level of context in which criteria will ultimately be used.

The following sections describe classification categories that are specific to either rivers and streams or lakes and reservoirs. The underlying assumption of the classification categories discussed is that, in some unique way, waterbodies having characteristics specific to a category respond to nutrient loading in similar and predictable ways. One of the objectives of the RTAG/STRTAG Workshop is to focus our attention on those classification categories that will provide the most useful predictive information.

4.2 Classification Categories Rivers and Streams

Direct and indirect effects of riparian and stream channel modifications on lotic ecosystems have been documented (Karr and Schlosser 1977, Karr, et al. 1983, Rankin 1995). However, the deleterious effects on aquatic life from polluted runoff, especially from the primary nutrients (nitrogen and phosphorus), and the interaction with habitat quality, is neither widely acknowledged nor generally understood by resource management and regulatory agencies (Rankin, et al. 1999). Only recently has the issue been addressed of how land use, physiographic relief, soil types, and lotic habitat interact to affect instream nutrient concentrations and, in turn, the quality of aquatic assemblages (Richards et al. 1996, Allan et al. 1997, Johnson et al. 1997 as cited in Rankin et al. 1999).

Flow weighted sampling of chemical constituents is required to accurately estimate total loadings of nutrients for the calculation of TMDLs. Large runoff events, which deliver a high proportion of the annual loading of nutrients in a short time period (Baker 1985), are known to affect water quality in downstream environments. However, direct evidence of negative, local effects of elevated concentrations of nutrients during these short-term events on resident aquatic assemblages is lacking (Rankin et al. 1999). Given the low acute toxicity of elevated nutrients during such short-term events, it is the residual effects like elemental flood subsidies (Meyer et al. 1988) of nutrient loadings that are likely of most consequence to aquatic community performance. The cumulative effects of these events on trophic and energy dynamics of lotic systems may be long lasting (Rankin et al. 1999).

Rankin et al. (1999) state that the retention of nutrients in a stream reach and nutrient fluxes are important in determining how nutrients affect aquatic assemblages. Lotic reaches that either export or assimilate nutrients into desired biomass quickly (e.g., streams with high quality habitat and high gradient) may be less impacted by short-term loadings of nutrients. The following is a description of commonly used stream classification criteria and how they relate to nutrient loading.

4.2.1 Land Use Characteristics

Land use characteristics convey the dominant character of a geographic area, such as urban, agricultural, forest, rangeland, water (reservoirs), barren land, tundra, wetland, and perennial snow or ice. These characteristics greatly affect stream condition. Additionally, the assimilation and removal of nutrients by an intact and healthy riparian buffer is significant (Fennesy and Cronk 1997, Lowrance et al. 1984, Peterjohn and Correll 1984). Other studies have demonstrated that tillage practices in an agriculturally dominated watershed can have substantial effects on the rate of nutrient delivery to streams (Chichester and Richardson 1992).

Maps and other geographic data describing and documenting land use are available from a variety of sources.

4.2.2 Geology

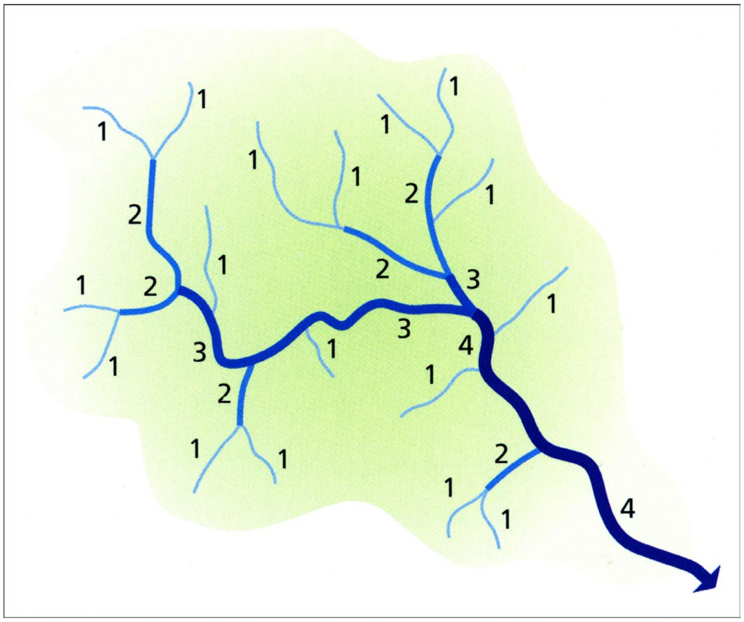
Geology provides the foundation for stream type classifications. Geology influences land use, stream order, entrenchment, sinuosity, gradient, and other stream characteristics. Bedrock composition has been related to algal biomass in some systems (Biggs 1995). For example, streams draining watersheds with phosporus-rich rocks (such as from sedimentary or volcanic origin) may be naturally enriched and, therefore, control of algal biomass by nutrient reduction in such systems may be difficult. In addition, nutrient content, and hence algal biomass, often naturally increases as elevation decreases, especially in mountainous areas (Welch et al. 1998). Some areas that are naturally-rich in phosphorus include watersheds draining volcanic soils and other areas have high weathering of nitrate from bedrock (Halloway et al. 1998).

Geologic data and information is readily available in the form of maps and digital data. The U.S. Geological Survey's (USGS) Mineral Resources Program is a useful source of information.

4.2.3 Stream Order

Stream order is a measure of the relative size of streams (Figure 4-1). The available scientific information about nutrient spiraling in lotic ecosystems indicates that headwater streams strongly influence the elemental dynamics of higher order streams and rivers within a watershed through the cumulative cascading of near-field effects in a downstream direction (Rankin, et al., 1999). The network of headwater streams within a watershed are critical to the functioning and quality of services provided by the larger order streams and mainstream rivers. That 75% of all streams in the U.S. are either first or second order (Leopold, et al., 1964) underscores the importance of the land water interface and the function of headwaters in maintaining watershed integrity (Rankin, et al., 1999).

Figure 4-1. Stream Order in a Drainage Network



Generally, the smallest streams in a watershed that have year-round water and no tributaries are first order streams. When two first order streams come together, they form a second order stream and when two second order streams come together, they form a third order stream. This process then continues on down the drainage network until the large river meets an ocean. Streams gradually increase their width and depth as they change from first to second order, and so on. Water discharge also increases as stream order increases.

There are different methods for determining stream order. According to Strahler (1952), stream order increases when streams of the same order intersect. Thus, the intersection of a first order stream and a second order stream will result in a second order stream identification for the newly joined reach. The second order stream used in this example will remain a second order stream until another second order stream intersects it, at which time it will become a third order stream.

4.2.4 Stream/River Size

Large rivers are more autotrophic than small rivers, with an increasing fraction of their organic carbon being fixed by primary producers within the stream channel, and with increasing stream order (Rankin et al. 1999). In these waters, nutrient turnover is rapid, resulting in higher concentrations of readily available forms of nutrients. In headwater streams that have been either channelized, had riparian vegetation removed, or the habitat otherwise degraded, the nutrient processing mimics that of large rivers: comparatively high nutrient turnover rates and high algal biomass are found.

Rankin, et al. (1999) found that watershed drainage area played a significant role in the assimilation of nutrient in lotic systems. They differentiated rivers and streams into four different size classes:

Headwater streams	0 – 20 $miles^2$
Wadeable streams	20 – 200 $miles^2$
Small rivers	200 – 1,000 $miles^2$
Large rivers	>1,000 $miles^2$

4.2.5 Nutrient-Sensitive Downstream Waterbodies

Rivers and streams act as water conduits, moving water as well as sediments and nutrients from one area of the state to another; oftentimes, draining into a lake, reservoir, or estuary. This poses a potential nutrient loading challenge for the receiving waterbody since increased nutrient loading can lead to a degradation of the waterbody's beneficial uses. Therefore, any criteria established for a river or stream must be mindful of the ultimate destination of the water and nutrients and protective of the downstream beneficial uses.

4.2.6 Flow Regime

Flows have important effects on stream channel morphology and bed material particle size. Specifically, since higher flows move larger particles, peak flows determine the stable particle size in the bed material (Grant 1987). Large, stable particles provide important habitat niches for invertebrates and fish. A highly unstable bed will reduce periphyton and invertebrate production (Hynes 1970, Welch et al. 1988). The size of peak flows is also important in determining the stability of large woody debris, the rate of bank erosion, and sediment loads. Increased bank erosion and channel migration will affect the riparian vegetation.

The concentration of nutrients (as a logarithmic function) in lotic systems increases significantly with increased flow (Edwards 1973, Brooker and Johnson 1984). However, a precise predictive relationship does not exist because similar concentrations can occur at different flows (Lowrance and Leonard 1988). For example, a two-inch rainfall immediately following fertilizer application will likely result in different instream nutrient concentrations than the same amount of rain at the end of the growing season.

Flow data are available from the United States NWIS-W data retrieval website at http://water.usgs.gov/nwis/. Stream flow data can be characterized using the categories presented in Smith et al. (1997). These authors classify streams into three different flow regimes (low, mid, and high):

Low flow	<28.3 m3/sec, or 1,000 cfs

Mid-sized flow	28.3 – 283 m3/sec
High flow	>283 m3/sec, or 10,000 cfs

4.2.7 Downstream Loading

Large runoff events, which deliver a high proportion of the annual loading of nutrients in a short time period (Baker 1985), are known to affect water quality in downstream (far-field), estuarine, or lentic environments. However, direct evidence of negative, near-field effects of elevated concentrations of nutrients during these short-term events on resident aquatic assemblages is lacking (Rankin, et al. 1999). Given the low acute toxicity of elevated nutrients during such short-term events, it is the residual effects of nutrient loadings that are likely of most consequence to aquatic community performance. The cumulative effects of these events on trophic and energy dynamics of lotic ecosystems may be long lasting.

4.2.8 Stream Gradient

Stream gradient, or slope, is the change in water surface elevation over a given distance as expressed as a percentage (valley gradient/sinuosity = channel gradient). Slope can often be estimated from sinuosity derived from aerial photos and topographical maps. Gradient is directly related to both bed-material load and grain size, and is inversely related to discharge (Schumm 1977).

Gradient classes are useful in grouping streams with similar response to flow and sediment inputs. Rosgen (1994) provides the following gradient classes:

Gentle gradient	< 2%
Riffle dominated	2 – 4%
Steep	4 – 10%
Very Steep	>10%

Very steep slopes are characterized as having frequently-spaced vertical drops and pools as bed features, with high debris transport. Steep slopes are characterized as having steep, cascading steps and pools as bed features. Riffle dominated streams and rivers have characteristic rapids and infrequently-spaced scour pools at bends or areas of constriction. Those rivers classified as having a gentle gradients have gentle slopes with riffles and pools as bed features.

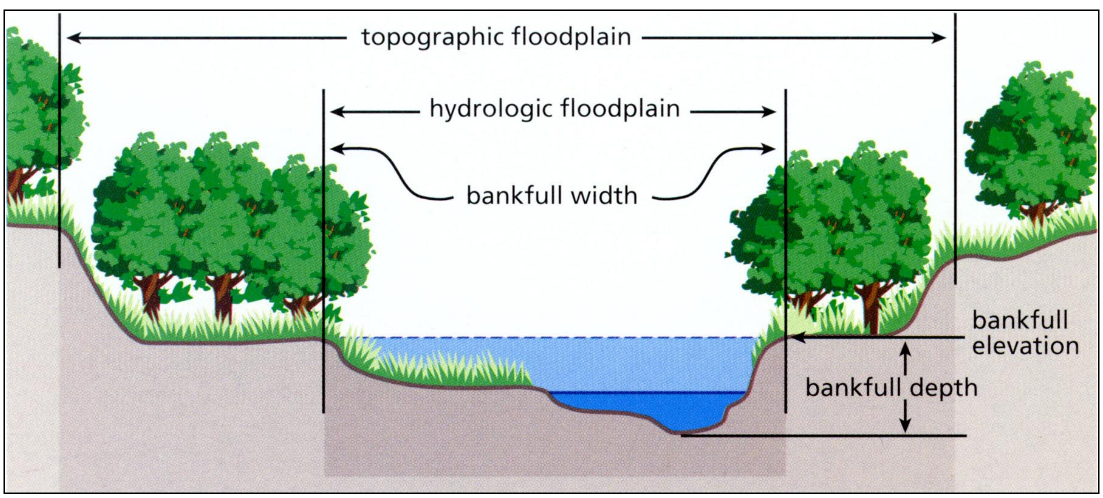
Leopold et al. (1964) and Morisawa (1968) have correlated stream gradient with mean particle size such that streams with higher gradients have larger average diameter substrate particles than streams with low gradients. Because phosphorus is delivered into streams attached to fine particles (particulate phosphorus), streams with a high bedload of fine sediment also have the highest concentrations of total phosphorus. Fresh sediments in high-gradient streams may create a buffer from high phosphorus concentrations by providing adsorption sites for phosphorus (Klotz 1988). By contrast, in low-gradient streams with high sediment retention such adsorption sites may be secured by existing phosphorus and the sediment will have little effect as a nutrient buffer.

The retention time for water and fine particles within the low flow channel of low gradient streams is longer than for higher gradient streams, resulting in an accumulation of total phosphorus. This provides more time for the available phosphorus to be utilized in potentially undesirable ways such as the production of excess algal biomass. The retention of nutrients in a stream reach and nutrient fluxes are important in determining how nutrients affect aquatic assemblages. Reaches that either export or assimilate nutrients into desired biomass quickly (e.g., streams with high quality habitat and high gradient) may be less impacted by short-term loadings of nutrients. Low gradient streams, because of longer nutrient and sediment retention times, are more susceptible to the effects of nutrients than otherwise similar high gradient streams.

4.2.9 Width-to-Depth Ratio

The width-to-depth ratio describes a dimension of bankful channel to bankful mean depth. Bankful discharge is defined as the momentary maximum peak flow, which occurs several days per year and is related to channel-forming flow (Figure 4-2). Stream characteristics and sediment transport relations rely heavily on the frequency and magnitude of bankful discharge. Bankful width, along with the associated discharge regime, serves as a consistent morphological index that relates to channel formation and maintenance.

Figure 4-2. Typical Channel Cross Section Illustrating Bankfull Width and Depth Concepts



Bankful width is used as a surrogate for bankful discharge. Bankful discharge is the flow volume which transports the largest portion of the annual sediment load, including bedload, over a period of years (Wolman and Miller 1960). It is that flow which mobilizes the majority of the bed material and develops and maintains the form of the channel (Olsen et al. 1997). It is a critical discharge, as channel-forming forces do not increase proportionately at flows greater than bankful due to over-bank dissipation of energy. Bankful flows generally correspond to the 1.5 to 2 year recurrence flow event (Bray 1982). Measurement of bankful width is a repeatable variable but is often difficult to identify in non-entrenched channels. As mentioned previously, sediment transport plays a significant role in nutrient load to downstream waterbodies. Rosgen (1994) classifies low ratios as those less than 12. Values greater than 12 are classified as moderate or high.

4.2.10 Entrenchment Ratio (Stream Confinement)

An important factor of classification is the relationship of the stream to its valley. This relationship of the width of the flood prone area to the width of the channel tells the degree to which the stream is incised or entrenched (vertically contained) into the valley floor. Relative entrenchment makes the distinction of whether the flat area adjacent to the channel is a frequent floodplain, a terrace (an abandoned floodplain), or is outside of a flood prone area. The flood-prone area is generally flooded in a 50-year storm event.

Rosgen (1994) presents the following ranges for entrenchment ratios:

Entrenched streams	1 – 1.4
Moderately entrenched streams	1.41 – 2.2
Slightly entrenched (well-developed floodplains)	>2.2

The continuum concept (for field measurements) allows for variation of ±0.2 units if other criteria still match the range of variables for that type of stream (Rosgen 1994).

4.2.11 Sinuosity

Sinuosity is the ratio of stream length to valley length. It can also be described as the ratio of valley slope to channel slope. Mapping sinuosity from aerial photos is often possible, and interpretations can often be made of slope, channel materials, and entrenchment once sinuosity is determined. Values of sinuosity appear to be modified by bedrock control, roads, channel confinement, specific vegetative types, and other factors. In general, as gradient and particle size decreases, there is an increase in sinuosity. The continuum concept allows for variation of ±0.2 units if other criteria still match the range of variables for that type of stream (Rosgen 1994).

The geometry (or pattern) of meanders in a stream are directly related to sinuosity. Curves are formed in the stream in reaction to physical forces that seek an equilibrium of smallest energy expense.

4.2.12 Channel Materials

The bed and bank materials of the river are critical to sediment transport and hydraulic influences; these materials also modify the form, plan, and profile of a river. Interpretations of biological function and stability also require this information. A good working knowledge of the soils associated with various landforms can often predict the channel materials at the Level I delineation level. Reliable estimates of the soil characteristics for glacial till, glacial outwash, alluvial fans, river terraces, lake and wind deposits, and residual soils can usually be derived from existing soils and geology maps. Rosgen (1994) reports the dominant particle size as the median size, or the size that is equal to or larger than 50% of the total sample.

4.3 SUMMARY FOR RIVERS AND STREAMS

Ecoregions and stream classification systems provide a framework for organizing habitat components, habitat variables, and narrative, as well as numerical, indicators. Level III ecoregions may provide a sufficient first iteration for categorizing watersheds in order to evaluate potential reference conditions for many habitat variables. Further subdivision of ecoregion organization may be useful in providing a more homogeneous organization of watersheds but may also be a daunting task given the limited amount of data on reference condition. Using beneficial uses as a subclassification scheme will reduce the need for true reference conditions by providing a context against which we can assess the status of a stream reach as it is not meeting, meeting, or exceeding its beneficial uses. A meaningful organization of stream networks ultimately depends on the identification of geomorphically-similar stream reaches that respond to nutrient loads in a similar fashion. Classification systems that incorporate these factors should be useful in developing a spatial framework for habitat indicators.

4.4 LAKES AND RESERVOIRS

Unlike rivers and streams that can span large geographic regions, most lakes and reservoirs are discrete water bodies that are typically confined to smaller regions. Most lakes and reservoirs are relatively small and they are typically components of much larger river systems. Because rivers and streams can cover large areas with different climates, landforms, geologies, and flow regimes, their characteristics can change drastically along their courses from their headwaters to their downstream boundaries. Therefore, many river classification schemes tend to focus on subdivisions of large spatial scales, and the hydraulic and geomorphological characteristics of specific channel sections within a larger river network. Since lakes are often fed by rivers, the broader classification schemes that refer to watersheds and river basins are also often applicable to lakes.

Because lakes are often viewed as isolated water bodies, most traditional lake classification schemes have focused on groups of physical, chemical, or biological characteristics within lakes that make them similar in terms of biogeochemical cycling processes. The most important of these is the eutrophication classification, which represents the overall nutrient status of the lakes. Lakes are classified as oligotrophic, mesotrophic, eutrophic, and hypereutrophic, depending on whether they have low, moderate, high, or extremely high levels of nutrient enrichment, respectively. These classifications are broad and embody a wide array of lake attributes including nutrients, phytoplankton (densities and productivity), turbidity, dissolved oxygen, and many other water quality parameters and biological factors that influence all components of aquatic ecosystems. The nutrient status influences the species composition, abundance, and productivity of all trophic levels from bacteria through fish. Eutrophication classifications are also used for all other types of water bodies, but the specific nutrient limits and biological responses within a category can be somewhat different than in lakes.

Lakes can also be classified in terms of many other water quality variables (depending on the objectives of the analysis) For example, aerobic versus anaerobic, stratified versus fully mixed, hard water versus soft water, acid versus alkaline, etc.,. Each of these simple classifications implies numerous other biogeochemical and biological relationships that are associated with those types of lakes. Differences in the vertical distributions of certain critical water quality variables can also be used to categorize lakes. An example is the orthograde, clinograde, positive heterograde, or negative heterograde oxygen profiles of lakes with different levels of productivity.

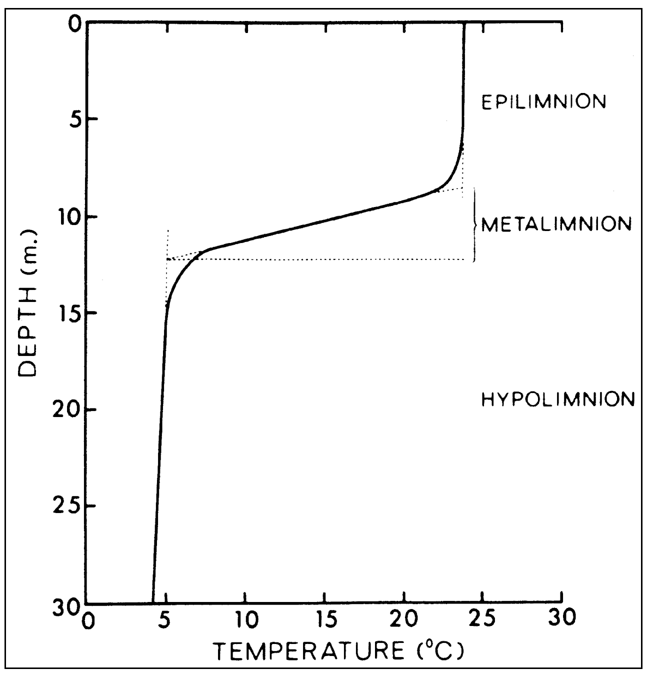
Other lake classification schemes focus on different spatial zones within lakes. Horizontally, lakes can be divided into the littoral zone and the pelagic zone. The littoral zone is the nearshore area adjacent to the lake shoreline that is shallow enough to support macrophytic vegetation, and the pelagic zone is the remaining offshore open water portion of the lake. The boundaries of the littoral zone depend on how far light penetrates the water column, since rooted vegetation can exist only where light reaches the sediments. The extent of the littoral zone therefore depends on both the bottom slope of the shoreline and turbidity. The littoral zone can be further subdivided into the upper, middle, and lower littoral zones, depending on the distribution of the three major types of rooted macrophytes that occupy each zone along a depth gradient. The upper littoral is dominated by emergent macrophytes, the middle littoral by floating-leaved macrophytes, and the lower littoral by submersed macrophytes (Wetzel 1983).

The biological communities and biogeochemical cycling processes can vary greatly between the littoral and pelagic zones. Primary production in the littoral zone is driven mostly by macrophytes, periphyton, and epiphytic algae, while primary production in the pelagic zone is driven by phytoplankton. The littoral zone is much more heterogeneous than the pelagic zone due to variations in the vegetation and bottom substrate. It is also typically oxygenated and fairly well lighted, but has many sources of cover. The littoral zone therefore has a different food web structure, as well as different habitat characteristics, than the pelagic zone; as a result, their biological communities, organism associations, and productivities are often very different.

The pelagic zone refers primarily to a region of the lake water column, while the littoral zone refers to both the nearshore waters and sediments. A parallel category that refers to the offshore sediments below the littoral zone is the profundal zone. The profundal zone represents the sediments and benthic environment in the pelagic portion of the lake.

Many other spatial classification schemes for lakes focus on vertical differences in lake processes. Seasonal temperature stratification is the most critical element, since it influences almost all other processes in lakes. Stratification isolates the surface waters from the deep waters during the periods of peak production. Atmospheric exchanges and photosynthetic activities are restricted to the upper waters, while sediment interactions and many decomposition processes occur primarily in the deeper waters. This results in major differences in the vertical distribution of water quality and biological processes. Spatial classification based on vertical temperature profiles separates the water column into three zones: the epilimnion, metalimnion, and hypolimnion. The upper mixed layer is the epilimnion, the thermocline region is the metalimnion, and deeper area below the thermocline is the hypolimnion (Figure 4-3). The metalimnion is often lumped together with the hypolimnion or epilimnion, since it is really a transition zone between these two major categories.

Figure 4-3. Typical Thermal Stratification of a Lake into the Epilimnetic, Metalimnetic, and Hypolimnetic Water Strata



Other vertical spatial classification schemes for lakes focus on light availability, for example the photic zone and the aphotic zone. The photic zone is the upper portion of the water column where net photosynthesis occurs, and the aphotic zone is the remaining lower portion of the water column where light is too low to support photosynthesis.

Lakes can also be classified in many other ways that distinguish their geographic location, geologic origin, size and shape, hydrologic characteristics, watershed features, biological communities, and primary uses. Many of these factors can influence the nutrient dynamics and potential productivity of lakes. Therefore, some of these classification schemes should be useful for grouping lakes into categories with similar background nutrient concentrations, levels of primary production, and responses to external nutrient loads.

Several classification categories must be explored to determine which variables are most important to distinguishing the nutrient responses of lakes. The potential lake classification factors can be grouped into the following major categories:

- location
- lake type
- size and shape
- flow
- watershed characteristics
- water quality
- stratification
- lake origin
- age
- dam operation (for reservoirs)
- fish community
- other biological characteristics

Each of these categories includes numerous individual classification factors, many of which are potentially useful for separating lakes into groups with different nutrient responses. These factors are discussed in the following sections.

Because there are many types of classification categories, each with many potential classification variables, it will be necessary to determine which key variables are the most critical for separating the nutrient responses of lakes (since it is only practical to include a few parameters in any classification hierarchy). Several of these categories interact or overlap in some ways. Therefore, it will be important to determine which parameters are the most independent and exert the most influence. However, parameters that integrate the influence of several factors may also be useful. Some categories have easily available information, while others will be difficult to obtain. Therefore, data availability and the cost and effort required to obtain the data will also be an important consideration. The final classification scheme should include only a few key variables that can be easily obtained for most lakes.

4.4.1 Location

Several location-related factors can influence the nutrient status of a lake. These include the ecoregion, the elevation, and the position in the drainage basin or along a river network.

Ecoregion differences incorporate many important factors including climate, geology, soil types, vegetation types, elevations, topography, and land use. Together these factors influence both the hydrology and external nutrient loading to the lake, as well as the temperature pattern and solar radiation, that drive primary production.

Although elevations are incorporated into the ecoregion categories, it may be appropriate to consider elevation as a separate factor in the more mountainous regions; high altitude lakes, for example, behave differently than lower altitude lakes. High elevation lakes in granitic basins are expected to have relatively low nutrient loads from the watershed (as well as lower temperatures) resulting in low productivity. Lower elevation lakes may have better developed soils, warmer climates, and larger watersheds, all of which may contribute to greater nutrient loading and higher primary productivity. However, residence times may be shorter due to higher flow rates from the larger watershed, and higher levels of turbidity may occur from the increased river influence, possibly inhibiting photosynthesis.

The position of a lake within a drainage basin or along a river network can also influence its nutrient status. Lakes situated high in the drainage basin, or near the headwaters of streams, are often at higher elevations and have lower levels of nutrient loadings and primary productivity (for the same reasons as higher elevation lakes). In contrast, lakes at the lower ends of large drainage basins, or along the lower reaches of major river networks, receive the cumulative nutrient loads from all of the upstream areas in the watersheds, and therefore may be more nutrient enriched. In addition, they are typically at lower elevations and are subject to warmer climates and greater primary productivity. Furthermore, lower elevations often have more nutrient rich soils than higher elevations. Heavily forested upland areas typically have lower background nutrient concentrations in runoff and base flows than grass or scrub areas in the lowlands.

Most lakes behave as nutrient sinks; a portion of the nutrient loads are taken up by algae that settle and deposit nutrients in the sediments. Therefore, a significant portion of the upstream nutrient loads from the watershed may be deposited in upstream lakes, particularly if they have long residence times. Surface releases from upstream lakes often have much lower nutrient levels than ambient nutrient levels in the streams feeding the lakes because of nutrient depletion by phytoplankton uptake. This results in reduced nutrient loads to the lakes directly downstream, particularly during periods when upstream lake releases make up a major portion of the incoming stream flow.

Information on lake location with respect to ecoregion, elevation, or position along a drainage basin or river network can be easily obtained from geographic data or USGS topographic maps. Categorization schemes will have to be developed to represent position along a stream or drainage basin. This could include factors such as stream order, distance from headwaters, upland areas versus mainstem river, and distance from upstream lake.

4.4.2 Lake Type

Lakes can be classified as open or closed systems, depending on whether the lake has a major outflow. Lakes with no apparent outlet may still lose water through seepage to groundwater, which results in a lake response similar to outflows to a stream. If the lake basin is located in impervious material, there may be no outlet. In these situations, the major source of water removal is evaporation.

In closed systems or seepage lakes with low seepage rates, there is limited opportunity for nutrients entering the lake to exit the system through normal transport processes. The only loss processes are sedimentation and burial in the deep sediments. Evaporation results in the concentration of nutrients and other constituents. In regions where evaporation losses are significant, nutrient levels may become elevated. Internal nutrient loads from sediment release and algal or macrophyte decomposition can produce higher nutrient levels in the water column than in open lakes (because none of these loads can be washed out of the lake through outflows). Since there is no flushing, phytoplankton may reach higher levels than in lakes with significant outflows.

In contrast, open systems can lose a significant portion of their nutrient loads through outflows. If the outflow rate is high, relative to the volume of the lake, flushing can offset phytoplankton growth and limit the maximum algal levels that can be attained. Open systems with short residence times respond rapidly to changes in nutrient loading regimes and inflow nutrient concentrations; there is less time for sedimentation processes to remove nutrients from the water column.

The determination of whether a lake is open or closed can often be determined from using maps or other sources of geographic data. In other cases, it can be obtained from interviews or site visits. However, the determination of the rates or significance of seepage losses requires water balance studies, and are typically not available for most lakes.

4.4.3 Size and Shape

Size is an important attribute that can influence lake response to nutrient enrichment. Smaller lakes are typically shallower and have smaller surface areas that are more heavily influenced by littoral processes and shoreline activities.

Shallow depths result in higher light intensities over a greater portion of the water column, increasing photosynthesis and lake productivity. Light penetration to the lake bottom also promotes macrophyte and periphyton growth. Mean depth is inversely correlated with productivity at all levels in large lakes (Wetzel, 1983). The deeper the lake, the greater the portion of the water column that cannot support photosynthesis. Although this correlation breaks down in shallow lakes, shallow lakes are still generally more productive than deeper lakes. Shallow lakes have more light, greater interaction with the sediments, and more littoral areas. Biological productivity is typically highest in areas where the photosynthetic zone overlaps the decomposition zone (Wetzel, 1983), as nutrients released during decomposition are immediately available for photosynthesis. In deep lakes, stratification isolates the lower decomposition zone from the upper photosynthetic zone during most of the peak growing season.

Shallow depths increase the water column response to sediment-nutrient releases or plant decomposition releases because less water is available to dilute these internal load sources. In addition, the shallower depths and increased productivity result in greater settling losses and allow more organic detrital material to accumulate in the sediments. This increases the sediment nutrient contents and sediment release rates.

Shallow depths may also prevent stratification and the development of an anaerobic hypolimnion. Maintenance of aerobic conditions at the sediment-water interface reduces phosphorus and ammonia release rates from the sediments. In very productive shallow lakes, the high levels of productivity may produce diurnal anoxia from plant respiration, eliminating the oxidized microzone at the sediment surface and increasing nutrient release rates during the night. The high oxygen demand associated with the decomposition of organic matter on the sediment surface may also prevent the development of an oxidized microzone at the sediment-water interface, even though most of the water column above the sediments is oxygenated. Deep productive lakes generally develop an anaerobic hypolimnion during the summer months. Although this maximizes nutrient release rates from the sediments, stratification prevents them from reaching the productive surface waters.

Since macrophytes compete with phytoplankton for light and nutrients, shallow macrophyte-infested lakes may have low phytoplankton densities, even though nutrients are abundant. Phytoplankton blooms often occur following macrophyte removal, since they are no longer light-limited from macrophyte shading.

Since the littoral zone is often a major portion of the overall area of small lakes, littoral production and decomposition processes play critical roles in the nutrient dynamics of small lakes. Attached littoral communities contribute more to the overall productivity of small or shallow lakes than to larger or deeper lakes. Large or deep lakes are dominated more by pelagic production. Littoral contributions to production and nutrient cycling are also higher in lakes with very irregular shorelines, since the shore areas are large relative to the surface area of the lake.

Mean depth and volume are the most useful parameters for characterizing the size of lakes. Mean depth is directly related to light penetration, photosynthesis, macrophyte and periphyton habitat, stratification, sediment release, and dissolved oxygen processes in lakes, so it plays a major role in nutrient dynamics and algal and plant response to nutrient enrichment. Therefore, mean depth is regarded as the best single index of lake morphometric conditions (Wetzel, 1983).

Lake volume determines the amount of water available to dilute nutrient loads. Together with inflow or outflow rates, it determines the lake residence time, which is important to nutrient accumulation and sedimentation processes, as well as phytoplankton flushing.

Lake surface area is also useful for characterizing the size of a lake, but it can be calculated from volume and mean depth. Other measures of size such as maximum depth, maximum length, maximum width, or shoreline length can also be used, but they are less useful than the other measures since they are not as closely related to lake processes.

The shape of a lake can also influence nutrient dynamics since lakes with very irregular or dendritic shorelines have much greater littoral zones than more regularly shaped lakes of similar surface area, depth, and volume. This attribute is most easily captured by the shoreline development ratio, which is the ratio of the actual lake perimeter to the perimeter of a circle with the same surface area of the lake.

Many measures of lake size are expected to be readily available. The surface area, maximum length, maximum width, and lake perimeter can be easily obtained for all lakes from maps and other forms of geographic data. Lake volume, surface area, and mean depth are interrelated, so if any two of these parameters are known, the other can be easily calculated. Mean depth and maximum depth have similar correlations in many lakes, so if one is known, the other can be estimated.

Lake volumes should be available for all reservoirs and for other major lakes where bathymetric surveys have been conducted. If the volume is unknown, it can be estimated by first estimating the mean depth from measurements of maximum depth, and then calculating volume by multiplying mean depth by surface area. If the volume and surface area are known, the maximum depth can be estimated by calculating the mean depth and using the above ratio to predict maximum depth.

4.4.4 Flow

The flow rate through a lake plays a major role in determining the nutrient and phytoplankton concentrations. The flow rates into the lake determine the nutrient loads from rivers and other tributaries. Inflows are usually the major external nutrient sources to lakes from the watershed. The outflow rates determine the amounts of nutrients and phytoplankton that are removed from the lake.

Flow rates can be characterized by inflow rate, outflow rate, and residence time, which is the lake volume divided by the inflow or outflow rate. Residence time quantifies the average time inflowing water remains in the lake. Residence time combines both flow and size characteristics, and is therefore more directly related to lake processes that control nutrient concentrations than inflow or outflow rates alone.

Outflows and nutrient sedimentation are the two major pathways for removing nutrients from the water column. Outflows permanently remove nutrients from the lake ecosystem, while sedimentation processes transport them to the sediments where a portion of them are later cycled back to the water column through sediment decomposition and release processes. Some of the nutrients that accumulate in the sediments are refractory compounds that are not regenerated and returned to the waters. Deposition also results in the continual burial of previously settled nutrients to the deeper inactive portion of the sediments that have minimal interaction with the water column. Deep sediment burial therefore effectively removes this portion of the nutrients from the lake ecosystem.

The lake residence time is important to sedimentation losses. Nutrient sedimentation results from uptake and settling by phytoplankton and by adsorption of phosphate and ammonia to settling detrital materials, especially clays. In some lakes, especially very hard water lakes, nutrients can be sedimented through coprecipitation processes. The longer the residence time of water in the lake, the more opportunity there is for settling and nutrient sedimentation to occur. Therefore, for a given loading rate, the greater the residence time, the lower the nutrient concentrations in the water column.

The shorter the residence time, the more quickly the lake reaches steady-state nutrient concentrations after changes in loads. The shortest residence times are generally associated with lakes and reservoirs on rivers or streams during high flow periods. If the stream inflow is the major nutrient source, the lake nutrient concentrations will approach the stream concentrations as the residence time decreases since there is no time for significant sedimentation losses. However, internal nutrient loads from the sediments may obscure this relationship.

The residence time reflects the flushing capacity of the lake. Since flushing removes phytoplankton, the shorter the residence time, the less opportunity there is for phytoplankton blooms. This is important primarily for relatively short residence times that are similar to the time scale of algal growth processes. In these situations, the shorter the residence time, the lower the resulting phytoplankton concentrations (assuming the same nutrient concentrations in the water column). At longer residence times, grazing and settling losses, as well as growth limitation by nutrients or light, may be more important than outflow and flushing losses. Since residence time influences both algae and nutrients in different ways, and since algae and nutrients directly influence each other, the net effect of variations in residence time on phytoplankton concentrations is complex and depends largely on the time scales involved.

The location of the outflows is also important. In a reservoir with a deep outlet below the depth of the thermocline, outflows remove nutrients from the hypolimnion, but have minimal effects on nutrients and phytoplankton in the epilimnion. Therefore, residence time or outflow rate considerations are not of such major importance during the summer stratification season, since nutrient and phytoplankton problems are associated with the epilimnion. However, outflow rates and residence times still influence the overall nutrient budgets, and can therefore influence the magnitude of the spring phytoplankton blooms at the beginning of the stratification season.

Changes in surface elevation or storage volume associated with outflows from reservoirs also influence nutrient dynamics. Lowering the surface elevation may drop the water level below the vegetated portion of the littoral zone, desiccating periphyton and macrophytes and reducing littoral influences on nutrient cycling. Lowering the water level may also drop the thermocline elevation below the level of a reservoir outlet, converting a hypolimnetic withdrawal to an epilimnetic withdrawal. Reductions in storage volumes associated with lower water levels reduce residence times, assuming the flow rates remain the same.

Flow rate and residence time information is available for many lakes, but not for all of them. It is most likely to be available from reservoirs, since stream flow or release monitoring may be conducted to manage the reservoir. If fluctuations in reservoir surface elevation are recorded, adjustments can be made to the flow estimates using water budget methods in conjunction with precipitation and evaporation information. Flow information may be available stream gauging stations located upstream or downstream of the lake. The USGS maintains a network of permanent stream gauges, but additional gauges are available in some areas from other agencies. If no stream gauge measurements are available, inflows can be estimated from precipitation data using the watershed area and typical runoff coefficients for the major land uses in the watershed. Known releases from upstream reservoirs should be included in these estimates.

Residence times are easily calculated from the volume and flow estimates. If volume and residence time information are available from other sources, then average flow rates can be calculated in the absence of direct flow measurements. In this situation, it must be assumed that the group responsible for the residence time estimate had sufficient flow information to accurately derive the number.

4.4.5 Watershed Characteristics

Watershed characteristics play a major role in determining the nutrient concentrations and productivity of lakes since, together with waste disposal activities in the watershed, they are the major determinants of external nutrient loads to the lake. Watershed characteristics, together with precipitation patterns, also control the lake hydrology.

The most fundamental watershed characteristic is the size of the drainage basin. In the absence of human nutrient sources, the greater the watershed area, the greater the nutrient loads and inflows for a given land cover and climate. The topography is also important since more erosion and runoff can be expected in steep terrain. The relative influence of external nutrient loads on the nutrient levels and productivity within the lake depend to some extent on the lake size and flushing rate. The flushing rate depends on both the size of the watershed and the size of the lake. Therefore, the ratio of the watershed area to the lake volume or surface area may be a more useful classification parameter that captures some of these interactions.

The land uses, vegetation types and coverage, and the soil characteristics and geology of the watershed also play key roles in determining runoff, nutrient loads, and soil erosion from the watershed. For example, dense forest vegetation is efficient at extracting both nutrients and water from the soils, and at minimizing the erosive effects of heavy rainfall. Therefore, both nutrient loads and runoff are typically small from forested areas relative to other types of land cover. In contrast, heavily urbanized watersheds have extensive impervious areas that maximize both runoff and nutrient accumulation, and since human activities further increase nutrient accumulation on the land, nutrient loads to urban lakes would be expected to be higher than to lakes in undeveloped forested areas. The geology and soil characteristics are also very important in determining the nutrient loads from watersheds. Some areas have geological formations and soils that are high in phosphorus or nitrogen, while others areas have formations or soils that are low in nutrients. High elevation lakes in rocky mountainous areas would be expected to have lower nutrient loads than more lowland lakes surrounded by fertile soils. Soil properties such as erodability, grain size, and cohesiveness influence erosion rates, and therefore the particulate nutrient loads to lakes.

4.4.6 Water Quality

In addition to nutrients, several water quality parameters influence the nutrient dynamics and productivity in lakes. Some have a direct impact on these processes, while others are indicative of important biogeochemical attributes of the watershed that are related to natural nutrient abundance. Key parameters include non-algal turbidity, color, pH, total dissolved solids, conductivity, alkalinity, and hardness.

Non-algal turbidity is associated with inorganic sediment particles and plant debris that enters the lake through stream inflows, or is resuspended from the bottom by wind and wave action. It is important to nutrient cycling because suspended inorganic particles such as clays can adsorb phosphate and ammonia, making them unavailable for phytoplankton uptake and transporting them from the water column to the sediments. More importantly, non-algal turbidity can control light transmission and the depth of the photic zone, causing light limitation of algal growth when turbidity is high. Therefore, turbid lakes may have low algal densities and low levels of the water quality impacts associated with eutrophication, even though the nutrient levels in the water are high. In addition to limiting photosynthesis, the turbidity influences stratification through its effects on the attenuation of solar radiation. Non-algal turbidity is largely dependent on the watershed soils, vegetation, geology, and hydrology.

Color indicates the amounts of dissolved organic matter in the water. Watershed inflows through organic soils or wetlands often carry dissolved humic substances into lakes, resulting in a yellow or brown stained water. Colloidal calcium carbonate often forms in hard-water lakes, producing a blue-green color. Colored water inhibits photosynthesis, both because it reduces light penetration, and because dissolved organic matter chelates nutrients, making them unavailable for algal uptake. Like turbid lakes, highly colored lakes often have very low productivity, even though nutrients are abundant. pH is also related to color since many of the organic compounds are humic acids.

Total dissolved solids (TDS) (or salinity) and conductivity are measures of the total amounts of dissolved chemicals in the water. Conductivity is a convenient field measurement for making this determination, and is less expensive than TDS analyses in the lab. TDS concentrations in lakes depend on both the levels of dissolved constituents introduced from the watershed, and on lake hydrologic processes such as precipitation and evaporation, which either dilute or concentrate the TDS concentrations. TDS is dominated by conservative constituents, so it is not as subject to the rapid changes that occur with biologically active constituents like nutrients. Therefore, in lakes that do not receive major contaminant loads from man, it is a good measure of the natural chemical loads from the watershed combined with the major lake processes that determine their fate in the lake. Since nutrients are natural components of the earth's crust, lakes with higher TDS levels would also be expected to have higher levels of nutrients.

Alkalinity is similar to TDS as an indicator of natural watershed nutrient inputs to lakes. Alkaline soils are typically rich in nutrients, especially phosphorus. Therefore, watersheds with alkaline soils will result in more alkaline lakes, which in turn would be expected to have higher phosphorus concentrations because of natural conditions in the watershed.

Hardness represents the calcium and magnesium concentrations in the water. Calcium and magnesium are two of the four major cationic components of TDS. Therefore, like TDS, lakes with high levels of calcium and magnesium are expected to have high loads of nutrients from the watershed. However, several important processes occur in hard-water calcareous lakes that alter the nutrient levels and productivity (Wetzel, 1983). High concentrations of calcium and carbonate result in calcium carbonate precipitation. Phosphate and essential micronutrients like iron and manganese also coprecipitate with the calcium carbonate. In addition, the settling particulates and colloids adsorb phosphate and certain micronutrients and dissolved organic compounds, effectively removing them from the water column. High pH values also reduce the availability of free carbon dioxide. The low phosphorus and micronutrient levels inhibit photosynthesis, resulting in low algal densities. Although phosphorus and algae are low, inorganic nitrogen levels are usually very high. This is because inorganic nitrogen is not coprecipitated or adsorbed, algal uptake is low, and nitrogen loads are typically high in calcareous regions. The low productivity generates less organic matter and maintains aerobic conditions at the sediment-water interface. The oxidized microzone inhibits phosphorus release from the sediments. The low levels of organic material restrict microbial activity and the release or synthesis of certain micronutrients. These factors further contribute to the low productivity and low phosphorus levels in hard-water lakes.

pH is correlated with alkalinity, hardness, and color. Alkaline and hard-water lakes are generally well buffered, with pH values above 8. Seepage lakes and lakes in igneous rock catchments tend to be less buffered and more acidic, with pH values lower than 7 (Wetzel, 1983). Lakes that are stained yellow or brown from humic acids are acidic and have low pH values, while hard-water lakes colored blue-green with colloidal calcium carbonate are alkaline and have high pH values. Although pH is related to the above parameters, which are in turn related to watershed characteristics correlated with nutrient concentrations in lakes, pH is also influenced by algal and macrophyte productivity. Photosynthesis removes carbon dioxide from the water, and can cause significant increases in pH during periods of peak production. Therefore, pH is a response variable, and may be less useful as a classification parameter than some of the other water quality variables above that are more independent of production levels in the lake.

Measurements of turbidity, color, pH, total dissolved solids, conductivity, alkalinity, and hardness may be available for well-studied lakes or lakes that have major monitoring programs. However, for many of the other lakes, only a few of these parameters are likely to be available. Turbidity, pH, and conductivity are easily measured in the field, so they are often available. Color can be estimated by observation, but particulate materials like phytoplankton and suspended clays can contribute to the color, and visual observations are somewhat subjective. Color can be determined more precisely using standard color scales and filtering the water to remove suspended material. Turbidity is most commonly measured as Secchi depth, which can be correlated to total suspended solids concentrations through empirical relationships. If algal densities are quantified separately, then the non-algal turbidity can be calculated by difference.

4.4.7 Stratification

Whether a lake becomes stratified or remains vertically mixed throughout the year plays a major role in both nutrient cycling processes and the potential primary productivity of lakes. Stratification occurs in deeper lakes. Most temperate lakes that are deeper than about 10 meters exhibit seasonal stratification (Wetzel, 1983). The thermocline depth varies throughout the stratified season, typically beginning with a shallow depth at the beginning of stratification in the spring, and increasing gradually throughout the summer and fall until mixing occurs. The stratification depth, the depth range of the thermocline, and the intensity of stratification depend on the complex interaction of several factors. These include meteorologic factors such as solar radiation, air temperature, wind speed, and humidity, which determine the rate of surface heating (or cooling); water clarity, which determines the depth of solar radiation penetration in the water column; wind speeds, directions, and frequencies, which determine the degrees of surface mixing, shear stresses and turbulence in the water column, and wind-driven circulation patterns; inflow rates and temperatures (densities), which determine volumes of water entering the lake, whether they flow into the epilimnion or hypolimnion, and the resulting heat fluxes and currents; outflow rates and the depths of outlet structures, which determine the volumes of water leaving the lake, whether they are withdrawn from the epilimnion or hypolimnion, and the resulting heat losses and currents; and heat exchange with the sediments.

Stratification affects nutrient cycling, water quality, primary production, and fish habitat in several ways. Unstratified lakes are relatively shallow, so vertical mixing occurs on a regular basis. This makes all inorganic nutrients released from sediments, decomposing organic material in the water column, metabolic processes in the lake, or external loads from the watershed immediately available for algal growth. Since the lakes are shallow, light penetrates a greater portion of the total water column, increasing the production rate per unit volume of water. The photic zone may extend to the bottom, promoting macrophyte growth across the whole lake area. Rooted macrophytes extract nutrients from the sediments, and release them to the water column through sloughing and decomposition. These internal nutrient loads are sometimes greater than external loads from the watershed. Mixing and atmospheric exchange often maintain aerobic conditions throughout the water column in shallow unstratified lakes. This allows the oxidation of nitrogen species, increases decomposition of organic matter, and maintains suitable habitat for fish and other aquatic animals. The presence of an oxidized microzone at the sediment-water interface greatly reduces the sediment release rates of phosphate and ammonia. Nutrient loads associated with river inflows always flow into the upper productive zone of shallow unstratified lakes, where they are immediately available for algal uptake.

In contrast, deeper stratified lakes have a lower hypolimnetic layer that behaves distinctly different than the surface epilimnetic layer in terms of nutrient cycling and primary production. Stratification prevents mixing of the waters between the epilimnion and hypolimnion. The epilimnion behaves similarly to the waters in shallow unstratified lakes, as described above. The deeper hypolimnetic waters are cooler and usually devoid of light, so primary production is absent. Macrophytes and nutrient “pumping” are confined to the littoral zone of the epilimnion, and are absent from the hypolimnion. Therefore, the average production rate per unit volume of water is lower in deeper stratified lakes. However, nutrients accumulate to high levels in the hypolimnion since algal uptake processes are absent, and organic materials remain in the water column longer and attain greater degrees of decomposition before settling on the bottom. In addition, the decomposition of settling organic matter combined with isolation from the atmosphere often produces anaerobic conditions in the hypolimnion. This prevents the development of an oxidized microzone at the sediment-water interface, which maximizes the rates of sediment nutrient release. However, stratification prevents mixing of the nutrient rich hypolimnetic waters with the lighted surface waters of the photic zone where photosynthesis occurs. The development of an anaerobic hypolimnion causes denitrification, which results in nitrogen losses from the water column through the production of gaseous nitrogen.

Stratification generally persists during the summer growing season of maximum light intensity. The lake mixes the nutrients accumulated in the hypolimnion with the productive surface waters during destratification, which occurs during the fall in most temperate lakes, and also during the spring in lakes that stratify during the winter from surface cooling. During stratified periods, nutrient loads from river inflows may enter the hypolimnion if the inflow temperature is lower than epilimnetic temperatures. This prevents them from contributing directly to the primary productivity of the surface waters until mixing occurs.

Stratification can be characterized by whether there are one or two stratified periods per year, the duration and period of stratification, the average depth of the upper mixed layer, the ratio of the mixed layer depth to the mean depth or maximum depth of the lake, the ratio of the mixed layer depth to the depth of the photic zone, the temperature ranges or average temperatures of the epilimnion and hypolimnion, and the thermal stability or density gradient between the two layers. Lakes that stratify and mix once per year are termed monomictic, and those that stratify and mix twice per year are termed dimictic.

Specific information on all of these characteristics can be determined from periodic temperature profiles, usually measured on a monthly basis. In the absence of temperature measurements, the existence and general type of stratification can be estimated from information on the lake depth and climatic conditions, or from information from other lakes with similar depth, climate, and hydrology.

4.4.8 Lake Origin

The geomorphologic processes responsible for the formation of a natural lake influences not only the size, shape, and geology of the lake itself, but also the morphometry and characteristics of the surrounding watershed. The geomorphology controls the watershed drainage characteristics, nutrient loading, lake hydrology, and lake chemistry, all of which control the metabolism and productivity of the lake. The lake morphometry along with the inflow volume influences the thermal stratification patterns, which in turn governs the distribution of nutrients, organisms, and dissolved gases. The shape and depth of the lake basin also determine the level of productivity in the lake. Deep lakes tend to be less productive since light and primary production are restricted to only a small portion of the upper water column, and sediment nutrient releases occur in the lower water column and are diluted by the greater volumes of water in deeper lakes. In addition, deep lakes are usually stratified during the peak growing season, isolating nutrients released from the sediments from the productive surface waters. In contrast, shallow lakes have light and primary production occurring over a larger portion of the water column, have more sediments in direct contact with the productive surface waters, and often remain unstratified. The shape of the lake also influences nutrient cycling and productivity. Steep sided lakes have smaller littoral zones compared to lakes with shallower sloping shorelines. Lakes with irregular shorelines have more littoral areas than comparable-sized lakes with more regular shapes. Littoral vegetation can contribute significantly to the overall productivity of these lakes.

Lake origins were differentiated into 76 types based on geomorphological inception in Hutchinson’s (1957) review of lake origin. These types were grouped into 11 major categories:

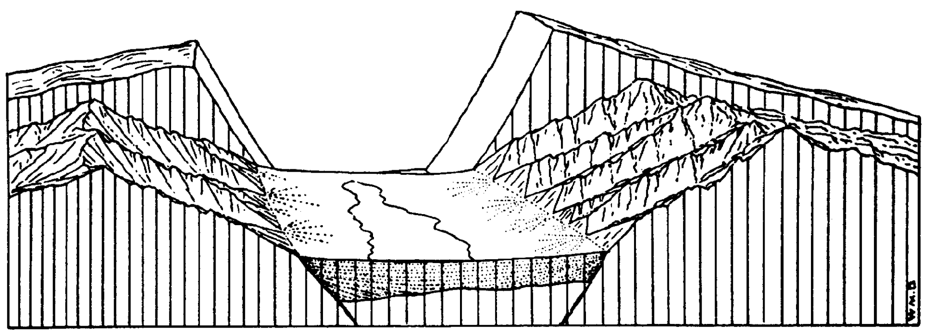
1. tectonic processes
2. volcanic activity
3. landslides
4. glacial activity
5. solution processes
6. river activity
7. wind processes
8. shoreline activity
9. organic accumulation processes
10. meteorite impact
11. reservoirs

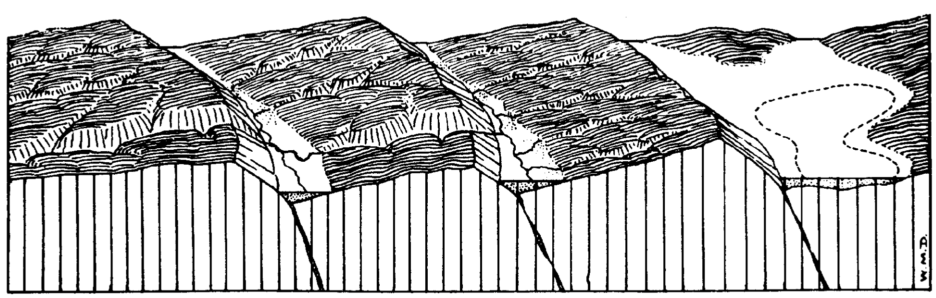
Because the geologic processes responsible for the formation of lake basins occur over widespread areas, lakes of similar origin are often found together in the same general region. In some lakes, several of the above processes may interact in the formation of the lake. In addition, lake origins are not always clear-cut, with different investigators proposing different mechanisms. The major categories of lake origin are described below.

4.4.9 Tectonic Processes

Tectonic lake basins are depressions caused by movements of the deeper parts of the earth's crust (excluding volcanic activity) (Hutchinson 1957). Lakes formed by faulting are one of the major types of tectonic lakes (Figure 4-4). Depressions occur between the land masses of both single fault displacements and down-faulted troughs (grabens). In single fault displacements, the region is broken into fault blocks that often become tilted, allowing water to accumulate on the most depressed parts of the blocks. In down-faulted troughs (grabens), a more elongate area is depressed, and the lake forms at the bottom of the depression. Lakes formed by faulting often have steep-sided basins, and are relatively deep and unproductive (Wetzel 1983).

Figure 4-4. Tectonic Lake Basins: A Depressed Fault Block Between Two Upheaved Fault Blocks and Diagram of the Great Fault Blocks of the Northern Sierra Nevada





Lake Tahoe and Pyramid Lake in Nevada are examples of graben lakes. Several lakes in the Sierra Nevada and in the Modoc lava fields of Northern California were formed from tilted fault blocks (Hutchinson 1957). In addition to lakes formed from major fault blocks, several smaller lakes occur along the San Andreas and Elsinore faults in California due to minor local subsidences (Hutchinson 1957).

Other tectonic mechanisms for the formation of lake basins include: 1) uplifting of marine sea beds to isolate large basins (e.g., Caspian Sea in Eastern Europe); 2) uplifting of marine sea beds with irregular topography from uneven sedimentation that causes water to accumulate in depressions (e.g., Lake Okeechobee in Florida); and 3) tilting, warping, or folding of the continental surface so that uplifted areas block former drainage outlets (e.g., Great Salt Lake in Utah), 4) impound river valleys (e.g., Lake Victoria in central Africa), 5) create basins between mountain ranges, or 6) change the hydrology so that drainage patterns are reversed and water flows into depressed areas of the altered land surface (Hutchinson 1957).

Volcanic Activity

Volcanic activity can result in the formation of lake basins in several different ways. The major mechanisms include depressions and cavities formed by: 1) explosive ejections of magma, 2) cooling and distortion of cooling magma, 3) collapse of overlying volcanic material where underlying magma has been ejected, 4) collapse of the surface crust of flowing lava fields over voids formed by the continued flow of the underlying molten lava, 5) the damming of river valleys by flowing lava, 6) the formation of barriers by a single volcano across a valley, and 7) a group of volcanoes forming a basin (Hutchinson 1957, Wetzel 1983). In addition to the purely volcanic activities listed above, subsidence along preexisting fault fractures may occur in conjunction with volcanic activity, so that extensive portions of the surrounding landscape may subside along with large scale caldera collapse from several volcanoes.

Lakes form in volcanic cavities when the groundwater table is above the bottom of the crater, or when an impermeable clay seal forms in the depression. Lakes formed by volcanic explosions are typically round, have small surface areas with diameters less than 2 km, but can be very deep. Lakes formed by the collapse or subsidence of the roof of a partially emptied magma chamber are larger, typically with diameters greater than 5 km. Some of the largest lakes associated with volcanic activity are formed when large-scale subsidence and caldera collapse occurs along preexisting fault fractures. Volcanic lakes often have basaltic basins and drainage areas, and the drainage areas are often restricted, so they are typically low in nutrients and relatively unproductive (Wetzel 1983). Examples of California lakes formed by volcanic activity include Medicine Lake in northeastern California, Snag Lake and the lake in Crater Butte in Lassen Park, and possibly Mono Lake (Hutchinson 1957).

Landslides

Landslides, rockfalls, and mudflows result in the formation of lake basins when they flow into valleys and block streams. Although these lakes are often temporary due to subsequent erosion of the unconsolidated debris after the lake rises and spills over the dam, sometimes they persist if the dam is high enough and an outlet develops in a different area, or if the landslide is large relative to the size of the stream (Hutchinson 1957). Large slides most commonly occur in mountain valleys where the stream is eroding relatively soft rock overlaid by more resistant material that becomes undercut and eventually collapses. Other causes include excessive rains acting on unstable slopes, and occasionally earthquakes. In addition to damming valleys, landslides can also produce smaller lake basins when a portion of the slide accumulates away from the valley wall where it originally detached (Hutchinson 1957).

Several California lakes have been formed by rockslides including Blue Lake, Pit Lake, and Lost Lake (all in northeastern California), Manzanilla Lake and Reflection Lake (in Lassen Volcanic National Park), and the two Kern Lakes west of Mount Whitney (in the Sierra Nevada) (Hutchinson 1957). Clear Lake (in the California coast range) was formed by a landslide on one side and a lava flow on the other side of an inter mountain valley (Hutchinson 1957).

Glacial Activity

Glacial forces have produced more lakes than any other geomorphological process (Hutchinson, 1957), but they occur only in areas that have been subjected to former glacial activity. The tremendous erosional and depositional forces associated with glacial ice movements result in the formation of lake basins through many processes. These include: 1) the damming action of rock debris (moraines) deposited during glacial recession; 2) damming action from the ice of existing glaciers; 3) glacial scouring of jointed, fractured, or weak rock material; 4) depressions created by the melting of ice blocks left in large glacial deposits (glacial drift or outwash); 5) depressions due to surface irregularities in glacial deposits (moraine, drift, or outwash); 6) erosion pools formed by water flowing down crevasses and under the ice of melting glaciers; 7) lakes formed in permafrost regions by the melting of ice deeper in the permafrost that accumulates at the surface; and 8) lakes on or within existing ice sheets produced by transitory thaw (Hutchinson 1957). Glacial lakes are formed when the glacial depressions are below the groundwater table, when glacial melt water fills the depression, or when drainage from the surrounding topography flows into the depression.

The size, shape, and characteristics of glacial lake basins depend on the particular glacial process and the surrounding topography and geology. Glacial action in mountainous areas of high relief produces different types of lake basins than the action of large ice sheets on more mature regions of gentle relief (Hutchinson 1957).

Lakes formed by glacial scouring in the valleys of mountainous areas are typically small and relatively shallow (<50 m) (Wetzel 1983). In contrast, glacial scouring by large ice sheets in non-mountainous areas can form very large lakes. For example, the Laurentian Great Lakes were formed by this process. Fjord lakes, formed by the deep scouring of long, narrow valleys in mountains at relatively low elevations, typically have long, narrow, deep, steep-sided basins.

Lakes formed by the damming action of moraine (or outwash) deposits or existing glacial arms are shaped by the topography of the dammed valley. This topography is also heavily influenced by the previous glacial activity. The moraine/outwash deposits or ice dams may occur in either the main river valley and block the outflow from a lateral tributary valley, or it may extend from a tributary valley and block the stream in the main valley. In some cases, separate glacial deposits occur within a valley, impounding the lake basin at both ends (e.g., the Finger Lakes in New York).

Kettle lakes, formed by melting ice blocks in glacial outwash plains, tend to be relatively small and shallow (<50 m), with highly irregular shape, size, and slope, and often with very irregular bottom relief with multiple depressions, ridges, and mounds (Wetzel 1983). This irregularity is due to the shape of the original ice fragment that melted to form the lake.

Lakes formed in permafrost regions are typically shallow. The shapes may vary, depending on the particular mechanism responsible for the lake formation. In some areas, disturbance of the vegetation cover allows the ground surface to thaw during the summer. Local subsidence occurs in the thawed soil, with melt water accumulating in the depression. These lakes grow as further thawing occurs at the lake boundaries, resulting in circular basins. Several of these basins may eventually fuse together to form a more complex basin, and levees may later form to isolate some of the basins. In other areas, lakes form by the melting of ice wedges, which are typically arranged in polygon patterns. Elliptical lakes are formed in some areas of nearly constant winds due to the thawing and erosion of permafrost at the downwind end of the lake (Hutchinson 1957).

Solution Processes

Solution lakes are created by cavities formed in soluble rock deposits that are slowly dissolved by percolating water. Most solution lakes are formed in limestone regions, but they can also form in other soluble rock formations including gypsum and halite (rock salt) (Hutchinson 1957). The lake depressions may develop by either the solution of beds at the surface, or by the collapse of underground cavities from the continual solution and erosion by groundwater. Solution lakes are often fairly circular and conically shaped, but may form very irregular depressions when several adjacent circular depressions fuse together. Solution lakes often form when the normal drainage cracks below sink holes become blocked with residual limestone or sediment. A less common type of solution lake is formed by the dissolution of ferric hydroxide and hydrous aluminum silicate in sandy sediments by the action of acid water created from plant extracts. This may leach a significant portion of the original sediment mass, typically leaving a circular depression where water accumulates. Various types of solution lakes occur in Florida, South Carolina, Indiana, Kentucky, and Tennessee (Hutchinson 1957), but they are not important in California, Nevada, and Arizona.

4.4.10 River Activity

The erosive, sediment transport, and depositional forces operating in rivers provide several mechanisms for the formation of lakes. These include: 1) plunge pool lakes excavated at the bases of waterfalls, 2) oxbow lakes formed when sediment deposition closes and the main river re-cuts a channel to isolate the loop of a meander, 3) lakes formed at the base of tributary streams by sediment deposition from the main stream creating a dam, 4) lakes formed in the main stream from large sediment deposits introduced through a lateral tributary that completely block the river, 5) lakes formed in the depressions of flood plains that occur by uneven deposition during floods, and 6) lakes formed in the deltas of major rivers where sediment deposition closes former drainage channels or 7) where previous delta lakes are divided into more lakes by sediment accumulation (Hutchinson 1957).

Plunge pool lakes occur under large waterfalls in the upper reaches of streams where the gradients are very steep. They typically form rock basins, with lakes remaining after the main course of the river has been diverted.

Lateral lakes formed by sedimentation blocking the base of a tributary are most common in the upper portions of the drainage where tributary valleys are steep. Lakes formed by sediment deposition from tributaries blocking the main stream (called fluviatile dams) are more rare, and usually occur in areas where steep gradient tributaries enter a wide and relatively flat river valley, with low gradient and low sediment transport capacity.

The other types of lakes formed by river activity occur in the lower reaches of river flood plains, usually in areas of gentle slopes. Deltaic lakes form at the downstream ends of river basins, where the river flows into a sea or a large lake. Oxbow lakes and deltaic lakes are often U-shaped, and are usually shallow. The outer sides of oxbow lakes are usually deeper than the inner sides, since meanders are formed by the erosion of the outer bank and deposition on the inner bank of a river bend.

Examples of lakes formed by river activity in California include the original Lake Tulare, Buena Vista Lake, Kern Lake, and the historical version of the Salton Sea.

4.4.11 Wind Processes

Wind action can form lake basins by redistributing sand or by deflating or eroding broken rock (Hutchinson 1957). Wind generated lakes occur primarily in arid regions such as the Southwest. Lake Basins may form in low areas between sand dunes, in areas where migrating dunes block a river valley, and in deflation basins where wind has eroded material from horizontally stratified rock or clay. The eroded material from deflation basins may be either carried away or deposited in crescent-shaped mounds at the downwind end of the basin. Since wind generated lakes occur in arid regions, they are often formed when the climate changes to a wetter period, so they may not be permanent. The lakes may form when a large amount of deflation produces a depression below the current groundwater table, or when the erosion leaves an impervious rock floor that becomes the lake bottom (Wetzel 1983).

Lakes formed between rows of sand dunes tend to be long and narrow, while those formed when a dune blocks a river valley tend to be triangular with the widest and deepest part close to the sand dune. The latter lakes occur both in coastal areas where a river is blocked from draining to the sea, and in sandy deserts where a dune terminates a river. Lakes formed in deflation basins vary in shape, with round regularly shaped lakes in some areas, long narrow lakes in others, and very irregular shapes in other areas (Hutchinson 1957). Wind generated lakes are often relatively shallow compared to other types of lakes.

4.4.12 Shoreline Activity

Shoreline sedimentation processes form lakes when deposition forms a bar or spit across the mouth of a bay or indentation in a coastal area or a larger lake, at the mouth of an estuary, or across a river valley (Hutchinson 1957, Wetzel 1983).

Marine coastal lakes are formed by longshore currents and sediment transport processes. Wave action suspends sediments and generates currents that transport the sediments in a direction parallel to the coastline. When the current traverses a deeper area associated with the inlet to a bay or estuary, the wave action may cease due to the greater water depths, resulting in sediment deposition. If the erosive forces of tidal currents, river inflows, and wave action are less than the deposition rates, a bar or spit will form parallel to the general coastline, eventually cutting off the entrance and forming a lake or lagoon. Marine coastal lakes commonly form in old drowned estuaries inundated by rising water levels or land subsidence (Hutchinson 1957, Wetzel 1983). A following period of slightly lowering sea level or the accumulation of sand hills over the blockage increases the width and stability of the blockage and the permanence of the lake (Hutchinson 1957). Coastal lakes may also form when two sand bars or tombolos form between the coast and an adjacent island.

The same littoral sedimentation processes that produce coastal lakes also operate inland in large lakes, producing shoreline lakes when a bar is deposited across a bay or river valley of a larger lake. In some cases, a lake may be split into two lakes by spits that approach each other from opposite shores due to current patterns related to lake morphometry (Hutchinson 1957, Wetzel 1983).

4.4.13 Organic Accumulation Processes

Several organic accumulation processes can result in the formation of lake basins. This includes: 1) damming by plant growth and organic detritus at the outlet of shallow depressions, 2) precipitation of large amounts of calcium carbonate from calcareous waters by the photosynthetic activity of attached blue-green algae and other plants, 3) and coral growth forming a rim around a basin in atolls (Hutchinson 1957, Wetzel 1983). These mechanisms are not important in California, Nevada, and Arizona.

4.4.14 Meteorite Impact

A relatively rare mechanism for the formation of lakes is the depression formed by the impact of a meteorite striking the earth. The force and energy of the impact creates tremendous pressure, heat, and shock waves. The intense heating expands water vapor and other gases in the rocks, producing an explosion and crater much larger than the size of the meteorite (Hutchinson 1957). This crater may then later become filled with water. Meteorite impacts are relatively rare events, and lakes of this origin are not important in California, Nevada, and Arizona.

4.4.15 Reservoirs

Reservoirs built by constructing a dam in a river valley or a smaller stream channel has produced many artificial lakes in California, Nevada, and Arizona. The larger lakes are typically built for water supply, hydropower generation, and flood control, while some of the smaller lakes have been built for recreation, aesthetics, and irrigation. In some lakes, dams have been built to enlarge existing lakes that were originally formed from natural processes. In some areas of the country, beaver dams have created relatively large and long-lived lakes that have become more permanent by sediment deposited over the dams (Hutchinson 1957). Other activities, such as excavation to extract mineral resources, and the construction of farm ponds for irrigation, have also resulted in artificial lakes.

Reservoirs and other artificial lakes constructed by man are young compared to most natural lakes. As a result, their shorelines are unmodified and follow the contours of the surrounding landscape. In many areas, this results in very irregular dendritic shorelines. Reservoir impoundments typically inundate terrestrial areas, providing nutrients and habitat from the drowned terrestrial vegetation during the early life of the lake. Most reservoirs are also relatively short-lived because of the large sediment loads carried by the dammed rivers.

4.4.16 Availability of Lake Origin Information

Although information on the origin of lakes will be available for reservoirs and many of the well-studied lakes, it will not be available for all lakes. Interpretation of the landscape and geology by a qualified geologist will be necessary to infer the origin for many of the unclassified lakes. This is not always straightforward, and different investigators may reach different conclusions concerning the origin. In some areas, knowledge of the origins of another lake in the same general region may be used to estimate the origin of other nearby lakes, since geologic events usually occur over large areas and often result in the formation of lake districts of similar origin. If it becomes apparent that classifying lakes based on origin is impractical because of data gaps, then a simplified approach that groups several similar categories may be used. Several of the above 11 categories are not important in California, Nevada, and Arizona. The simplest classification scheme would be to separate the lakes into reservoirs built by man and lakes formed by natural processes. The major importance of lake origin is the way in which it influences the lake size, shape, hydrology, geology, and watershed characteristics, which in turn affect nutrient dynamics and productivity. Since these characteristics are covered more directly by other classification criteria, it may be more useful to use these other criteria instead of lake origin.

4.4.17 Age

The age of a lake can affect the nutrient status in several ways. The general evolution of most lakes is from lower to higher productivity. Over time, nutrient and sediment loads accumulate in the lakes, causing the lakes to fill in and become shallower and smaller, and causing the nutrient concentrations to increase through increased internal loading from the sediments, increased primary production, and long term nutrient accumulation.

The rates of filling and nutrient loading depend on many factors, both external and internal to the lake. External factors include: drainage area; topography; soil types and erodability; vegetation types and coverage; nutrient contents of soil particles and pore waters; precipitation patterns and other meteorological factors; and land use activities that contribute nutrients, disturb soils so that they are more susceptible to erosion, or change the runoff patterns in the watershed. These factors control the external nutrient and sediment loads to the lake, as well as the lake hydrology and flushing rate. Human activities in watersheds have tended to increase the rates of both nutrient and sediment loading. The major nutrient sources include the use of fertilizers and detergents, agriculture, septic systems, and wastewater disposal. Land disturbance through agriculture and construction activities increases erosion and lake sedimentation rates. Land development increases the impervious areas of the watershed, which increases both surface runoff and stream flows. This leads to increased sediment and nutrient loads from the land surface, as well as increased loads from stream channel erosion.

The internal factors that influence the rates of lake filling and nutrient loading include the size and depth of the lake relative to the external loads, the level of primary production, the extent of the littoral zone, sediment nutrient contents and regeneration rates, and the extent of macrophyte coverage. Increased primary production increases the sedimentation flux of organic material. This increases the nutrient contents of the sediments, and therefore the rate of nutrient recycling from sediments, as well as the rate at which the lake fills in. As the lake fills in, more areas become shallow, increasing the extent of the littoral zone, and increasing the potential macrophyte habitat. Littoral vegetation and macrophytes are efficient at extracting nutrients from the sediments and releasing them to the water column through decomposition and metabolic processes. This increases the internal nutrient loading to the lake waters, which in turn increases phytoplankton production in the pelagic zone. The increased organic loads from these plants increases the bacterial metabolism in the lake, which further increases the rates of nutrient regeneration and the production of inorganic nutrients for photosynthesis. Eventually, the lake may become so shallow that it is dominated by the littoral zone. Littoral vegetation and dense macrophyte beds tend to trap sediments that flow into the lake, thereby increasing the rate at which the lake fills in. Most of these changes occur gradually over long time periods, but they are often accelerated by human influences in the watershed.

In addition to the gradual filling and increased production and nutrient levels that occur as most lakes age, there are also some short-term changes that occur in reservoirs during the first several years after they are impounded. Terrestrial vegetation and soils are typically inundated during the construction of a reservoir. Decomposition of the terrestrial organic matter in the vegetation and soils can provide a major source of nutrients in the early life of the reservoir. Productivity and nutrient levels may drop after much of this material is decomposed and the lake reaches a new equilibrium with respect to the external and internal nutrient loading regimes.

Information on the age of a lake will probably be hard to obtain, except for reservoirs and lakes that have been constructed by man. Some information will be available from lakes that have been well studied. Lake age can be determined using one of several sediment dating techniques. Most methods involve measuring radioisotope contents at the bottom of deep sediment cores. Radioactive carbon-14 (14C) measurements are the most useful technique for determining the age of older lakes. This approach allows age determinations up to about 40,000 years, but is limited for measuring relatively recent sediment. Recent sediments can be dated using lead-210 (210Pb), which because of its shorter half-life, is useful for age determinations up to about 150 years (Wetzel 1983). Cesium-137 (137Cs) can also be used to age sediments deposited in the last

45 years, but lakes that young should have easily available historical records and are most likely constructed by man.

In addition to the radioisotope dating techniques, some lake sediments have distinct alternating dark and light layers that represent seasonal variations in the composition of suspended matter that settles to the bottom. If one dark and light layer accumulates over a year, the pair is called a varve, and they can be used to date sediments in the same way as counting tree rings.

4.4.18 Dam Operation

Reservoirs can be categorized by several features of their dam and outlet operation that influence the hydrology and water quality in the lake. These include the dam height; whether the lake has a surface, deep, or multiple outlet structure; the depths of the outlets; the release volumes; and the seasonality of releases. The dam height is directly related to the depth of the lake, since the deepest portions of most reservoirs are just upstream of the dam where the river valley was impounded. The positions and depths of the outlet structures determine whether water will be drawn from the epilimnion or the hypolimnion, or from both. The volumes of release determine the hydrologic impacts of the releases, the amounts of nutrients and algae that will be flushed from the lake, and how far the lake surface will fluctuate. The latter factor has a major impact on littoral zone processes. The seasonality of the releases determines whether releases occur during the stratified season, the mixed season, or both.

All of the above characteristics of the dam and outlet operation have a major influence on the stratification pattern in the lake, as well as on the removal of nutrients and plankton in the water column. Deep outlets during the stratified season remove nutrient rich and oxygen deficient bottom waters, while surface or shallow outlets remove phytoplankton and lower levels of nutrients. The withdrawal depths and volumes influence the depth of the thermocline, which in turn influences the lake metabolism and productivity.

Information on the dam and outlet configuration and operating characteristics should be available from the utility or agency responsible for constructing and operating the reservoir.

4.4.19 Fish Community

Fish community characteristics can be used to categorize lakes since fish are typically the most sensitive and important organisms that must be protected from lake eutrophication problems. Fish can be categorized by whether there is a warm or cold water fishery, the particular types of fish that are present, and the relative abundances of the various fish types. However, fish communities are greatly influenced by the nutrient and phytoplankton concentrations in lakes, and may be considered more as response variables than primary criteria for classifying the nutrient status of lakes. Fish abundance increases with the overall productivity of the lake, which depends directly on nutrient and phytoplankton concentrations. In moderately productive stratified lakes, low dissolved oxygen levels typically develop in the hypolimnion during the summer months. This may restrict the lake to warm water fisheries, since the cooler hypolimnetic waters are not suitable for fish survival. Cold water fisheries can generally exist only in relatively unproductive lakes with low or moderate nutrient levels.

Fisheries information will probably only be available for well-studied lakes or lakes that are monitored by resource agencies or utilities. Relative information on fish catch can be obtained from creel censuses. Direct measurement of fish populations is much more difficult and expensive.

4.4.20 Other Biological Characteristics

The growing season is a measure of the potential productivity of an ecosystem since it captures aspects of both light intensity and temperature that are critical to primary productivity. The longer the growing season, the higher the phytoplankton, attached algae, and macrophyte densities that can be sustained, which in turn supports greater invertebrate and fish populations. The increases in general productivity influences nutrient cycling processes, as well as controlling the amount of excessive plant growth that occurs.

Growing season is defined based on terrestrial climatic conditions rather than conditions in the water body, so information is readily available. It is typically defined by the number of days per year exceeding a particular temperature. Although it does not include measures of solar radiation or light intensity directly, it includes them indirectly since solar radiation determines the air temperatures and the seasonal climatic cycles. Air temperature varies with both latitude and elevation. Therefore, a lake at a higher elevation will have a shorter growing season than a lake at the same latitude with a lower elevation. Growing season may be defined as the number of frost-free days per year, or the period in which the daily average temperature is above some threshold, such as 10 oC. Growing season can be easily determined from climatic data summaries available from the National Weather Service or various local sources. Corrections can be made to adjust for elevation changes for high elevation lakes remote from a local weather station.

4.5 Recommendations for Selection of Key Lake Classification Parameters

Although many of the above classification parameters could be used to divide lakes into different categories with respect to nutrient response, the focus should be on a few key parameters that have the most direct universal influence. These parameters should also be relatively independent from influence by other factors, and should be widely available or relatively easy to obtain. The three major factors that determine the nutrient status of lakes are the watershed loading characteristics, the lake size, and the hydraulic characteristics of the lake.

The watershed loading characteristics determine the external nutrient loads to the lake. This depends on the size of the watershed, the land uses and vegetation cover, the topography and soil characteristics, and the climate. All of these factors except the watershed size are generally incorporated into the ecoregional classifications. Therefore, the ecoregional classification can be used to distinguish between different land covers, topographies, soil characteristics, and climates. This leaves watershed area is the additional key parameter that should be added to characterize watershed influences. If an additional watershed parameter is desirable, it should probably be dominant land use (or vegetation cover).

The lake size is the most important factor influencing the nutrient response of the lake itself. The lake mean depth is clearly the most important independent variable, since it determines whether the lake will stratify, whether littoral communities and macrophytes will be dominant, the relative influences of primary production and sediment nutrient releases on the rest of the water column, and whether hypolimnetic anoxia is possible. However, an additional variable is also desirable to relate the size of the lake to the size of the watershed. The ratio of the watershed area to lake volume should be used as an additional size classification parameter for this purpose. This combined parameter would eliminate the need for the separate watershed area parameter described above.

The lake hydrology is best characterized by the residence time. Residence time determines the nutrient and phytoplankton flushing rates, and determines the amount of time available for nutrient sedimentation from the water column. Residence time integrates both flow and lake size information into a single parameter.

Beyond these key parameters, other parameters such as some measure of stratification (e.g., stratified depth to mean depth ratio) or water quality may also be useful. For example, non-algal turbidity is useful for separating lakes with high nutrients but low productivities. Some measure of the background constituent loads from the watershed, for example TDS, conductivity, alkalinity, or hardness, could also be useful. The most appropriate parameter from this group would be the one with the most complete data set.

Some of the other potential classification parameters are not recommended because they can be confounded by several interacting factors, or because the information will be difficult to obtain for many lakes. For example, lake origin, lake age, and fish community information is not available for many lakes. (Lake origin influences the size, shape, and hydrology of the lake, as well as the geologic characteristics of the watershed.) However, these factors can interact in many ways, and are better served by using these factors directly as classification parameters rather than using lake origin.

5.0 CAUSAL AND RESPONSE PARAMETERS CONSIDERED FOR NUTRIENT CRITERIA DEVELOPMENT

Quantifying whether a waterbody is over enriched with respect to nutrients is not a task that can be easily accomplished in a simple and direct manner. Because of this, easily quantified measures (parameters) are employed and used as indicators. These indicators include chemical, physical, and biological parameters that can be directly (ideally) or indirectly (as lines of evidence) linked to the effects of nutrient over enrichment. In general, there are four major characteristics to consider in assessing habitat measures as environmental indicators:

- The indicator must be relevant to the environmental/biotic endpoint.
- The indicator must be applicable to the waterbody in which it is used.
- The indicator must be responsive to human-caused stressors.
- The indicator must exhibit adequate measurement reliability and precision.

This section presents a list and discussion of parameters that can be used either directly, or indirectly to assess the impacts of nutrient over enrichment in lakes/reservoirs and rivers/streams. The effectiveness and availability of each parameter will be discussed and a list of recommended parameters presented.

Most of the parameters can and are used to assess both lentic (lakes/reservoirs) and lotic (rivers/streams) systems and, as such, there will not be a separate discussion for lentic and lotic parameters. Some parameters however, are exclusive to a specific waterbody type. These parameters will be identified and discussed separately.

5.1 KEY LIMITING NUTRIENTS

Phosphorus and nitrogen are the key nutrients that control primary productivity in most water bodies. Therefore, nutrient standards generally focus on these two constituents. The limiting nutrient in a particular water body is the nutrient that is present in the lowest level relative to the cellular needs of the algae. Based on the Redfield ratio, nitrogen requirements are about 7.2 times the phosphorus requirements on a weight basis. Therefore, if total nitrogen in the water is more than 7 times the total phosphorus, then phosphorus will be in low supply and limit algal growth. If the nitrogen is less than 7 times the phosphorus, then nitrogen will be limiting. However, the actual nutrient stoichiometry of algae varies somewhat between species, and more importantly with nutrient supply due to processes such as luxury consumption, which is the excess uptake and storage of nutrients when they are abundant to provide a temporary cellular supply for later deficiencies.

As a general rule, lakes tend to be phosphorus limited more often than nitrogen limited, so nutrient criteria to manage lakes often focus on phosphorus alone. However, many lakes are nitrogen limited, and many lakes are approximately balanced with Nitrogen-to-Phosphorus ratios close to 7. In addition, the N/P ratio often varies seasonally due to variations in external loads, internal loads from the sediments, and other internal biogeochemical cycling processes within the lake that deplete or augment one nutrient relative to the other (e.g., phosphorus coprecipitation and adsorption on calcium carbonate, nitrogen fixation from the atmosphere by blue-green algae). Therefore, the limiting nutrient may change seasonally throughout the year, or from one year to another.

Nitrogen, however, may have more importance as a limiting element of biomass in streams than in lakes. Lohman et al. (1991) reported low NO_3-N causing nitrogen limitation at 16 sites in 10 Ozark Mountain streams and cited sources for nitrogen limitation in northern California and the Pacific Northwest. Nitrogen was clearly the limiting nutrient in the upper Spokane River, Washington (Welch et al. 1989). Chessman et al. (1992) observed nitrogen to limit more than phosphorus in Australian streams. In streams and rivers of the eastern U.S., phosphorus can be a limiting factor in algal and macrophyte growth, and has been observed with greater frequency than nitrogen limitation (Newbold et al. 1983, Sharpley et al. 1994).

Other potentially limiting nutrients include carbon, silicon, and various micronutrients. Carbon dioxide continually exchanges between the surface water and the atmosphere, so free carbon dioxide is generally abundant for algal growth and is therefore rarely considered to be a limiting nutrient. However, in very hard-water lakes and rivers with high pH values, the carbonate system equilibria may shift so that little of the abundant dissolved inorganic carbon is present as free carbon dioxide. Silicon is important as a limiting nutrient for diatoms. Although diatoms are an important component of the algal community in many lakes, and in rivers as either sestonic (in slow-moving pools) or as attached as mats, other types of algae can thrive when silicon depletion limits diatoms. Many trace elements and other compounds such as vitamins are also critical for algal growth. However, these are needed only in trace amounts, and they are not generally measured in monitoring programs, so they are not considered to be important for setting nutrient standards.

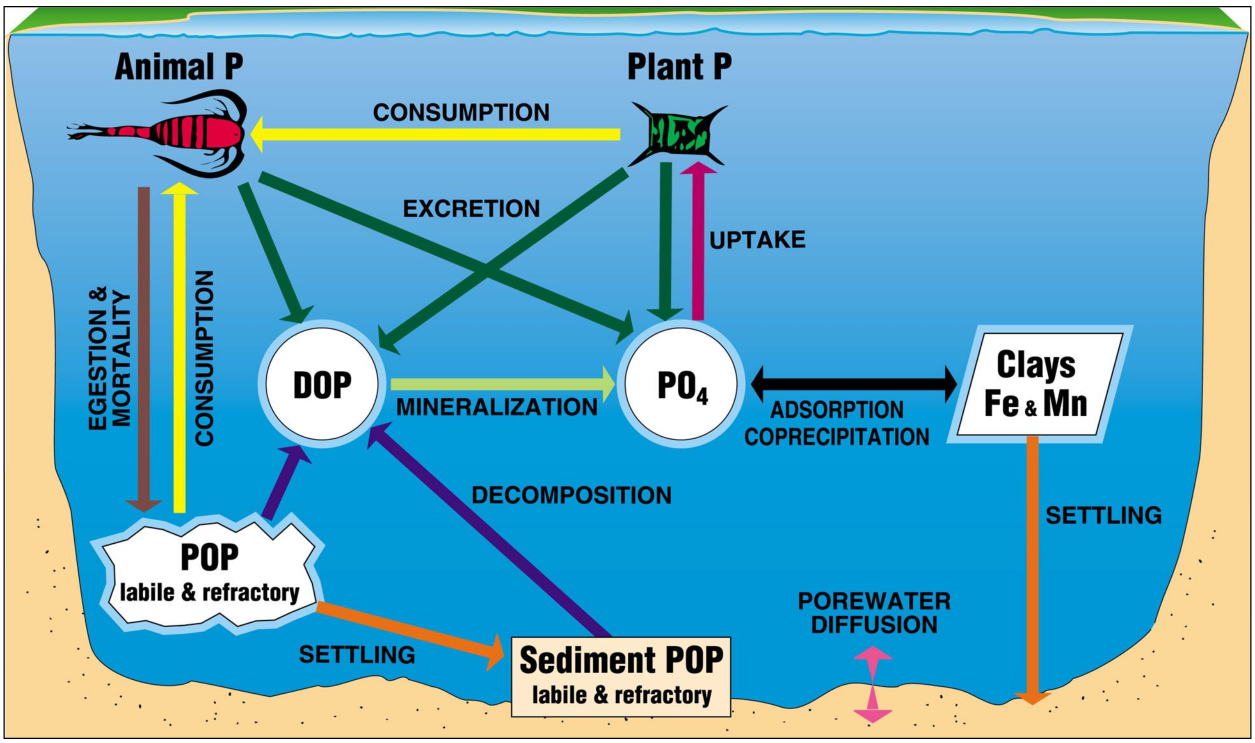
Although only dissolved inorganic nutrients are generally available for algal growth, most lake nutrient criteria are based on total phosphorus and total nitrogen. This is because during periods of high productivity, dissolved nutrients such as phosphorus and nitrogen are often depleted or present at very low concentrations due to rapid algal uptake. The nutrient concentrations can drop rapidly to very low values, even though algal densities are extremely high, and remain low for several months until the lake mixes in the fall. The total nutrient concentrations include both dissolved nutrients and nutrients bound in plankton and organic detritus. Therefore, they are more representative of the total nutrient pools available to support algal growth.

Dodds, et al. (1997) found a poor relationship between dissolved nutrients and periphyton biomass in streams. They found total nitrogen and phosphorus to be more related to stream biomass.

5.1.1 Phosphorus Cycle in Lakes

Phosphorus is the key variable most commonly used to characterize the trophic status of lakes. Phosphorus is present in both dissolved and particulate forms. The particulate forms include organic phosphorus incorporated in living plankton, organic phosphorus in dead organic matter, inorganic mineral phosphorus in suspended sediments, phosphate adsorbed to inorganic particles and colloids such as clays and precipitated carbonates and hydroxides, phosphate adsorbed to organic particles and colloids, and phosphate coprecipitated with chemicals such as iron and calcium. The dissolved forms include dissolved organic phosphorus (DOP), orthophosphate, and polyphosphates. The organic forms of phosphorus can be separated into two functional fractions. The labile fraction cycles rapidly, with particulate organic phosphorus quickly being converted to soluble low-molecular-weight compounds. The refractory fraction of the colloidal and dissolved organic phosphorus cycles more slowly, regenerating orthophosphate at a much lower rate. Figure 5-1 illustrates the phosphorus cycle.

Figure 5-1. Phosphorus Cycle in Aquatic Ecosystems



Dissolved phosphorus may be reported as total dissolved phosphorus, total phosphate, orthophosphate, and dissolved organic phosphorus (DOP). Care must be taken in interpreting monitoring data to determine if a reported total phosphorus value represents both dissolved and particulate forms (unfiltered sample), or only total dissolved forms (filtered sample). Confusion is also common in interpreting phosphate data, since it may not be clear if it represents only orthophosphate, or orthophosphate plus polyphosphates. The latter should be reported as total dissolved phosphates.

Dissolved orthophosphate, sometimes reported as soluble reactive phosphorus, is the only form that is generally considered to be available for algal and plant uptake. Although this is the primary bioavailable form, total phosphorus, including all dissolved and particulate forms, is a better determinant of lake productivity. This is because most of the phosphorus is tied up in plankton and organic particles during periods of high productivity. Often more than 95% of the total phosphorus is incorporated in organisms, especially algae (Wetzel, 1983). Any orthophosphate released by excretions, decomposition of organic matter, and mineralization of dissolved organic phosphorus is immediately taken up by phytoplankton. Phosphorus uptake and turnover rates are extremely fast, on the order of 5 to 100 minutes, during summer periods of high productivity (Wetzel, 1983). Therefore, the dissolved orthophosphate concentrations in the water column are often very low in highly productive systems. Phosphorus uptake and turnover rates are much slower during the winter due to the colder temperatures and lower light intensities. Uptake rates and optimum phosphate concentrations for growth vary among algal species, so seasonal changes in phosphate influence the structure and seasonal succession of phytoplankton communities.

Phosphorus concentrations and distributions between phosphorus forms vary both spatially and seasonally and can change rapidly due to both biogeochemical cycling processes and seasonal variations in phosphorus loading. The major cycling processes include algal and plant assimilation of orthophosphate, decomposition of organic detritus, mineralization of DOP, DOP and phosphate excretions by aquatic organisms, phosphate adsorption/desorption to suspended particulates and sediments, coprecipitation of phosphate, sediment release, macrophyte release, and sedimentation of plankton and other particulate forms of phosphorus. The external load sources include inflowing rivers and streams, direct runoff from the surrounding watershed, groundwater inflows, atmospheric deposition, and waste discharges. The phosphorus loads from the watershed depend on the phosphorus contents of the soils and parent rock material, vegetation characteristics including surface detritus and organic content of the soils, the amounts of animal wastes present, and human activities in the watershed such as fertilization and detergent use.

In oligotrophic lakes, both total and dissolved phosphorus often show little variation with depth. However, in more nutrient enriched lakes, total phosphorus concentrations are typically much higher in the hypolimnion than in the epilimnion, particularly near the lake bottom. This is due to the absence of algal uptake in the hypolimnion, decomposition of settled organic material, phosphorus release from the sediments, and stratification preventing the mixing of the bottom waters with the productive surface waters. Anaerobic conditions or very low oxygen levels commonly develop in the hypolimnions of nutrient enriched lakes. This destroys the oxidized microzone in the top few millimeters of the sediments that is normally present when the overlying waters are oxygenated. Sediments are anaerobic and highly reduced below the oxidized microzone due to bacterial metabolism associated with the decomposition of settled organic material. The oxidized microzone acts as a barrier to phosphorus release from the sediments. Sediment phosphate and iron are solubilized under reducing conditions, which makes them available for diffusion and release to the water column. However, under aerobic conditions, oxidation converts soluble ferrous iron to insoluble ferric iron, which in turn coprecipitates phosphate as ferric phosphate. This coprecipitation in the oxidized microzone prevents much of the phosphorus released from organic decomposition in the sediments from migrating upward to the water column. When the hypolimnetic oxygen in the bottom waters is less than about 1 mg/l, the oxidized microzone disappears and sediment phosphorus release becomes high, resulting in the accumulation of high phosphate concentrations in the hypolimnion.

The reverse pattern occurs in the epilimnion. High algal productivity during the spring and summer removes bioavailable phosphate and incorporates it into algal cells. The algal cells continually settle, transporting the phosphorus to the hypolimnion and sediments. Phosphate often drops to neglible concentrations throughout the summer, particularly if phosphorus is limiting and the lake is productive. The algae immediately take up any orthophosphate generated through biogeochemical cycling processes. When the lake destratifies in the fall, phosphate and total phosphorus in the epilimnion increase suddenly due to mixing with the phosphorus rich hypolimnetic waters. However, mixing also eliminates the low dissolved oxygen levels from the former hypolimnion. This results in the rapid oxidation of ferrous iron and the coprecipitation of ferric phosphate, removing some of the phosphate from the water column to the sediments. In addition, the oxidized microzone reforms in the sediments, slowing sediment phosphorus release. Phosphorus concentrations typically have little vertical variation due to mixing during the destratified season, except perhaps for elevated phosphorus near the bottom from sediment release or resuspension. When stratification develops again in the following spring, the cycle repeats with low phosphate and high algal concentrations developing in the epilimnion, and high phosphorus concentrations developing in the hypolimnion.

In shallow unstratified lakes, vertical variations in phosphorus may not develop, except for higher concentrations at the bottom from sediment release. The seasonal phosphorus patterns are often similar to the epilimnions of stratified lakes, with low phosphate levels and high algal and particulate phosphorus levels developing during the summer growing season. However, macrophytes are often abundant in shallow lakes where light penetrates close to the bottom. Rooted macrophytes can obtain a significant portion of their phosphorus requirements from the sediments, and can inhibit phytoplankton growth by shading and competition for light. Since dissolved phosphorus is continually regenerated by the decomposition of sloughing plant fragments and since shading may impede phytoplankton uptake, phosphate depletion may not occur during the summer as it would in phytoplankton dominated lakes. However, epiphytic algae on macrophyte leaves can also remove phosphate from the water column, even if phytoplankton populations are low due to shading. Macrophyte sloughing commonly occurs throughout the growing season, with a major pulse occurring in fall when the plants senesce. Phosphorus levels may jump abruptly following senescence if the macrophyte densities are substantial. Macrophyte effects can also be important in stratified lakes that are relatively shallow, and in the littoral areas of deeper lakes.

Since the bottom area to water volume ratio is high in shallow lakes, sediment release can be a major source of internal phosphorus loading, even if macrophytes are absent. Shallow lakes often have larger phosphorus levels than deep lakes in the same region since internal loads from the sediments can be a substantial portion of the total loads in shallow lakes. Sediment loading can be particularly high if the lake has a shallow anaerobic hypolimnion that eliminates the oxidized microzone at the sediment-water interface, or if turbulence produced by wind waves or boat traffic disturbs or resuspends sediments. High release rates can also be created through bioturbation effects if benthic invertebrate activity is high. Internal phosphorus loading from sediments or macrophytes can slow lake restoration progress for many years after external loads from the watershed or waste discharges are reduced. The sediments will continue to store large nutrient pools accumulated from decades of previous loading activities. These nutrients will continue to cycle back to the water column until they are eventually buried into the deeper inactive sediments through accumulation of cleaner sediments. This is true in both shallow and deep lakes, but is more pronounced in shallow lakes since the water volume is small relative to the sediment area.

The temporal dynamics of the phosphorus cycle make it more appropriate to use total phosphorus, rather than orthophosphate or some other form, in establishing nutrient criteria that reflect the trophic status of a lake. Orthophosphate is typically very low and sometimes immeasurable during the peak growing season of highly productive lakes. The orthophosphate concentration is more useful for determining phosphorus limitation of algal growth than for assessing productivity.

Total phosphorus can range from <5 ug/l in very unproductive lakes to >100 ug/l in very eutrophic lakes, although the usual range is between 10 and 50 ug/l in uncontaminated systems (Wetzel, 1983). Typical total phosphorus concentrations for different trophic categories are 8 ug/l in oligotrophic lakes, 27 ug/l in mesotrophic lakes, and 84 ug/l in eutrophic lakes (Vollenweider, 1979; Wetzel, 1983). The 1986 EPA Water Quality Criteria recommend a maximum phosphorus concentration of 25 ug/l in lakes to prevent eutrophication problems, and maximum concentrations of 50 ug/l in streams that enter lakes. Although inflow phosphorus concentrations drop in lakes due to phytoplankton uptake and settling, they may not drop 50 percent unless the residence is very long. This is particularly true if internal loads from sediments and macrophytes are important. Therefore, the 50 ug/l recommendation for inflowing streams may not adequately protect lakes.

5.1.2 Phosphorus Cycle in Streams

The dynamics of nutrient limitation in lotic environments is not as straight forward as that for lake environments. Unlike pelagic lake environments where phosphorus is often bound and tightly cycled within the biota, lotic environments are open and therefore continually receive phosphorus from upstream, groundwater, or runoff. Current also helps reduce limitation by reducing diffusion barriers. Under natural conditions, much of the phosphorus delivered to streams is bound in organic forms (e.g., in leaves, woody debris, invertebrates, etc.) and is then transferred between and among the different trophic levels within the lotic ecosystem. The role of macroinvertebrates in this transformation process is very important. Ward (1989) states that invertebrates may act as temporal mediators; their feeding activities result in a more constant supply of detritus to downstream communities by reducing the buildup of benthic detritus below levels subject to episodic transport during spates.

When anthropogenic sources of phosphorus are delivered to a stream, the ratio of dissolved phosphorus immediately available to algae may be high relative to particulate forms of phosphorus such as those attached to soil particles (Robinson et al. 1992).

Total phosphorus consists of both dissolved phosphorus, which is mostly ortho-phosphate, and particulate phosphorus, including both inorganic and organic forms (Sharpely, et al., 1994). Runoff from conventional tillage is generally dominated by particulate phosphorus; however, the proportion of total phosphorus as dissolved phosphorus increases where erosion is comparatively low such as with no-till fields or pasture (Sharpely, et al. 1994). Streams with low gradients and a morphology that enhances deposition of sediments as occurs in a channelized stream may continually release dissolved phosphorus from sediments much in the same manner as observed in lentic ecosystems. Nutrient recycling occurs during downstream transport and has been termed “nutrient spiraling” (Newbold, et al., 1983).

Relationship to Beneficial Uses

Phosphorus is an essential nutrient for plant growth. However, an increase in plant-available phosphorus may not necessarily increase primary productivity, as other factors (e.g., light and substrate) may be limiting (Scrivener, 1988). In small forest streams, light is often the limiting factor, while larger streams tend to be light saturated and nutrient limited.

In aquatic ecosystems, phosphorus is usually the limiting nutrient (Mohaupt, 1986). A survey of streams in Washington indicated that phosphorus was more likely to limit primary productivity in glacial streams and in streams draining grainitic watersheds, while nitrogen was more often limiting in streams draining volcanic landforms (Thut and Haydu, 1971). An increase in primary productivity usually leads to an increase in secondary productivity.

The desirability of increased biotic production is highly dependent upon local and downstream beneficial uses. For many headwater streams, a small or moderate increase in primary productivity might be desirable and be considered beneficial as it would likely result in increased fish production. However, if plant respiration begins to deplete dissolved oxygen or results in an increase in unsightly aquatic algae, it could be considered as an adverse effect.

Measurement

Methods for measuring the concentrations of the different phosphorus compounds in water are well known and performed by several accredited laboratories around the US. An important step is to determine which phosphorus species are of most interest and to identify the measurement technique most important to those species. Basically, there are three ‘phases’ of nutrients that can be measured (1) soluble (soluble reactive phosphorus [SRP]); (2) total (total P); and (3) mat nutrients (total phosphorus in the algal mat normalized to ash-free dry mass). The advantages and disadvantages of each are discussed below as presented by Biggs (2000). Note that CV stands for coefficient of variability.

Soluble

Advantages. A relatively direct measure of the bioavailable form of phosphorus and therefore mechanistically sound. Point source effluent effects can be assessed directly. Temporal variability is moderate to low relative to other nutrients (e.g., CV ~20-110% for SRP; Biggs and Close, 1989; Biggs 1995). Analyses are relatively quick and cheap. Data are generally readily available.

Disadvantages. Single measurements in time are poor indicators of nutrient supply regime because of the effects of biotic uptake and remineralization (Jones, et al., 1984; Dodds 1993; Biggs, 1995). The contribution of subsurface springs/seeps is difficult to account for. About a year of monthly measurements is best to obtain a reliable estimate of mean supply concentrations. Nutrients bound to organic matter might become available if the organic matter is deposited in quiescent areas, and therefore the projected dissolved nutrient supply could underestimate the actual supply. Low levels of detection are required for analysis.

Total Nutrients

Advantages. Incorporates all forms of the nutrient (dissolved and those bound to both organic and inorganic particulates), and thus yields a measure of the overall, potential, nutrient supply. Nutrients from subsurface inflows and groundwater are broadly incorporated in the measure. Total measurements are widely used variable in lake eutrophication management so this variable might be useful for comparing lentic versus lotic enrichment processes (Dodds et al 1998).

Disadvantages. Correlated with chlorophyll in the water column (Jones et al. 1984). Thus, a proportion of particulate nutrients in streams is probably derived from suspended algae, creating potential for circular reasoning in its application. Therefore the approach requires the following assumptions: that particulates and algae will eventually settle in quiescent areas; a proportion of the nutrients in these deposited particulates and algae will become available to benthic algae; and the proportion of bioavailable nutrients will be similar among streams and overtime, regardless of differences in the type of particulates (organic versus inorganic). Analyses require a digestion step, which makes processing more expensive. Frequent monitoring is required to get good estimates of mean concentrations (weekly for a year) because of moderate-high temporal variability (CV ~30-500% for total phosphorus) (Biggs and Close 1989).

Mat Nutrients

Advantages. A direct measure of nutrient status of the algae and can be related to specific growth rates through mechanistic models such as the Droop model (Auer and Canale 1982). Integrates the history of nutrient supply, including mineralized nutrients from deposited organics and subsurface supply from seeps and groundwater.

Disadvantages. It is difficult to relate back to supply concentrations of dissolved or total nutrients (therefore, it is difficult to use as a basis for managing nutrient loadings). The results are likely to be biased to varying degrees by the amount and type of non-algal particulates deposited in the mat. The influence of particulates will increase as the algal biomass:particulates mass ratio decreases. Analysis requires a digestion step and a measurement of organic biomass, which increases costs. Moderate temporal variability, so moderate-high sampling frequency is required (CV of mat % phosphorus commonly ~90-200%) (Biggs 1995).

5.1.3 Data Availability

Data for SRP and total phosphorus are readily available.

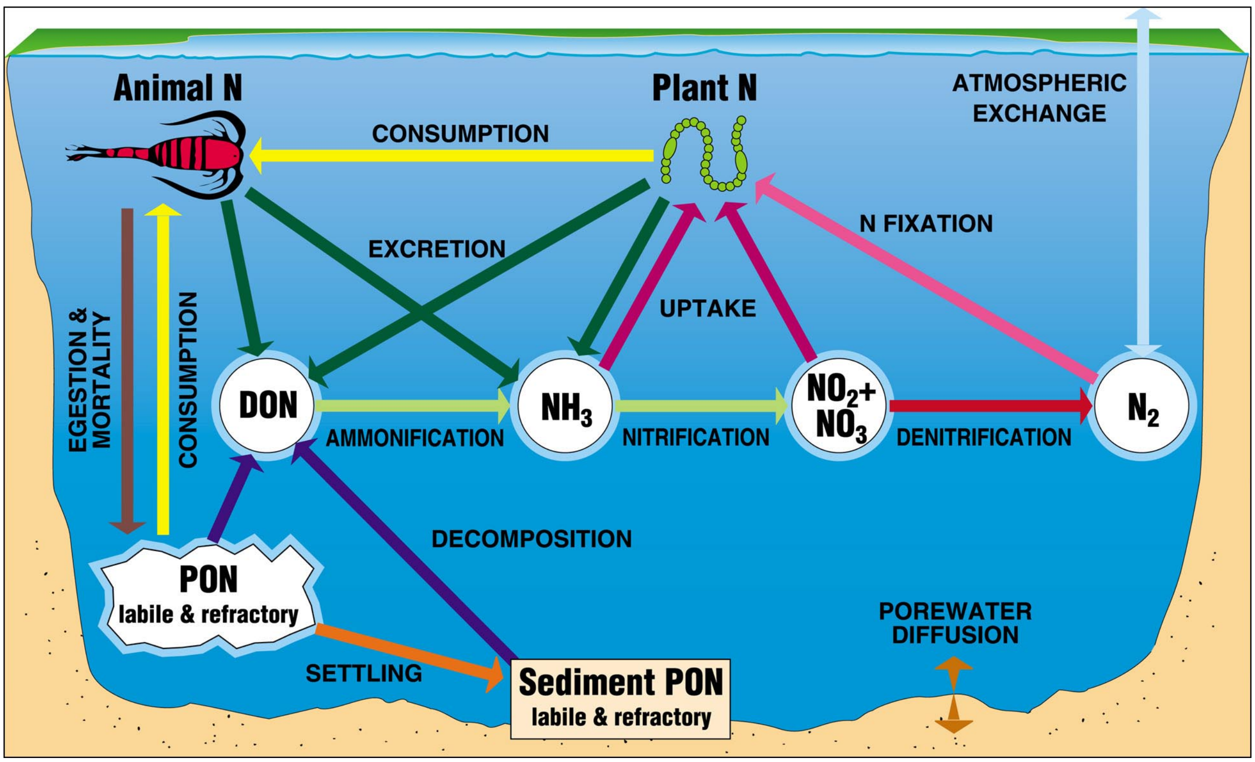
5.1.4 Recommendation

Establish criteria for total phosphorus.

5.2 NITROGEN

Nitrogen occurs in numerous dissolved and particulate forms. The particulate forms include organic nitrogen incorporated in living plankton, organic nitrogen in dead organic matter, and ammonia adsorbed to inorganic particles and colloids. The dissolved forms include dissolved organic nitrogen, ammonia, nitrite, nitrate, and dissolved molecular nitrogen gas (N_2). The organic forms of nitrogen include many compounds such as amino acids, ammines, nucleotides, proteins, and humic compounds (Wetzel, 1983). The nitrogen cycle is illustrated in Figure 5-2.

Figure 5-2. Nitrogen Cycle in Aquatic Ecosystems



Dissolved nitrogen may be reported as total dissolved nitrogen, total nitrogen, ammonia, nitrite, nitrate, nitrate plus nitrite, total Kjeldahl nitrogen (TKN), and dissolved organic nitrogen. TKN represents organic nitrogen plus ammonia nitrogen. Care must be taken in interpreting monitoring data to determine if a reported total nitrogen or TKN value represents both dissolved and particulate forms (unfiltered sample), or only dissolved forms (filtered sample).

Nitrogen concentrations and distributions between nitrogen forms vary both spatially and seasonally and can change rapidly due to both biogeochemical cycling processes and seasonal variations in nitrogen loading. The major cycling processes include algal and plant assimilation of nitrate and ammonia, decomposition of organic detritus, deamination and ammonification, nitrification, denitrification, nitrogen fixation by blue-green algae and bacteria, DON and ammonia excretions by aquatic organisms, ammonia adsorption/desorption to suspended inorganic particulates and sediments, sediment decomposition and release, macrophyte decomposition and release, sedimentation of plankton and other particulate forms of nitrogen, and gaseous exchange with the atmosphere.

Nitrate and ammonia, the major dissolved inorganic forms of nitrogen, are the only forms that are available for algal and plant uptake. Most algae preferentially uptake ammonia over nitrate since more energy must be expended to reduce nitrate to ammonia before it can be biologically assimilated. Therefore, uptake and photosynthesis rates are higher for ammonia than nitrate at the same concentrations. However, very high ammonia concentrations can have a toxic effect and inhibit photosynthetic uptake, particularly at high pH. Under these conditions, nitrate uptake rates may exceed ammonia uptake rates.

The main source of ammonia in lakes and rivers is the decomposition of organic matter (proteins, other organic compounds) by heterotrophic bacteria. Aquatic animals also excrete ammonia, but this source is small relative to decomposition. Intermediate dissolved organic nitrogen compounds are also released, but they do not accumulate to high levels because deamination and ammonification by bacteria is rapid (Wetzel, 1983). However, some of the dissolved organic nitrogen compounds are more resistant to bacterial degradation than others.

Nitrate and nitrite are generated through nitrification of ammonia. In aerobic waters, bacterial nitrification oxidizes ammonia to nitrate in a two-stage reaction in which ammonia is first oxidized to nitrite, and then nitrite is oxidized to nitrate. Nitrite oxidation is very fast, so nitrite levels in lakes and rivers are usually very low unless the waterbody is very nutrient enriched. Nitrate is the dominant oxidized form in lakes and rivers. Highest nitrite concentrations are typically found in areas where there is a transition from aerobic to anaerobic conditions, such as the metalimnion or upper hypolimnion of lakes, or the sediment interstitial waters near the lower boundary of the oxidized microzone. These represent areas that have low enough oxygen levels to slow down the nitrification reactions, but still high enough to prevent significant denitrification reactions. In addition to nitrification as a nitrate source, nitrate is also often the dominant dissolved nitrogen form in external loads from surface waters, groundwater, and the atmosphere. The riparian zone of streams plays a very important role in the nitrogen cycle as both aerobic and anaerobic conditions are usually present. Green and Kauffmann (1989) indicate that riparian zones are important for denitrification.

In anaerobic waters and sediments, bacterial denitrification rapidly reduces nitrate and nitrite to nitrogen gas (N_2). Nitrate is used as a hydrogen acceptor during the oxidation of organic matter under anaerobic conditions. Some of the N_2 produced during denitrification leaves the lake through outgassing, and some is fixed by blue-green algae and bacteria.

Particulate organic nitrogen in plankton and detritus is removed from the water column through sedimentation. Bacterial activity in the sediments decomposes the particulate organic nitrogen to release dissolved organic nitrogen and ammonia. Since most of the sediments are anaerobic, nitrification cannot occur, so ammonia levels increase in the sediment porewaters. Nitrification does occur in the oxidized microzone at the top of the sediments. Any nitrate or nitrite that diffuses into the anaerobic sediments from the water column or oxidized microzone is quickly denitrified to N_2. Ammonia sorbs to sediment particles under aerobic conditions in the oxidized microzone. Once the hypolimnion becomes anaerobic and the oxidized microzone disappears, the adsorptive capacity of the sediments diminishes, and sediment release of ammonia increases substantially.

Dissolved nitrogen gas (N_2) enters lakes and rivers through both atmospheric exchange and denitrification reactions. Both blue-green algae and bacteria can fix N_2, although nitrogen fixation by blue-green algae is usually greater than by bacteria. However, N_2 fixation requires more energy than assimilation of ammonia or nitrate, so blue-green algae typically fix nitrogen when ammonia and nitrate concentrations are low (Wetzel, 1983). Blue-green algae dominate the phytoplankton during periods when nitrate and ammonia are depleted by algal uptake because of their ability to fix nitrogen. Nitrogen fixed by bacteria in wetlands surrounding lakes or inflowing streams can also be a significant nitrogen source in some situations. In some cases, certain riparian plants, such as alder, can add nitrogen to riverine ecosystems by fixing atmospheric nitrogen.

In lakes, the seasonal dynamics of the nitrogen cycle along with the effects of stratification and dissolved oxygen profiles determine the temporal and spatial variations of the different nitrogen forms in the water column. However, the nitrogen speciation of major external load sources, and whether they enter the epilimnion or hypolimnion, can also play an important role, particularly if the external loads are high and the lake residence time is low.

Ammonia concentrations are usually low in aerobic waters because of algal assimilation and bacterial nitrification. Minimum concentrations typically occur in the epilimnions of lakes and in streams during the peak growing season. Higher concentrations occur lake hypolimnions, since algal uptake is minimal and ammonia is released through decomposition of particulate organic material in the water column and sediments. Higher ammonia concentrations can develop in anaerobic areas such as lake hypolimnions, deep pools in rivers, and sediments, since nitrification cannot occur there. In addition, the absence of an oxidized microzone maximizes ammonia release from the sediments. Stratification in lakes prevents most of the ammonia from reaching the productive surface waters where it could be utilized by algae. Ammonia concentrations can increase substantially during the fall in macrophyte dominated lakes due to the rapid decomposition of plant tissue following senescence. Ammonia concentrations in lake surface waters increase during the fall when stratification breaks down and hypolimnetic waters high in ammonia mix with surface waters.

Nitrate concentrations in the epilimnions of lakes are typically lowest during the peak growing season, and may be lower than detection limits if nitrogen is limiting growth. Concentrations are usually higher in the hypolimnion as long as it remains aerobic since ammonia concentrations are higher and algal uptake is minimal. However, under anaerobic conditions, nitrate will be absent from the hypolimnion since any nitrate will quickly be reduced to N_2 through denitrification. Nitrite is generally low in both the epilimnion and hypolimnion. Both nitrification and denitrification of nitrite are very rapid processes, which prevents nitrite accumulation under aerobic or anaerobic conditions.

Dissolved N_2 gas in lakes is usually at equilibrium with N_2 in the atmosphere during periods when the lake is well mixed. During stratification, the N_2 in the epilimnion may drop due to the reduction in solubility as the temperature rises, while the N_2 in the hypolimnion may increase due to denitrification (Wetzel, 1983).

Dissolved organic nitrogen (DON) released from decomposition of organic matter often represents over half of the total dissolved nitrogen in lakes, although it may be less in areas where inorganic nitrogen loads are high (Wetzel, 1983). Approximately two-thirds of the DON occurs as amino compounds, mostly polypeptides and complex nitrogen compounds, and less than one-third occurs as free amino compounds (Wetzel, 1983). Free amino acids are very low due to rapid uptake and decomposition by bacteria. Dissolved organic nitrogen is usually more abundant than particulate organic nitrogen (PON), with DON/PON ratios ranging from 5 to 10 (Wetzel, 1983). The DON/PON ratios decrease as lakes become more eutrophic and a greater portion of the nitrogen pool becomes tied up in algae and organic detritus. DON/PON ratios are closer to 1 in the epilimnions of productive lakes (Wetzel, 1983). Particulate organic nitrogen is generally highest during phytoplankton blooms due to algal assimilation of dissolved inorganic nitrogen.

As with phosphorus, the external nitrogen sources to lakes and rivers include inflowing rivers and streams, direct runoff from the surrounding watershed, groundwater inflows, atmospheric deposition, and waste discharges. In addition, nitrogen also enters lakes and rivers though atmospheric exchange and nitrogen fixation. The nitrogen loads from the watershed depend on the nitrogen contents of the soils and parent rock material, vegetation characteristics including surface detritus and organic content of the soils, the amounts of animal wastes present, and human activities in the watershed such as fertilization. Septic systems can also be significant sources since organic nitrogen and ammonia in the septic fields are oxidized to nitrate, which is highly mobile in soils. Therefore, it can enter lakes through shallow groundwater flows directly to the lake or through stream inflows from the watershed. In contrast, phosphate tends to be retained in soils by adsorption, so septic systems are not such a large phosphorus source unless they are situated close to receiving waters or are not operating properly. Atmospheric deposition is also more significant for nitrogen than for phosphorus in most areas due to contamination by combustion emission products.

The temporal dynamics of the nitrogen cycle make it more appropriate to use total nitrogen (dissolved and particulate), rather than only the bioavailable forms such as ammonia and nitrate, in establishing nutrient criteria that reflect the trophic status of lakes and rivers. Ammonia and nitrate are typically very low and sometimes immeasurable during the peak growing season of highly productive lakes. Ammonia and nitrate are rapidly taken up by phytoplankton, so much of the nitrogen is bound in plankton and organic detritus. In rivers, Dodds, et al., (1997) report that total nitrogen concentrations were more indicative of the nitrogen form that is ultimately bioavailable for benthic algal growth (periphyton) than dissolved nitrogen.

5.2.1 Relationship to Beneficial Uses

Certain nitrogen compounds have toxic effects at relatively low aqueous concentrations. Nitrate has been linked to methemoglobinemia (blue-baby) syndrome in human infants at concentrations of 10 mg/l of nitrate-nitrogen (EPA, 1986). Nitrate will also react with hemoglobin, and this can be hazardous for infants. Trout and salmon have also been shown to be sensitive to low concentrations of nitrate-nitrogen. Crunkilton and Johnson (unknown publication date) report that brook trout embryos exhibited increased mortality and decreased growth when exposed to nitrate-nitrogen concentrations as low as 6.25 mg NO_3-N/l.

Un-ionized ammonia is toxic to some aquatic invertebrates and fish at concentrations as low as 80 ppb, with chronic effects occurring at concentrations as low as 2 ppb (EPA, 1986). The toxicity of ammonia is affected by temperature, pH, and salinity.

Nitrogen is one of the most important nutrients in aquatic ecosystems. Most of the non-toxic effects of nitrogen result from the fact that increased inorganic nitrogen stimulates primary productivity and ultimately can result in stimulating secondary production (invertebrates and fish).

The desirability of increased biotic production is highly dependent upon local and downstream beneficial uses. For many headwater streams, a small or moderate increase in primary productivity might be desirable and be considered beneficial as it would likely result in increased fish production. However, if plant respiration begins to deplete dissolved oxygen or results in an increase in unsightly aquatic algae, it could be considered as an adverse effect.

Increased nitrogen loading in lakes is potentially much more serious than an increase in stream nitrogen because of the potential accumulation of nutrients (Schindler, et al., 1976). Over time, the accumulation of relatively small nitrogen inputs may stimulate algal growth to the point where eutrophication begins and beneficial uses such as swimming and fishing become impaired. Because of this, it may be required that criteria for rivers and streams be set at lower concentrations than local conditions warrant so that the beneficial uses of downstream waterbodies are protected.

5.2.2 Measurement

Methods for measuring the concentrations of the different nitrogen compounds in water are well known and performed by several accredited laboratories around the US. An important step is to determine which nitrogen species are of most interest and to identify the measurement technique most important to those species. As mentioned previously, Kjeldahl nitrogen combines both organic nitrogen and total ammonia. Total ammonia includes both the ionized (NH_4^+) and unionized (NH_3) forms. Dissolved nitrite and nitrate are often combined, as the concentration of nitrite in natural waters is generally small. Dissolved organic nitrogen can be obtained from the difference between Kjeldahl nitrogen and total ammonia. Adding Kjeldahl nitrogen to dissolved nitrite + nitrate yields total dissolved nitrogen.

Attention must also be given to the method of reporting the concentrations of the various species. For example, a concentration of 10 mg/l of nitrate includes the weight of both the nitrogen as well as the oxygen atoms in the nitrate molecule, while a concentration of 10 mg/l nitrate-nitrogen refers only to the amount of elemental nitrogen present as nitrate. The difference in the molecular weight of nitrate and nitrogen means that 10 mg/l of nitrate is only approximately 2.3 mg/l of nitrogen.

The same advantages/disadvantages apply to the various phases of nitrogen analyses (soluble, total, and mat) as with phosphorus.

5.2.3 Data Availability

Nitrogen data are readily available for most waterbodies. Our experience has shown us that, for Western Forested Streams, total Kjeldahl nitrogen values were reported most frequently. In this data set, total Kjeldahl nitrogen was reported seven times as often as total nitrogen concentrations. Additionally, there exists a very high correlation (slope of regression line = slightly greater than unity; n = 740 data points) between total nitrogen and total Kjeldahl nitrogen.

5.2.4 Recommendation

Establish criteria for total nitrogen for lakes and streams, or TKN for streams if there is a strong relationship with total nitrogen, as observed in the Ecoregion 2 rivers and streams dataset.

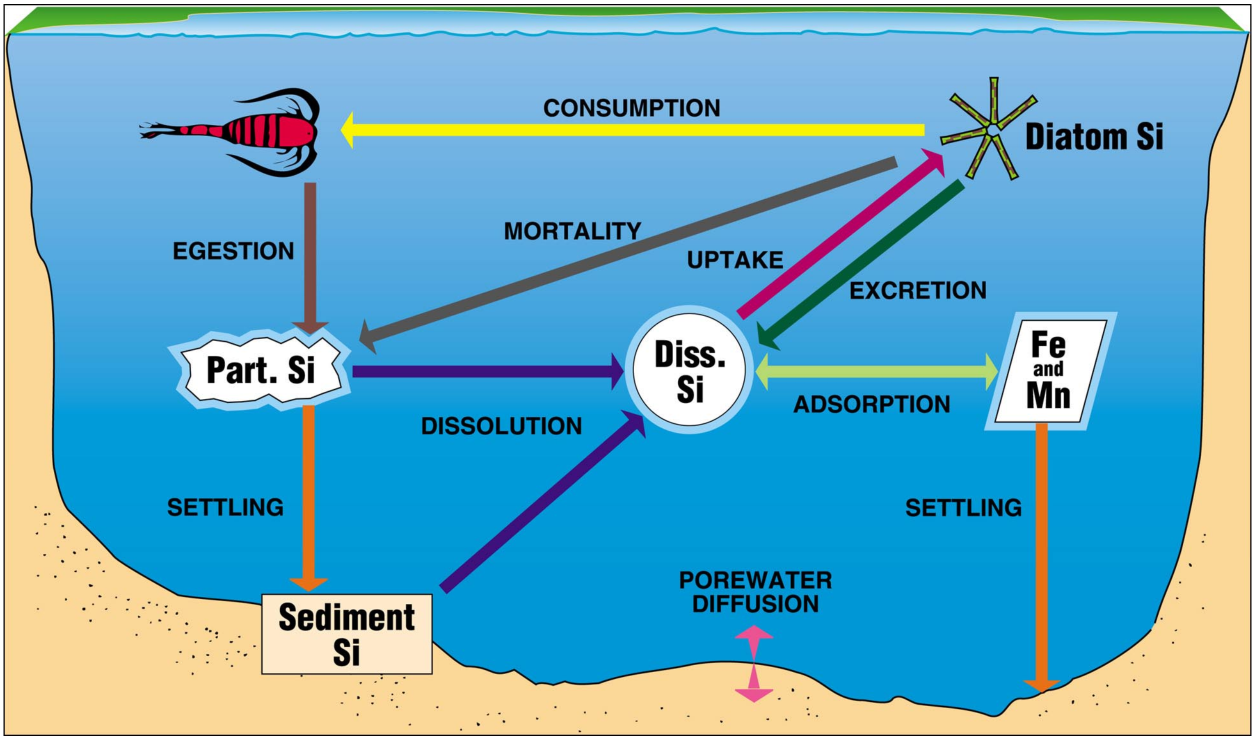
5.3 OTHER NUTRIENTS

5.3.1 Silicon

The earth is differentiated into the core, mantle and crust. The core is made up of the heavy elements, iron and nickel. The mantle forms a semi-fluid layer of less dense minerals with the composition of perovskite ($Mg\ SiO_3$) and olivine ($FeMgSiO_4$) while the crust, made up of lower density aluminosilicate minerals, 'floats' on the mantle. Although the Earth as a whole has a large iron component (35%), the composition of the crust is quite different with 28% silicon.

In order to become available for biological activity, the silicate rocks must be broken down. This is achieved by weathering, which may be (a) mechanical (physical processes of wind or ice) or (b) chemical (reaction with acidic and oxidizing substances). The rate of chemical weathering varies with the (1) physical conditions of temperature and rainfall and (2) the mineral composition of the rocks. The silica cycle is illustrated in Figure 5-3.

Figure 5-3. Silica Cycle in Aquatic Ecosystems



Chemical reactions are generally accelerated by high temperature and precipitation such that weathering decreases from tropical forests to temperate forests to grasslands to deserts.

Igneous and metamorphic rocks contain primary silicate minerals formed under high temperature and pressure deep in the earth. They are found in two classes--the ferromagnesian and the felsic series--depending on the presence, respectively, of magnesium or aluminium in their crystalline structure. Minerals in each series that are found as isolated crystal units, such as Olivine and Ca-Feldspar, are most susceptible to weathering. As a mineral is weathered, if forms a second mineral with slightly higher resistance to weathering due to increased linkages between the crystals. Both the ferromagnesian and felsic weathering series terminate in Quartz Si_4O_8, which has tetrahedral crystals linked in three dimensions and is therefore very resistant to chemical weathering. Consequently, chemical weathering during soil development leads to the loss of other constituents and the accumulation of quartz in the sand fraction.

The dominant form of weathering is the carbonation reaction involving carbonic acid. Plant roots and soil microbes release CO_2 into the air spaces in the soil leading to elevated concentrations of carbonic acid and increased weathering. Therefore organisms exert biotic control over the geochemical process of rock weathering.

An example of carbonic acid attack on silicate rock is weathering of the primary mineral Na-feldspar to the secondary mineral kaolinite with the loss of sodium and soluble silica (silicic acid).

$$2NaAlSi_3O_8 + 2H_2CO_3 + 9H_2O \rightarrow 2Na^+ + 2HCO_3^- + Al_2Si_2O_5(OH)_4 + 4H_4SiO_4$$

(The dominance of the bicarbonate anion in runoff indicates carbonation weathering.) Because only some of the constituents are lost, this is called incongruent dissolution and further weathering will lead to the loss of more soluble silica. Alternatively, olivine ($FeMgSiO_4$) undergoes congruent dissolution in water to release iron (Fe), magnesium (Mg) and Silicon (Si). The iron precipitates in the soil while the magnesium and silica are lost in runoff.

Acidity can also be increased in the soil by a range of organic acids (eg., acetic, citric, fulvic, humic, phenolic and oxalic) released by plant roots, microbes, and fungi. They promote weathering due to increased acidity and also by a process called chelation in which they combine with products of weathering to prevent the development of equilibrium between dissolved products and mineral elements.

Organic acids dominate the acidity of the upper part of a soil profile while carbonic acid is important below. Organic acids dominate weathering in cool temperate forests where decomposition processes are slow and incomplete, whereas carbonic acid drives chemical weathering in tropical forests where lower concentrations of fulvic acid remain after the decomposition of plant litter.

Looking at the concentrations of major elements in the continents and rivers indicates the large amount of silicon in rocks and soil. The rate at which elements may be mobilised through weathering varies: Ca > Na > Mg > K > Si > Fe > Al > P as elements become less soluble and form more precipitates.

Relationship to Beneficial Uses

The silicon in both freshwater and oceanic systems is taken up by diatoms, which are relatively large phytoplankters. They incorporate it into their cell wall (frustule) in the form of the mineral, opal. Phytoplankton species composition varies according to nutrient availability - adding phosphorus to a nutrient poor lake stimulates diatom growth, which may deplete Si supply and favor a shift to other plankton species such as green algae. However, since rivers provide a stable supply of silicon via soil weathering, these systems tend to resist becoming silicon dependent and, as such, silicon is not important to the development of nutrient criteria for rivers and streams.

Measurement

Silicon can be easily measured by any accredited lab.

Data Availability

Even though silicon is a limiting nutrient for diatoms, it is not generally part of freshwater datasets.

Recommendation

Not recommended as nutrient criteria parameter for rivers/streams or lakes.

5.3.2 Micronutrients

The micronutrients essential for algal growth include iron, manganese, zinc, copper, cobalt, molybdenum, vanadium, selenium, and vitamin B_{12}. In most natural waters, micronutrients are abundant enough that they are not growth limiting. Therefore, they are rarely measured in nutrient monitoring programs. However, certain micronutrients have been shown to be growth limiting in a few particular cases (Wetzel 1983). Since micronutrients are rarely limiting and since very little monitoring data are available, micronutrients are not important to the development of nutrient criteria.

5.4 WATER QUALITY RESPONSE

In addition to nutrients and biological measures of lake and river productivity, a few additional water quality parameters can be used as criteria for assessing the impacts of eutrophication. These are key variables that respond adversely to the effects of nutrient enrichment and which are critical to protecting aquatic habitats. These include turbidity, dissolved oxygen, and pH.

5.4.1 Turbidity

Turbidity refers to the amount of light that is scattered or absorbed by a fluid (APHA, 1980). Hence turbidity is an optical property of the fluid (Hach, 1972), and an increasing turbidity is visually described as an increase in cloudiness. Turbidity in streams is usually due to the presence of suspended particles of silt and clay, but other materials such as finely divided organic matter, colored organic compounds, plankton, and microorganisms can contribute to the turbidity value of a particular water sample.

In lakes, increases in turbidity due to algal blooms is important for both aesthetic reasons and because it limits light availability to the aquatic community and may therefore impair the aquatic habitat. In addition, algal turbidity is a measure of the algal densities and degree of eutrophication, and therefore an important indicator of all of the other water quality problems associated with excessive nutrient enrichment. Turbidity can change dramatically throughout the year due to algal blooms, storm water inflows from the watershed, large sediment loads associated with high stream flows, and strong winds or heavy boating which resuspends lake sediments.

Relationship to Beneficial Uses

Turbidity is an important parameter of drinking water for both aesthetic and practical reasons. Suspended matter provides areas where microorganisms may not come into contact with chlorine disinfectants, so high turbidity levels may limit the efficacy of normal treatment procedures (EPA, 1986b).

Turbidity also has a direct detrimental affect on the recreational and aesthetic use of water. The more turbid the water, the less desirable it becomes for swimming and other water contact sports. In many communities tourism and recreation are important economically and turbid waters could adversely affect the attractiveness of a water body for fishing, boating, swimming, or other water-related activities.

Most of the biological effects of turbidity are due to the reduced penetration of light in turbid waters. Less light penetration decreases primary productivity, with periphyton being most severely affected. Declines in primary productivity can adversely affect the productivity of higher trophic levels.

High turbidity levels adversely affect the feeding and growth of salmonids and other fish species. Lloyd, et al., (1987) report that the ability of salmonids to find and capture food is impaired at turbidities in the range of 25 – 70 NTU. Other studies indicate that growth is reduced and gill tissue is damaged after 5-10 days of exposure to water having turbidity levels of 25 NTUs (Sigler, 1980; Sigler, et al., 1984). At 50 NTUs some species of salmonids are displaced (Sigler, 1980; Harvey, 1989).

Increases in turbidity due to algal blooms is important for both aesthetic reasons and because it limits light availability to the aquatic community and may therefore impair the aquatic habitat. In addition, algal turbidity is a measure of the algal densities and degree of eutrophication, and therefore an important indicator of all of the other water quality problems associated with excessive nutrient enrichment. Turbidity can change dramatically throughout the year due to algal blooms, storm water inflows from the watershed, large sediment loads associated with high stream flows, and strong winds, which resuspend sediments.

Measurement

Turbidity can be measured directly and accurately using photoelectric turbidimeters. The turbidity is reported in nephelometric turbidity units (NTU). An earlier technique was the Jackson candle turbidimeter, which used a clear cylinder with a flame at the bottom. Water was slowly added to the column until the flame disappeared into a uniform glow. The results were reported in Jackson turbidity units (JTU). This older technique had several limitations including inability to measure low levels of turbidity and fine particles, and particle color affected the results. Photoelectric turbidimeters are generally used today since they are more accurate and do not have these limitations. Unfortunately, there is no standard conversion between NTU and JTU due to differences in the techniques.

Turbidity in lakes is most commonly measured as Secchi depth since it is easy and inexpensive. A disk standard size (20 cm diameter) and color (white or black and white) is lowered into the lake until it can no longer be seen, and is then raised until it can be seen again. The Secchi depth is the average of the two depths at which the disk first disappears and then reappears. No water samples or analytical procedures are required. Because the measurement is so simple, it can be used in volunteer monitoring programs. Also, since it is so cheap, frequent measurements can be made to capture the temporal variations in algal blooms, to look at long-term trends in productivity, or even to help predict the onset of a bloom. Secchi depth, together with phosphorus and chlorophyll a, has long been used in quantifying the trophic status of lakes. Secchi depth is one of the three variables used in Carlson's (1977) trophic indices of lake eutrophication. It is also one of the four parameters recommended by EPA (2000) for characterizing lake nutrient status. Secchi depth can be used in empirical relationships to estimate chlorophyll a, total phosphorus, or total nitrogen. Although some relationships are available in the literature, it is best if site-specific relationships are developed using algal and nutrient measurements from the lake. Secchi measurements are affected by interference from inorganic suspended solids and color, so care must be taken in interpreting the results. Measurements can also be affected by technique (e.g., whether the disk is lowered on the sunlight or shaded side of the boat), factors that influence the degree of lighting (e.g., weather and time of day), and equipment (e.g., whether a white or black and white disk is used). Therefore, consistency should be maintained throughout a monitoring program.

Turbidity is viewed by some as being the single most sensitive measure of the effects of land use on streams. This is due partly to the fact that relatively small amounts of sediment can cause a large change in the turbidity. On the negative side, this rapid response can lead to variability.

Data Availability

Turbidity data are readily available.

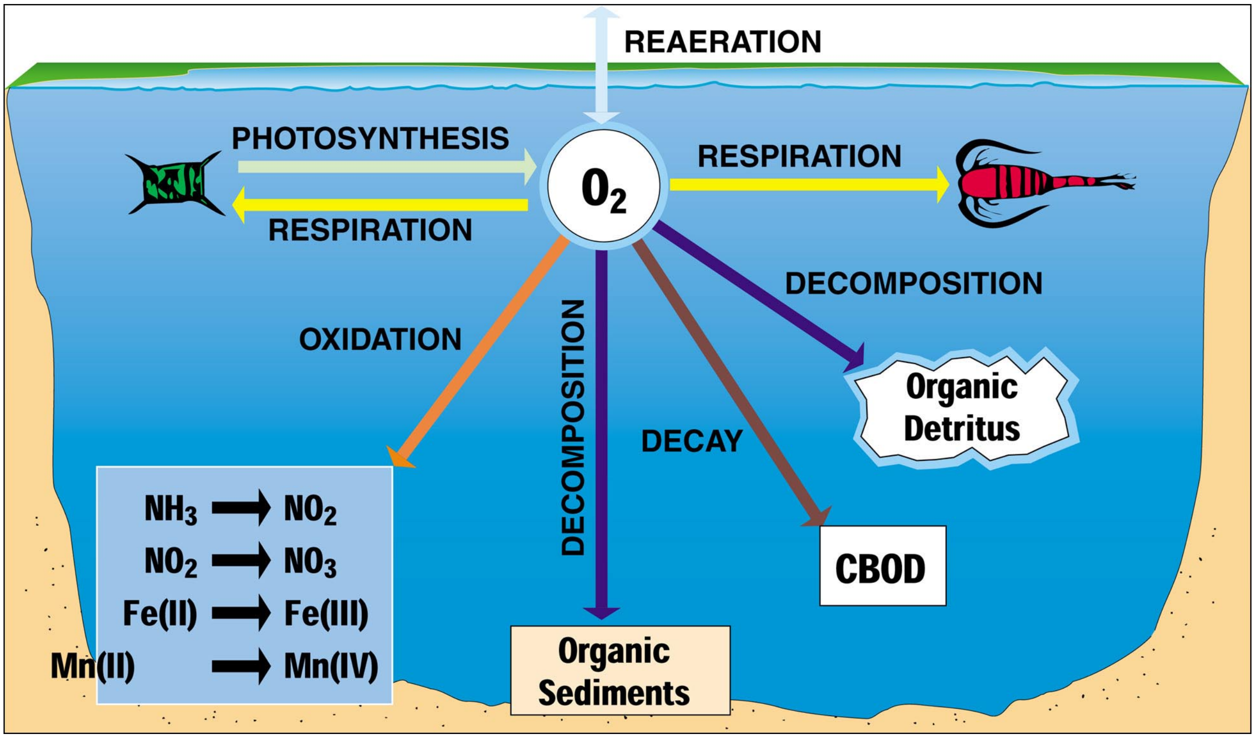
Recommendation

Turbidity has a direct linkage to eutrophication and should be included as an indicator.

5.4.2 Dissolved Oxygen

Dissolved oxygen is critical for the survival of most aquatic organisms. It also influences nutrient dynamics since it affects the solubility, availability, and forms of nutrients in the water column and surface sediments, as well as sediment release rates and losses to the atmosphere. Very low oxygen levels or anaerobic conditions prevent oxidation of ammonia to nitrite and nitrate, reduce nitrate and nitrite to nitrogen gas which can be lost to the atmosphere, reduce the adsorptive capacities of particles to increase sediment release of ammonia and phosphate and lower sedimentation losses from the water column, and eliminate the oxidized microzone which in turn reduces and solubilizes ferric phosphate to release the phosphate and iron from the sediments. In contrast, aerobic conditions promote oxidation of ammonia to nitrite and nitrate, prevent denitrification of nitrate and nitrite and the corresponding N_2 losses to the atmosphere, increase the adsorptive capacity of particles which inhibits ammonia and phosphate release from the sediments and increases sedimentation losses from the water column, promote development of the oxidized microzone which causes coprecipitation of ferric phosphate and prevents phosphate release from the sediments, and oxidize soluble ferrous iron in the water column to insoluble ferric iron which coprecipitates phosphate as ferric phosphate. The oxygen levels also influence the nature of the microbial community and the rates organic decomposition, mineralization, and microbially-mediated redox reactions. The effects of dissolved oxygen on nutrient concentrations and speciation also influences productivity, since it affects the bioavailability and cycling rates of the nutrients to algae and plants. The dissolved oxygen process is illustrated in Figure 5-4.

Figure 5-4. Dissolved Oxygen Processes in Aquatic Ecosystems



Oxygen enters lakes and streams through atmospheric exchange and photosynthetic activity. Water temperature, atmospheric pressure (and therefore elevation), and salinity influence the solubility of oxygen in water, which in turn influences reaeration rates. The major losses are oxygen consumed during microbial decomposition processes, respiration by all organisms, and chemical and biological oxidation reactions such as nitrification and oxidation of reduced iron and manganese. In addition, oxygen levels in inflows, inflow rates, inflow depths (dependent on inflow temperatures and lake temperature profile), outflow rates, and outlet depths relative to thermocline depth also influence the oxygen budgets in lakes, especially if residence times are low. These processes in combination with the seasonal stratification cycle, climatic factors that influence surface reaeration, and external BOD and COD loads determine the temporal and spatial variations of dissolved oxygen in lakes.

In general, oxygen concentrations in the epilimnions of stratified lakes, or throughout shallow unstratified lakes, decrease during the summer due to the reduced solubility of oxygen with increasing temperature. However, in very productive lakes, the oxygen concentrations can vary widely throughout the day due to photosynthetic oxygen production during the daylight hours and high respiratory oxygen consumption at night in the absence of photosynthesis. These processes can exceed reaeration rates, preventing equilibration with the atmosphere and resulting in diurnal cycles of supersaturation and undersaturation. Oxygen can drop to very low levels during the summer or fall in shallow productive lakes due to the rapid decomposition of organic material from senescing macrophytes or previous phytoplankton blooms, along with low reaeration rates resulting from light winds and low oxygen solubility.

Hypolimnetic oxygen levels are very different between productive and unproductive lakes. In unproductive lakes, hypolimnetic oxygen consumption is low, so the hypolimnion often has higher oxygen levels than the epilimnion. The lower temperatures of the hypolimnion produce higher oxygen solubility, and the hypolimnetic waters are usually saturated during spring mixing prior to stratification. In productive lakes, the reverse situation occurs, with very low or anoxic oxygen conditions in the hypolimnion. The high algal densities of productive lakes result in high organic loads to the hypolimnion and sediments, which rapidly consume oxygen during decomposition. The oxygen levels drop throughout the summer stratification period, with lowest levels typically occurring at the bottom near the sediments, where organic accumulation and microbial decomposition activity is highest.

In addition to these patterns, oxygen variations also sometimes occur in the metalimnion, or thermocline, region of lakes. Some lakes have higher oxygen levels in the thermocline than in the epilimnion or hypolimnion. This occurs when light penetrates the thermocline and algal photosynthetic oxygen production in this region exceeds oxygen consumption from decomposition, respiration, and nitrification. Some other lakes have the reverse oxygen pattern, with lower levels in the thermocline than in the epilimnion or hypolimnion. This is less common, but can occur when decomposing organic matter settling from the epilimnion accumulates near the thermocline due to reduced settling velocities in the much colder and denser hypolimnetic waters, or when dense populations of microcrustaceans congregate near the thermocline and produce large respiratory losses of oxygen (Wetzel, 1983).

During fall when stratification breaks down and the lake mixes again, oxygen levels throughout the water column return to saturation values in equilibrium with the atmosphere at the new temperature as wind mixing and reaeration reoxygenate the waters. Brief periods of very low oxygen can sometimes occur in surface waters at the beginning of destratification if the hypolimnion was anoxic and the hypolimnetic volume was much larger than the epilimnetic volume.

Additional complications occur in cold climates where the lakes freeze during the winter. Surface freezing prevents oxygen exchange with the atmosphere, and the ice and snow covers can prevent photosynthetic oxygen production. These are the two major processes that supply oxygen to lakes other than oxygen in inflows. In shallow productive lakes, respiratory consumption can significantly reduce oxygen levels during the winter, resulting in fish kills. In less productive and deeper lakes, the oxygen levels may not drop as low, since decomposition and respiration rates are low due to the cold temperatures, and because oxygen levels were very high prior to freezing because of the maximum oxygen solubility at sub-freezing temperatures.

Besides the seasonal vertical variations in the oxygen profiles of lakes, significant horizontal variations can also occur. Plants, benthic algae, and epiphytes in the littoral regions may generate more photosynthetic oxygen production than the phytoplankton of the open waters, but may also consume more oxygen from respiration, resulting in larger diurnal oxygen fluctuations. In addition, the decomposition of senescing macrophytes in the shallow littoral areas can introduce large organic loads that consume oxygen during decomposition. These effects may be most important in the late summer and fall at the end of the growing season.

In general, most mountainous streams have cool temperatures, rapid reaeration rates, and relatively low oxygen demand; thus stream water is normally close to or at saturation. Situations in which stream water may not be near saturation include: very slow, low-gradient streams where the rate of reaeration is slow; sites where fresh organic debris causes a large BOD; warm eutrophic streams where high levels of photosynthesis and respiration cause diurnal fluctuations in dissolved oxygen; and ponded sites such as those created by beavers.

Dissolved oxygen concentrations can also vary between the surface stream water and the water flowing through alluvial material in the stream bed. Dissolved oxygen within these alluvial materials is termed intergravel dissolved oxygen. Oxygen replenishment to these intergravel waters comes primarily from the exchange of well aerated surface waters.

Intergravel dissolved oxygen is controlled by the same factors as surface water, but there is no photosynthesis or reaeration. Oxygen demand comes from the fine organic debris entrained in the gravel and from the respiration of infaunal organisms. In spawning streams, fish eggs can also exert a measurable oxygen demand. For these reasons, dissolved oxygen concentrations within the streambed are generally lower than in the adjacent stream water.

Relationship to Beneficial Uses

Dissolved oxygen is critical to the biological communities in streams and lakes and to the breakdown of organic material and, as such, critical to almost every beneficial use classification.

Measurement

Dissolved oxygen concentrations can be quantified using either chemical (Winkler method) or potentiometric methods. The Winkler method is very accurate, providing there are minimal interferences from suspended material, other oxidizing agents, or certain organic compounds. It is also very difficult to perform in the field.

Field measurements of dissolved oxygen are most often made using the potentiometric method. This method uses a small, portable meter and probe that measures the rate of diffusion of dissolved oxygen across a membrane.

The timing of dissolved oxygen measurements is very critical. During the day, warming of stream water can depress the saturation concentration of dissolved oxygen and accelerate the rate of metabolic uptake. For slow-moving rivers and streams with high primary productivity, large diurnal fluctuations in dissolved oxygen concentration can result from algal photosynthesis and respiration. During the day, photosynthesis in excess of respiration is a source of oxygen. At night, photosynthesis ceases and respiration becomes an oxygen sink. Usually, in streams having high algal productivity, the lowest dissolved oxygen concentrations are found to occur just before dawn.

Data Availability

Dissolved oxygen data are readily available.

Recommendation

Water quality criteria already exist for dissolved oxygen. Dissolved oxygen concentrations can be directly linked to excessive primary productivity but its usefulness as an indicator is decreased unless diurnal measurements are made. Therefore, we recommend that diurnal DO concentrations be used as an indicator of beneficial use impairment.

5.4.3 pH

pH is defined as the concentration of hydrogen ions in water in moles per liter. Because the range of hydrogen ion concentrations in water can span more than 14 orders of magnitude, pH is defined on a logarithmic scale.

For practical purposes, the parameter of interest is not the absolute concentration of hydrogen ions, but the chemical activity of those ions. Therefore, the common measurement techniques of pH are based on hydrogen ion activity and not the absolute concentration of hydrogen ions.

Relationship to Beneficial Uses

pH can have direct and indirect effects on stream and lake water chemistry and the biota of aquatic ecosystems. A pH range from 5 to 9 is not directly toxic to fish, but a decline in pH from 6.5 to 5.0 resulted in a progressive reduction in salmonid egg production and hatching success (EPA 1986). The emergence of certain aquatic insects also declines below a pH of 6.5. Because of this, EPA has concluded that pH should range from 6.5 – 9.0 in order to protect aquatic life.

Indirect effects of pH on stream chemistry result from the hydrogen ion activity and the interactions between pH and a variety of other chemical equilibria. For example, at 5 ^{o}C the equilibrium concentration of un-ionized ammonia can increase ten-fold with a pH shift from 6.5 to 7.5. Similarly, the solubility of many metal compounds changes greatly with pH, which is of critical interest in areas having elevated concentrations of heavy metals sequestered in their sediments. Carbonic acid in cool, CO_2 saturated streams can stimulate a wide range of weathering reactions, and this will affect the aqueous concentration of a number of dissolved ions (e.g., silicon) (Reynolds and Johnson, 1972).

Elevated concentrations of river nutrients affects pH indirectly. Eutrophication stimulates algal growth. Waterbodies with high algal growth can exhibit considerable variation in pH diurnally. Maximum pH values usually occur in the afternoon when photosynthetic activity consumes carbon dioxide and dissolved oxygen concentrations are at a maximum. Increased pH levels can, in turn, increase the concentration of un-ionized ammonia and cause toxicity. Minimum pH values are observed at night (just prior to dawn) when carbon dioxide is being released by algal respiration. Lowered pH's can result in increased concentrations of dissolved heavy metals, which can cause toxicity.

Measurement

pH can be measured either colorimetrically or electronically. Since colorimetric methods are subject to interference from turbidity, color, colloidal matter, oxidants and reductants, they are suitable only for rough estimates. The most commonly used technique to measure pH is electronically using a pair of electrodes, with one being a constant-potential electrode and the other being an indicating electrode.

pH is temperature dependent and highly influenced by carbon dioxide and, as such, measurements should be made in the field immediately after taking the sample.

Data Availability

pH data are readily available.

Recommendation

Water quality criteria already exist for pH. pH levels can be directly linked to excessive primary productivity but its usefulness as an indicator is decreased unless diurnal measurements are made. Therefore, we recommend that diurnal pH levels be used as an indicator of beneficial use impairment.

5.4.4 Temperature

Water temperature is an easily measured parameter that has considerable chemical and biological significance. It is measured on a linear scale in either degrees Fahrenheit (^oF) or degrees Celsius (^oC). The Celcius scale is preferred and can be obtained easily from ^oF by using the equation:

$$^oC = 5/9(^oF - 32)$$

Stream temperatures are the net result of a variety of energy transfer processes, including radiationinputs, evaporation, convection, conduction, and advection (Brown 1983). Stream temperatures reflect both the seasonal change in net radiation and the daily changes in air temperature. These patterns of energy inputs and outputs are modified by stream characteristics such as flow velocity, flow depth, and groundwater inflow. Typically, peak daily temperatures occur in the late afternoon, and daily minima occur just before dawn. The seasonal pattern of stream temperatures generally is similar to the pattern of incoming solar radiation, but with a lag of 1 to 2 months (Beschta et al. 1987).

Relationship to Beneficial Uses

Increased water temperatures are known to increase biological activity. Keeton (1967) indicates that a 10^oC increase in water temperature can double the metabolic rate of cold-blooded organisms. This could result in reduced levels of dissolved oxygen and cause stress to aquatic organisms. Temperature also controls the rate of many chemical reactions. Eastman (1970) reports that a 10^{o}C increase in water temperature can double the rate of a chemical reaction. Additionally, the equilibrium between ammonium and un-ionized ammonia is highly dependent upon temperature and rapid temperature changes can initiate a series of repercussions regarding nitrogen cycling.

Elevated concentrations of river nutrients can have an indirect effect on water temperature. Eutrophication can result in increased sestonic and attached algal concentrations. This increase can result in increased absorption of incoming solar radiation, thus increasing water temperature.

Measurement

Temperature can be measured by either a thermometer or an electronic sensor. Thermometers are relatively inexpensive but should be calibrated if accurate measurements within 1^{o}C are required. Electronic sensors have the advantage of allowing continuous monitoring.

Average stream temperature measurements should be made in more turbulent reaches if possible. Water temperatures near the bottom of pools can be 5-10oC cooler than the surface water (Bilby, 1984). Usually, thermal variations within a stream result from inflows of cool water sources, such as groundwater or intergravel water, into slow moving reaches, pools, or backwater areas. In such cases, a single surface temperature can be misleading.

Data Availability

Temperature data are readily available.

Recommendation

Temperature is a controlling factor for several chemical reactions that can be used as indicators of beneficial use impairment (e.g., ammonia concentration, dissolved percent saturation).

5.5 BIOLOGICAL RESPONSE

5.5.1 Riparian Zone

Characteristics of the riparian zone are rarely considered as water quality parameters, yet the riparian zone directly affects many of the beneficial uses of water. The type and amount of riparian cover is an important controlling factor for stream temperatures and bank erosion, and both temperature and bank erosion can be directly related to habitat quality. The riparian zone also plays a key role in defining channel morphology and creating fish rearing habitat through the input of large woody debris. Finally, the riparian zone plays an important role in controlling the amount of sediment and nutrients reaching the stream channel from upslope sources.

Two parameters are used to assess the health of the riparian zone. The first parameter is the width of the riparian canopy opening and the second parameter is riparian vegetation. These two parameters will be discussed in the following sections.

5.5.2 Riparian Canopy Opening

The riparian canopy opening refers to the gap between the canopy of the riparian vegetation on opposite banks of a river or stream. Often, small streams are completely shaded by woody vegetation and hence have no riparian canopy opening in their undisturbed state. In steep, narrow, V-shaped valleys, considerable shading can result from the dominant upslope species rather than from the riparian vegetation. In lower-gradient and higher order streams, the stream channel by definition is wider and there commonly is a gap or opening between the parallel strands of the riparian vegetation. Streams with an alluvial valley floor tend to have more extensive and complex strands of riparian vegetation that develop in response to periodic flooding and high water tables.

These riparian and upslope forests that shade undisturbed stream channels can be altered by both natural disturbances (landslides, debris flows, and stream channel erosion) and anthropogenic land-use practices. Often, a highly interactive response exists between changes in morphology and changes in the riparian forest (Wissmar and Swanson, 1990). For example, channel or bank erosion often changes the size and location of the stream channels, which results in a corresponding loss of streamside vegetation and an increase in the width of the riparian canopy opening.

Monitoring of the riparian canopy opening offers a relatively rapid means of assessing the influences of anthropogenic activities on both the streamside vegetation and stream channel.

Relationship to Beneficial Uses

An increase in the width of the riparian canopy opening will allow more direct radiation to reach the stream and raise peak summer water temperatures. Less shading will also result in greater temperature fluctuations on both a seasonal and daily basis. A reduction in canopy cover may increase the amount of re-radiated long-wave radiation, thereby allowing more heat loss at night. Heat loss can be crucial to the icing up and formation of anchor ice in colder environments.

In light limited streams, an increase in the width of the riparian canopy opening can increase primary productivity (Gregory, et al. 1987). This may induce a corresponding increase in invertebrate and fish production. However, increased primary productivity may be offset by decreased inputs of detrital food subsidies, leaves, and other organic material from the riparian zone. The net balance between the increased primary production and decreaced detrital inputs will depend on the size of the stream and the presence or absence of other limiting factors, such as plant-available nutrients.

Changes in the size and structure of the canopy will adversely affect a wide range of animal species dependent upon riparian habitats (Deusen and Adams, 1989). A reduction in the width of the riparian zone may reduce the purported ability of the riparian zone to trap excess nutrients and sediments coming from upslope (Green and Kaufmann, 1989). An increase in canopy opening is likely to reduce the long-term delivery of large woody debris into the stream channel

(Grant, 1988). Large woody debris is an extremely important element in channel morphology, sediment transport, and quality of aquatic habitat in many forested streams (Bisson, et al., 1987).

Measurement

A detailed procedure for measuring and analyzing changes in the riparian canopy opening has been published as the RAPID (Rapid Aerial Photographic Inventory of Disturbance) technique (Grant, 1989). This requires a historical sequence of aerial photographs on a scale of at least 1-to-24,000. The basic approach is to:

- Identify initiation sites where the increase in riparian canopy opening begins.
- Determine the spatial links between the initiation sites and downstream increases in the width of the riparian canopy opening.
- Determine the continuity of open reaches along the stream.
- Measure the width of the riparian canopy opening.

The advantages of RAPID are as follows:

- There is an extensive database of aerial photographs that researchers can use to compare against existing canopy opening widths.
- The method is fairly straightforward and relatively easy to perform.

The major disadvantages include:

- The RAPID type approach is not as sensitive to change as ground-based measurements since an increase in canopy opening cannot be detected until a substantial increase in stream width has occurred. By this time, much of the original banks and vegetation will have been impaired.
- In some cases, it may be difficult to relate changes in the riparian canopy opening and the width of the stream channel to the potential causal factors such as landslides, forest harvest, extreme floods, or debris flows.

Data Availability

RAPID data is readily availably, however recent aerial photography requiring the appropriate scale may not be available.

Recommendation

This parameter can be very important in determining the limiting factor for primary productivity. However, since current conditions may not be readily available, its usefulness as a general indicator may be premature. Instead, this parameter should be included in cases where site-specific studies are recommended.

5.5.3 Riparian Vegetation

Riparian vegetation has been defined as that "growing on or near the banks of a stream or other body of water on soils that exhibit some wetness characteristics during some portion of the growing season" (AFS). Other authors have specified that the soil should be saturated within the rooting depth of the plants for at least some portion of the growing season (Platts et al. 1983; Minshall et al. 1989).

These definitions suggest that riparian areas are a particular type of wetland. EPA defines wetlands as "Those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support a prevalence of vegetation typically adapted for life in saturated soil conditions." In most ecoregions, one can find a variety of vegetation types on the streambanks and flood plains, including coniferous and deciduous trees, grasses, shrubs, forbs, ferns, and mosses (MacDonald, et al., 1991).

Relationship to Beneficial Uses

Riparian vegetation, and the exploitation of this vegetation, affects most of the beneficial uses of water through a variety of different processes. Some of the most important biological and physical effects of riparian vegetation on the beneficial uses of water include:

- providing organic material that can be used as food sources for aquatic organisms
- supplying large woody debris that alters sediment storage, influences channel morphology, and enhances fish production
- shading the stream and reducing temperature fluctuations
- reducing bank erosion
- providing habitat and cover for both aquatic and terrestrial organisms
- the social benefits of streamside aesthetics

The relative importance of these different functions is heavily influenced by vegetation type. Deciduous trees provide large amounts of leaves and other organic material, which are generally higher in nitrogen than coniferous debris, and thus more readily broken down by invertebrates in healthy ecosystems (Bilby 1988). More rapid breakdown leads to more rapid utilization and higher productivity.

Coniferous trees are an important source of large woody debris. Coniferous branches, boles and root wads tend to be larger than their deciduous equivalents, and this increases both their stability within the stream channel and the diversity of aquatic habitats, especially at higher flows (Sedell et al. 1984, Bisson et al. 1987). Coniferous wood does not decay as rapidly as alder and most other deciduous species, which also contributes to channel and habitat stability (Sedell et al. 1988).

Both coniferous and deciduous trees are effective in shading the stream and thereby reducing peak summer temperatures. Streams with little or no vegetative canopy may have lower winter minima and be more susceptible to anchor ice (Platts 1984).

All types of vegetation can be effective in reducing bank erosion, although they differ in the type of protection (Hackley 1989, Platts and Nelson 1989). Large trees and root wads can divert or deflect flow in small or moderate sized streams, and their roots can provide substantial protection during high flows. Grassy banks may provide a more complete cover, but they are not as resistant to undercutting or abrasion.

Few studies have been done on the filtering and buffering capacities of riparian vegetation (Green and Kauffman 1989). In most undisturbed forest ecosystems, the nutrient and sediment yields are so low that the filtering capacity of the riparian zone is not a key concern. In agricultural areas, however, nutrient exports are important and the riparian zone has been shown to be a sink for sediment as well as nitrogen, phosphorus, calcium, magnesium, and potassium sulfate (Lowrance et al. 1984, Lowrance et al. 1986, Green and Kauffman 1989). The influence of different vegetation types on sediment and nutrient yields, and in some situations water yield, is complicated by differences in other factors such as the prevalence of overland flow, height of the water table, rooting depth, root densities, chemical properties of the soil, nitrogen fixing ability of the plants, and seasonal growth patterns.

Various types of riparian habitat provide different types of habitat (Raedeke 1988). Species such as otters, beavers, deer, and bald eagles all have different habitat needs and are more or less dependent on riparian vegetation.

The importance of the riparian vegetation to the adjacent aquatic ecosystem diminishes in the downstream direction because of the increase of discharge and stream size (Bilby 1988). In small streams the riparian vegetation maybe the dominant source of organic matter, while in larger streams instream primary production tends to dominate (Hynes 1970). Removal or alteration of the riparian vegetation in a single reach can significantly alter temperature and water quality in low discharge, narrow streams, but the impact of a comparable change is likely to be undetectable in large streams or rivers (Bilby 1988).

Measurement

Although riparian vegetation affects many aquatic habitat and water quality parameters, generally it is more effective to monitor these parameters directly rather than monitoring the riparian vegetation since any direct linkage would be difficult to make (MacDonald et al. 1991). It follows that, with the exception of temperature, any precise measurement or characterization of riparian vegetation provides an accuracy, which cannot be translated into a more precise assessment of water quality or the impairment of beneficial uses.

Some of the most commonly measured parameters include vegetation type, vegetation cover, and vegetation density. Vegetation type is usually a qualitative categorization which can be as simple as tree, shrub, grass, or bare (Platts et al. 1983). More commonly, the vegetation type is based on the dominant overstory species or specified plant communities (Platts and Nelson 1989).

Vegetation cover usually refers to the downward projection of the canopy onto the ground surface (Husch et al. 1982). Visual estimation techniques can be used to provide a quick, qualitative measure (Platts et al. 1987). Quantitative measurements usually rely on point or line intercept methods.

Forest cover density can be assessed by measuring light intensity or by using a spherical densiometer (Lemmon 1957). The latter uses a point sampling technique to determine the amount of clear sky in the hemisphere centered over the observer.

Monitoring of the riparian vegetation is another means of assessing natural and anthropogenic impacts in the riparian zone and evaluating whether beneficial uses have been impacted. However, riparian vegetation cannot be used as a direct indicator of water quality except in the case of stream temperatures. For this reason, most water quality monitoring programs use relatively simple, qualitative indicators to assess the type, density, and cover of the riparian vegetation. Detailed quantitative monitoring is most appropriate for:

- assessing stream shading and predicting the thermal effects of changes in the riparian canopy
- predicting the size and future recruitment of large organic debris
- measuring the amount of cover for fisheries
- assessing bank stability and bank erosion as a function of vegetative cover

Data Availability

Dominant riparian vegetation types are generally available for most rivers using BASINS 2. Specific information regarding width of the riparian zone and vegetation species may be more difficult to locate.

Recommendation

Since there is no direct relationship between riparian vegetation and the effects of nutrient loading on a river, this parameter is not recommended to be included as an indicator.

5.5.4 River Aquatic Flora

The flora responsible for primary production in aquatic environments can be classified taxonomically, functionally, or morphologically. In classical plant taxonomy, the primary groups of aquatic plants are the algae, vascular macrophytes, and mosses. In most streams, the bulk of the productivity is due to algae (Hynes 1970).

Aquatic ecologists often use a functional classification with three primary categories: (1) free-floating, or planktonic forms (sestonic), (2) plants attached to the substrate (periphyton), and (3) plants rooted into the substrate (Weitzel 1979). The relative importance of these three categories is determined largely by the physical features of the habitat. Free-floating plants, for example, are significant only in still waters or large rivers where there is sufficient time for them to build up their populations. Rooted aquatic plants are rarely found in areas where bed material is coarse or subject to frequent transport. Attached plants, mainly benthic algae, are most important in gravel-bedded headwater streams.

Morphologic classification systems for aquatic flora can be simpler than the taxonomic and functional approaches. The usual distinction is between microflora and macroflora, but these are arbitrary size classes, and in the initial growth stages macroflora species can be part of the microflora (Hynes 1970).

Most studies of aquatic flora have concluded that the attached plant community is better suited to water quality monitoring (Weitzel 1979). Two terms are commonly used to refer to the attached flora, Aufwuchs and periphyton. Although some authors consider these synonymous, Aufwuchs a German term meaning attached growth) refers to all organisms growing on or attached to a substrate, and this includes heterotrophic organisms such as bacteria, bryozoa,and sponges, as well as small mobile organisms (protozoans and insect larvae) living within the mat (Power et al. 1988, Ruttner 1953, Wotton 1988). Periphyton often has a slightly narrower definition: aquatic flora growing on submerged substrates, which may or may not include the microflora (Cattaneo 1987, Hutchinson 1975, Odum 1971, Weitzel 1979). In forested streams in the Pacific Northwest, the attached algal communities are commonly referred to as benthic or epibenthic algae (Hudon and Legendre 1987). Diatoms usually are the most important and diverse algal group in benthic communities (Pryfogle and Lowe 1979). Epiphytic algae refers to attached microalgae (diatoms) that grow on the surface of macrophytes (Cattaneo and Kalff 1980).

Basu and Pick (1996) reported a positive relationship between sestonic chlorophyll a and total phosphorus in 31 eastern Canadian rivers (r2=0.76). They found no relationship between chlorophyll a and water residence time. Niewenhuyse and Jones (1996) compiled data from the literature (n=292) to show that summer mean chlorophyll a concentrations in temperate streams bore a strong (r2=0.67) curvilinear relationship with mean summer total phosphorus concentrations. They also found that stream catchment area had a significant effect on chlorophyll at all concentrations of total phosphorus in that smaller watersheds had less chlorophyll in their associated rivers at similar total phosphorus concentrations. This suggests that physical factors, and in this case, hydraulic flushing rate, may co-regulate chlorophyll, with smaller watersheds having relatively higher flushing rates. Heiskary and Markus (2001) also found a good relationship between stream total phosphorus and total nitrogen levels and sestonic chlorophyll a concentrations. They also found that watershed size was related to chlorophyll a concentrations, again, because of the hydraulic flushing effect. They also report that streams having high TSS or turbidity can exhibit lower chlorophyll a concentrations at identical nutrient concentrations because of photolimitation. Biggs (2000) states that, regarding periphyton, hydrology will influence stream chlorophyll so that even at the same nutrient levels, chlorophyll a concentration could be different.

Relation to Beneficial Uses

Benthic algae can be the dominant group of primary producers in stream ecosystems (Hynes 1970, Wetzel 1983). Mats of attached algae form rich assemblages of plant, bacterial, and animal species, all of which are important components of the food web (Weitzel 1979, Power et al.

1988). In small headwater streams, the contribution of organic matter by benthic algae may be outweighed by inputs of organic matter from riparian and forest vegetation. With increasing stream size, however, the importance of autotrophic production increases. Increased benthic algal production is linked to increased production of benthic invertebrates and fish (Gregory et al. 1987).

Ecologically, an increase in primary production can increase the production on fish and invertebrates in streams. However, nocturnal respiration can cause oxygen depletion in waters with high primary productivity and low reaeration rates. Even relatively small reductions in dissolved oxygen levels can have detrimental effects on both fish and invertebrate populations. Additionally, anaerobic conditions can alter a wide range of chemical equilibria, which can result in the mobilization of certain toxic pollutants as well as generate noxious odors.

Partial or complete removal of the riparian canopy will increase direct solar radiation, which may increase algal growth. Gregory et al. (1987) found that at 20% of full sunlight, benthic algal communities are photosynthetically saturated in headwater streams of the Cascades.

The relationships between benthic algal biomass in streams and nutrient concentrations are not well defined compared to those for lakes. While Dodds and Welch (2000) suggest setting criteria for both total nitrogen and total phosphorus, they state that the criteria should be based on the amount of chlorophyll that is acceptable, or at what point the benthic algal biomass is interfering with beneficial uses.

In downstream portions of slow moving rivers, all three functional plant groups (free-floating, attached, and rooted) can affect the beneficial uses of the water and be ecologically important habitats (Power et al. 1988). Large masses of benthic algae represent a potential nuisance by breaking loose and clogging water intakes, contributing to the oxygen demand, altering the substrate and benthic fauna habitat, interfering with angling and degrading the aesthetic environment of the stream. Additionally, aquatic macrophytes can adversely impact recreational uses such as swimming and boating as well as degrading the aesthetic value of the waterbody.

Based on results from 19 cases of enrichment, and survey results in which the coverage by filamentous forms increased with biomass, a threshold level for nuisance conditions of 100 – 150 mg chl a/m^2 was suggested (Horner et al. 1983, Welch et al. 1988). Welch, et al. (1988) report that this biomass level did not adversely cause oxygen depletion or impair benthic fauna. Welch et al. 1989 report that, since very low concentrations of SRP are required to saturate a river, as the SRP concentration entering the river segment increases, the effect is more apt to be expressed in the stream distance in which algal biomass exceeds some level (e.g., nuisance threshold level) than on the maximum biomass near the source of some high nutrient input. Presumably, the higher the inflow concentration from some source, the greater will be the stream distance in which SRP exceeds the threshold-nuisance-saturating level before uptake lowers SRP to that level. Additionally, Welch et al. (1989) state that due to the relatively low growth-saturating concentrations of SRP in running water, there is little hope that biomass at any stream point could be controlled by controlling ambient SRP. However, they state, the stream distance adversely affected below a nutrient source is a logical option to be managed. As stated previously, a relatively low nuisance threshold for biomass can occur at very low concentrations of SRP however, models used by Horner et al. (1983) and Seeley (1986) suggest that some control on maximum biomass occurs at higher SRP levels as well. Therefore, selecting a higher biomass as a threshold for nuisance conditions would allow for a higher SRP, yet result in a shorter stream reach exhibiting nuisance biomass levels.

Dodds and Welch (2000) state that ultimately, criteria based on existing data will need to be set based on what amount of benthic chlorophyll is acceptable and not on how nutrient amounts and ratios will influence algal communities.

Measurement

Of all of the aquatic plants, algae have long been the most widely used indicator of water quality and stream condition (Hynes 1966, APHA 1976, Weitzel 1979). Some advantages of using algae include the following:

- Their presence and growth integrate numerous physical factors.
- Their relatively short life cycle makes them useful indicators of short-term impacts.
- They are sensitive to certain pollutants, such as herbicides and excessive inputs of nutrients, which may not directly affect other organisms.
- Sampling can be easy and inexpensive depending on the situation.
- Fairly strong correlation between total phosphorus and sestonic chlorophyll a (Basu and Pick 1996, Niewenhuyse and Jones 1996) and between total nitrogen and sestonic chlorophyll a (Heiskary and Markus 2001).
- Relatively standard methods exist for evaluating the structural and functional characteristics of algal communities (EPA 1989).

Disadvantages to the use of algae and other aquatic plants are as follows:

- They are highly variable with location (Pryfogle and Lowe 1979).
- They are highly sensitive to small changes in current velocity, substrate type, and other physical factors (Weitzel et al. 1979).
- Considerable expertise and time are needed to identify both attached and free-floating micro-flora species.
- The use of qualitative information, such as presence or absence of particular species, may be invalid or appropriate only on a very coarse scale (Weitzel 1979. Weitzel et al. 1979).
- Quantitative relationships between nutrient concentrations and benthic algal biomass are not well characterized (Dodds et al. 1997, Dodds and Welch 2000).
- Other factors which influence algal biomass such as grazing (Welch et al. 1988). Shading via either in stream turbidity or riparian cover, flow velocity as it relates to nutrient uptake rate and biomass accumulation/sloughing, time available for biomass accrual, and substrate type.

Welch et al. (1989) use models based on growth kinetics and accumulation parameters and present a growth kinetics model for steady state biomass that can be used by water managers. They coupled this with a formulation to estimate the stream length for which periphyton biomass could be greater than the nuisance threshold. Dodds et al. (1998) condidered breaks in the cumulative distribution curves for region field measurements to classify stream trophic boundaries. This method allows one to determine where a specific stream fits into the larger database. Biggs (2000) presents a nomograph for use to predict periphyton biomass (chlorphyll a) as a function of dissolved inorganic nitrogen, soluble reactive phosphorus, and days available for accrual and tied it into a probabilistic model to assess risk of exceeding user-specified chlorophyll values.

Dodds et al. (1997) used a regression model that used field data to determine which parameters could be used to explain the most variance in benthic chlorophyll a. New regression models could be developed or existing models used to test California data. Another method would be to use a 'reference station' approach, which would target a stream or reach where existing periphyton biomass levels are acceptable and describe the nutrient characteristics (accounting for the other factors that could control biomass). Dodds et al. (1997) found that the regression, reference station, and probabilistic modeling approaches converged on similar nutrient targets.

Data Availability

Readily available data for benthic algal coverage is sparse. There are some datasets generated as the result of Total Maximum Daily Load studies (TMDLs). Additional data will need to be generated.

Recommendation

The main response variable to eutrophication is primary productivity and ultimately, criteria will need to be set based on what amount of benthic and sestonic chlorophyll a is acceptable. Beneficial use classification will be the driver. Benthic chlorophyll a can be quantified as mass per unit area or percent coverage, while sestonic chlorophyll a criteria will be based on mass per unit volume.

5.5.5 Lake Phytoplankton

Increased phytoplankton abundance and the resulting water quality problems is the primary problem associated with nutrient enrichment in lakes. Phytoplankton concentration is probably the most important response variable for characterizing the level of nutrient enrichment in lakes since it integrates the effects of nutrients, light, temperature, and hydrodynamic flushing. The major complications in using phytoplankton as an indicator are the short-term nature of phytoplankton blooms, the rapid seasonal changes in phytoplankton densities and species composition that require frequent sampling to fully characterize, and spatial patchiness in the distribution of phytoplankton. In addition, macrophytes compete with phytoplankton for light, so phytoplankton concentrations are commonly low in shallow macrophyte dominated lakes that are very nutrient enriched because of shading effects. Once macrophyte densities are reduced, algal blooms may proliferate.

Phytoplankton populations continually change over time due to variations in the rates of growth, respiration, grazing, settling, and parasitism. Growth rates are the most important, since they must offset all of the other losses in order for the population to increase. Growth rates depend on nutrient supplies, light, and temperature.

Phytoplankton have distinct seasonal variations in abundance and species composition as light, temperature, and nutrient supplies vary throughout the year. Different species have different growth responses to light, temperature, and each of the limiting nutrients, allowing different species to have more of a competitive advantage under different combinations of these physicochemical conditions. A few species with the highest growth rates under the prevailing environmental conditions will tend to dominate the phytoplankton assemblage, but other species will also be present at lower densities. Size and species selective grazing by zooplankton and protozoans, as well as species-specific parasitism, can also alter the population dynamics and species composition of the phytoplankton. All of these factors vary with both time and space, so the abundance and composition of the phytoplankton community has distinct seasonal variations, but also varies spatially within the lake.

The population dynamics and seasonal changes in species composition are very lake-specific since the environmental conditions depend on the interactions of many processes in both the lake and watershed. However, in spite of this variability, certain general trends in population dynamics and species succession have been observed in temperate lakes (Wetzel 1983).

Populations are typically low during the winter when light and temperatures are low, resulting in low growth rates even though nutrients may be abundant. Many different types of algae adapted to low light and temperature may be present at this time. As light and temperature increase during the spring, growth rates quickly increase, and the algal community typically reaches a spring maximum, followed by a decline during the summer months. Diatoms and cryptophytes are often the dominant algal groups in spring since they are adapted to both lower light and temperatures than some of the other algal types. The spring bloom often occurs after the onset of stratification, since epilimnetic mixing keeps the phytoplankton in the photic zone more than during destratified periods when the whole water column mixes. The summer decline following the spring maximum is due to several factors including nutrient depletion, warmer temperatures outside of the preferred optimums for growth, and increased grazing by zooplankton, whose populations also increase in response to both warmer temperatures and the increase in algal food supply. In lakes that are enriched in nitrogen and phosphorus, silica may become depleted by diatom uptake, causing diatom populations to decline. Green algae often then become the dominant algae type during the summer since they are better adapted to warmer temperatures and greater light intensities. If the lake is very phosphorus enriched, inorganic combined nitrogen may become depleted, which limits further green algae growth. At this point, nitrogen-fixing blue-green algae have a competitive advantage over all non-nitrogen-fixing algae, so they begin to proliferate and dominate the phytoplankton community. Blue-green algal blooms are an indicator of heavy phosphorus enrichment. All algal types may drop during the summer if both nitrogen and phosphorus become depleted. However, rapid nutrient recycling from the phytoplankton community, zooplankton grazing wastes, and decomposition or organic debris will continue to provide some nutrients to sustain algae throughout the summer. A second maximum in phytoplankton abundance often occurs during the fall when stratification breaks down and nutrient rich hypolimnetic waters are mixed back into the surface photic zone. Diatoms often dominate the fall maximum due to the lower temperatures and light intensities.

The above description of phytoplankton succession commonly applies to eutrophic temperate lakes, but is somewhat different in oligotrophic lakes, high altitude lakes, and more tropical lakes. In oligotrophic lakes, the summer phytoplankton populations are typically low, and blue-green algae are not abundant. Spring and fall maximums, commonly dominated by diatoms, still occur, but do not reach such high densities as in more productive lakes. In high altitude and more northern lakes, the growing season is shorter, and may result in a single summer maximum rather than spring and fall maximums. The overall productivity may also be lower. In more tropical or southern climates, the growing season is longer, which results in larger and more constant phytoplankton populations. For example, seasonal population changes may be on the order of only about 5 in tropical lakes, in comparison to about 1000 in temperate lakes (Wetzel 1983).

Phytoplankton abundance and primary production rates can vary significantly with depth due to vertical variations in light intensity, temperature, and nutrient supply. Light decreases exponentially with depth. Photosynthesis is often reduced near the surface due to photoinhibition by excessive light and ultraviolet radiation. Therefore, photosynthesis is often highest below the surface, but then decreases rapidly with increasing depth. The maximum photosynthesis rates vary with temperature. Although different species have different temperature optimums, species adapted to warmer temperatures typically have higher growth rates than those adapted to colder temperatures. Some phytoplankton can regulate their densities and buoyancy to position them in the water column where conditions are most favorable for their growth. For example, many blue-green algae accomplish this through the production of gas vacuoles. Other algae reduce settling rates by reducing their density and increasing frictional resistance with the water. This is accomplished through several mechanisms including production of gelatinous sheaths, accumulation of fats, regulating the ion content of cells, and cell shapes with protrusions and projections that increase frictional resistance (Wetzel 1983). The maximum photosynthesis rates will occur where there is the best combination of light intensity, nutrient supply, and temperature. This depth may vary with species, but it will also continually change over time as nutrients become depleted, light is reduced by algal shading, and temperature changes from heating and cooling. In addition, vertical mixing in the photic zone during windy periods continually redistributes both phytoplankton and nutrients.

Phytoplankton abundance and primary production rates can also exhibit significant horizontal spatial variability in lakes. This includes both random patchiness due to variations in microhabitat, grazing pressures, and hydrodynamic transport processes, as well as more systematic variations between littoral zones and pelagic areas, and between areas near to and remote from major stream inlets or other nutrient sources. Areas near stream inlets often have higher nutrient concentrations, but may also have higher levels of turbidity. The topography of the lake basin can also contribute to spatial variability in phytoplankton through its effects on the internal distribution of light, temperature, nutrients, and transport processes. For example, shallower areas may be subject to greater nutrient release from sediments, greater light penetration throughout the water column, warmer temperatures, and less stratification than deeper waters. However shallow areas may also have more turbidity from sediment disturbances.

Phytoplankton species assemblages can sometimes be used to assess the nutrient status of lakes since certain algal associations tend to occur as lakes become more enriched. Some of the typical associations are described in Wetzel (1983) and Hutchinson (1967). For example, different types of diatoms typically dominate in lakes of different productivity. *Cyclotella* and *Tabellaria* are often dominant in oligotrophic lakes, while *Asterionella*, *Fragilaria crotonensis*, *Synedra*, *Stephanodiscus*, and *Melosira granulata* are often dominant in eutrophic lakes. Blue-green algae such as *Anacystis*, *Aphaizomenon*, and *Anabaena* are often dominant in eutrophic lakes during the summer. *Chrysophytes*, such as certain species of *Dinobryon* and *Mallomonas*, may dominate in oligotrophic lakes because of their ability to take up phosphorus at very low concentrations. Dinoflagellates such as certain *Peridinium* and *Ceratium* species are sometimes dominant in mesotrophic and oligotrophic lakes. The use of phytoplankton species assemblages to classify lakes is limited since many of the species overlap between oligotrophic and eutrophic lakes, and they also shift seasonally as conditions change in the lakes. Summer nutrient depletion in the epilimnions of highly enriched lakes may result in the same assemblages that occur in oligotrophic lakes (Wetzel 1983).

Relation to Beneficial Uses

Phytoplankton abundance is the key variable used to assess the eutrophication impairment of lakes and is central to all water quality problems associated with nutrient enrichment. Increased phytoplankton is responsible for most turbidity problems, dissolved oxygen problems, and pH problems in lakes.

Chlorophyll a concentrations, in addition to phosphorus, are often used to determine the eutophication status of lakes. The annual average chlorophyll a concentrations reported in Vollenweider (1979) and Wetzel (1983) for a series of lakes with different degrees of eutrophication were 1.7 ug/l in the oligotrophic lakes, 4.7 ug/l in the mesotrophic lakes, and 14.3 ug/l in the eutrophic lakes. The ranges of the annual means in these lakes were 0.3 to 4.5 ug/l in the oligotrophic lakes, 3 to 11 ug/l in the mesotrophic lakes, and 3 to 78 ug/l in the eutrophic lakes (Vollenweider 1979, Wetzel 1983). The peak chlorophyll a concentrations were on the order of about 3 times the annual means, and averaged 4.2 ug/l in oligotrophic lakes, 16.1 ug/l in mesotrophic lakes, and 42.6 ug/l in eutrophic lakes (Vollenweider 1979, Wetzel 1983).

Measurement

Phytoplankton abundance is most commonly reported as chlorophyll a, which is the concentration of the photosynthetic pigment in all algal species combined. Other measures of algal abundance include biomass, cell counts, primary productivity measurements, and other photosynthetic pigments. Most of these measures represent the entire phytoplankton community, but cell counts can be easily separated into different species. Different size classes of algae can also be measured by using a sequence of different filter sizes to separate the algae before measurement by any of these techniques.

Chlorophyll a is the most widely reported measure of phytoplankton abundance. Chlorophyll a is the primary photosynthetic pigment, and it occurs in all algae. In contrast, the other photosynthetic pigments are restricted to particular types of algae. These include chlorophyll b, c, d, and e, carotenoids, and biliproteins. For example, chlorophyll b is found only in green algae and euglenophytes, and chlorophyll d is found only in certain red algae. Alpha-carotene is found only in certain green algae and cryptomonads. Biliproteins are pigment-protein complexes that occur only in certain blue-green algae, cryptomonads, and red algae. Light energy absorbed by these other chlorophylls, carotenoids, and biliproteins is ultimately transferred to chlorophyll a (Wetzel 1983).

Phytoplankton biomass can be measured as either the dry weight of algae, or as the dry weight of one of the major macronutrients in algae (carbon, phosphorus, nitrogen). Carbon is the most common choice. Biomass can also be estimated from chlorophyll a measurements using the ratio of chlorophyll a to biomass or carbon (or some other nutrient). Chlorophyll a is typically about 1 to 2 percent of the ash-free dry weight, but can be several times higher or lower than this range depending on species, physiological state, cell age, light intensity, and nutrient availability.

Phytoplankton cells can be counted directly using a microscope to enumerate individual species or major classes of algae (e.g., diatoms, greens, blue-greens, flagellates, etc.). This approach is very time consuming and requires a skilled taxonomist if group or species separations are required. Total cell counts of all species can be done more easily using automated instruments.

Data Availability

Total phytoplankton measurements as chlorophyll a are widely available from routine water quality monitoring programs. Measurements of the abundance of different phytoplankton groups are typically available only from ecological surveys. Phytoplankton productivity measurements are more rare, and are generally performed only in special studies.

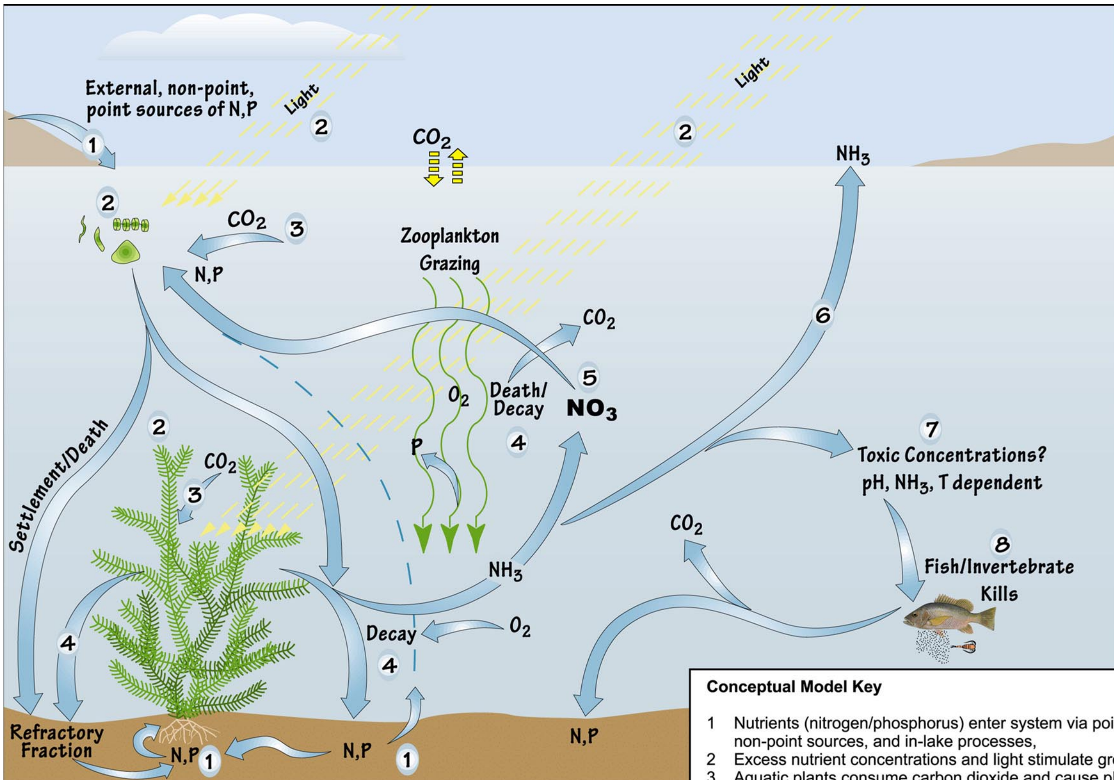
Recommendation

Phytoplankton chlorophyll a is recommended as one of the main response variables to assess eutrophication in lakes. Criteria will need to be set based on beneficial uses.

5.5.6 Macrophytes

Macrophytes are common components of littoral areas and shallow lakes. Macrophytes can be classified into four major groups based on their morphology, physiology, and the areas in which they grow. These are emergent macrophytes, floating-leaved macrophytes, submersed macrophytes, and freely floating macrophytes (Sculthorpe 1967). The first three are rooted macrophytes that are attached to the lake sediments, while the fourth is unattached and floats on or near the surface. Figure 5-5 illustrates macrophyte interactions with nutrients and other organisms.

Figure 5-5. Macrophyte Interactions with Nutrients and Other Organisms in Aquatic Ecosystems



Conceptual Model Key

1 Nutrients (nitrogen/phosphorus) enter system via point sources, non-point sources, and in-lake processes,
2 Excess nutrient concentrations and light stimulate growth,
3 Aquatic plants consume carbon dioxide and cause pH to rise,
4 Aquatic plants fragment or die releasing ammonia, phosphorus, carbon dioxide, and consume oxygen into water column or sediments,
5 Ammonia undergoes nitrification to yield nitrate. Nitrate is recycled back into system,
6 Ammonia diffuses into atmosphere,
7 Ammonia can become toxic if pH and ammonia concentrations are high enough, and
8 High concentrations of unionized ammonia can cause fish and invertebrate kills and release ammonia into the water column and consume additional dissolved oxygen.

Emergent macrophytes are similar to terrestrial plants and extend above the lake surface. They grow in shallow waters in the upper littoral zone to a depth of about 1.5 meters. Floating-leaved macrophytes have floating or slightly aerial leaves and grow in the middle littoral zone between depths of about 0.5 to 3 meters. Submersed macrophytes can grow at all depths in the photic zone. Although the vascular angiosperms are limited to a maximum depth of about 10 meters, nonvascular macrophytes such as macroalgae can grow in deeper waters with light. Freely floating macrophytes typically occur in protected areas.

Rooted macrophytes typically obtain most of their nitrogen and phosphorus from the rich supplies in the sediments, and can grow well even in oligotrophic lakes. Although many submersed macrophytes can obtain nitrogen and phosphorus from both sediments and water, concentrations in porewaters are generally much higher. However, nutrient assimilation from water can be substantial when the concentrations are high. Epiphytic algae growing on the surface of macrophytes can also remove nutrients from the water.

Macrophyte growth is limited primarily by light and water temperature. Macrophytes compete with algae for light in the water column. In areas where macrophytes are abundant, algal densities are typically low due to shading by the plants. Conversely, in areas where turbidity is high due to phytoplankton blooms, macrophytes are not usually abundant.

In contrast to phytoplankton, whose populations can increase tremendously over a period of a few days, macrophytes have a more regular growth cycle. Many macrophytes are perrenial and typically start to grow in the spring, reach maximum densities in the summer, and then die back in the fall due to natural senescence. Large amounts of nutrients are released to the water when the plants decompose during this senescent period. Nutrients are also released continuously during the growth season due to the continual fragmentation and sloughing of plant tissues, as well as the senescence of early cohorts for multiple cohort species. The peak summer biomass of perrenial species typically has a turnover rate of 1.5 to over 3 times per year (Wetzel 1983). Since most of these plant nutrients are originally obtained from the sediments, the macrophytes can be a significant source of internal nutrient loading to the lake.

Phytoplankton blooms are common following macrophyte senescence due to the nutrient release and the sudden availability of light. Phytoplankton blooms are also common in lakes where macrophytes have been reduced or eliminated through various control measures, such as harvesting or herbicide application. In these instances, removal of the plant canopy makes light available for rapid phytoplankton growth.

Macrophytes can be relatively abundant even in oligotrophic lakes, since the sediments generally contain adequate nutrients and light is abundant due to low algal turbidity. However, the macrophyte communities in less impaired lakes are often more diverse. In nutrient enriched or impaired lakes, extensive monospecific populations often develop.

Although macrophytes can be abundant even in oligotrophic lakes, macrophyte productivity tends to be higher in eutrophic lakes, probably because of the more organic and nutrient enriched sediments in productive lakes (Wetzel 1983). However, there are no direct relationships between nutrient concentrations in water or sediments and macrophyte abundance. This prevents their use as indicators of nutrient enrichment.

Relation to Beneficial Uses

Although macrophytes are normal ecological components of shallow lakes and littoral areas, they can cause impairment of beneficial uses such as swimming, boating, and fishing when they are present in high densities across much of the surface of shallow lakes. Macrophytes can also produce fish kills by causing pH extremes associated with carbon uptake and metabolism, and by causing ammonia toxicity from both high pH levels and ammonia release during senescence.

Because of their large biomass, macrophytes can be a large portion of the total nutrient pools in the water column of macrophyte dominated lakes. The internal nutrient loads from macrophyte decomposition can be larger than the external nutrient loads from the watershed, so they should be included in lake nutrient budgets. Nutrient loads occur throughout the growing season from the continual sloughing and decomposition of plant tissue during growth, as well as the senescence of early cohorts for multiple cohort species. The largest nutrient loads typically occur during the fall when major senescence occurs. Phytoplankton blooms commonly occur following senescence due to these large loads. Since macrophytes typically obtain most of their nutrient supply from the historical nutrient accumulation in the sediments, they can be a major source of internal loading that persists long after external loads from the watershed are curtailed.

Measurement

Macrophyte productivity can be evaluated by measuring the changes in biomass over time, or by direct measurements of macrophyte photosynthesis in the lake. Rough estimates can also sometimes be made from macrophyte harvest records. On an areal basis, submersed macrophyte productivity often equals or exceeds phytoplankton productivity, with annual values ranging from 50 to 1000 g C/m^2-yr. Emergent macrophyte productivity is much higher and is among the highest of any plant community (including terrestrial), with annual values ranging from 1500 to 4500 g C/m^2-yr. Daily macrophyte productivity varies substantially over the growing season in accordance with light and temperature. Macrophyte productivity is somewhat difficult to measure because of the extreme heterogeneity in macrophyte distribution and productivity, the large seasonal variations in productivity, and because the rooting systems often represent a large portion of the plant biomass and are not always sampled (Wetzel 1983).

Macrophyte abundance is typically expressed in terms of macrophyte density and percent coverage. Maximum seasonal densities of submerged macrophytes can be on the order of 500 g/m^2 dry weight in productive lakes (Wetzel 1983). Emergent and floating-leave macrophytes can have much larger densities. The nutrient pools within macrophytes can be estimated by multiplying the macrophyte density, percent coverage, lake area, and nutrient contents of the plants. Annual nutrient release rates from macrophyte decomposition can be estimated by multiplying the estimated nutrient pools at peak seasonal biomass by estimated annual biomass turnover rates. These values can be compared with the other nutrient pools and load sources to determine the importance of macrophytes on the nutrient status of the lake.

Data Availability

Macrophyte data are fairly sparse. Estimates of percent coverage are sometimes available. If macrophyte harvest programs are in place, annual harvest records are sometimes available.

Recommendation

Macrophytes are not recommended as an indicator to set nutrient criteria since there are no direct relationships between macrophyte abundance and the nutrient concentrations in water or sediments.

5.5.7 Zooplankton

Zooplankton are important components of lake ecosystems since trophically they represent the primary consumers. They are the primary consumers of phytoplankton and an important food source for many fish.The zooplankton communities in lakes are generally grouped into four major components: rotifers, cladocerans, copepods, and protozoans. Larval stages of insects, fish, and some stages of a few other invertebrates also sometimes occur in the zooplankton, but are not significant.

Although most rotifers are sessile and are associated with littoral communities, about 100 species are completely planktonic and are an important component of lake zooplankton. Most rotifers are omnivorous and feed on bacteria, small phytoplankton, and suspended detritus. However, a few species are predatory and feed on protozoa, smaller rotifers, and small crustaceans. Rotifers are often a major component of energy transfer and nutrient cycling in lakes (Wetzel, 1983).

Cladocerans are planktonic crustaceans that swim by means of large antennae, and feed primarily by filtering food particles through setae with complex leg movements. Although most filter-feeding cladocreans are herbivorous or omnivorous and feed on phytoplankton and organic detritus, some species are predatory and seize larger food items such as protozoa, rotifers, and small crustaceans. In addition, some littoral cladocerans are adapted for scraping up larger pieces of detrital material from littoral areas (Wetzel 1983).

Copepods consist of three major groups: calanoid, cyclopoid, and harpacticoid copepods. Calanoid copepods are almost entirely planktonic, while cyclopoid copepods include both planktonic and benthic species. Harpacticoid copepods are almost entirely littoral, inhabiting vegetation and sediments. Copepods swim by moving their legs, antennae, and mouth appendages, and they seize food items with their mouthparts. Calanoid copepods selectively feed on suspended food particles such as specific species of phytoplankton. Cyclopoid copepods are raptorial and seize food items with their mouthparts. Many species of cyclopoid copepods are carnivorous and feed on microcrustaceans, oligochaetes, and dipteran larvae, while other species are herbivorous and feed on phytoplankton. Harpacticoid copepods have mouthparts that are adapted for seizing and scraping food particles from sediments and littoral vegetation (Wetzel 1983).

The protozoans include ciliates, flagellates, amoebas, and other forms. Although they are generally less abundant in both biomass and numbers than the other zooplankton groups, during certain periods they can be a significant component of the overall zooplankton production, as well as a significant component of the biomass (Wetzel 1983, Pace and Orcutt 1981). Most protozoans feed on bacteria, algae, detritus, and other protozoans. Although they are generally aerobic, many protozoans can survive and grow well even when oxygen conditions are very low (Wetzel 1983).

Zooplankton productivity can be used as an indicator of lake nutrient enrichment since zooplankton productivity generally increases with phytoplankton productivity (Wetzel 1983). Zooplankton growth, feeding, and reproductive rates depend primarily on food supply and temperature (Winberg 1970, Wetzel 1983). The productivity of filter-feeding herbivorous zooplankton is generally much greater than the productivity of predatory zooplankton. Most of the zooplankton production is not directly utilized by predators, but instead enters the detrital pools. Only about 10 percent or less of the annual production of herbivorous zooplankton is typically incorporated into secondary consumers such as carnivorous zooplankton and fish (Wetzel 1983).

Zooplankton production rates vary substantially both seasonally and from year to year due to the large fluctuations in zooplankton population dynamics that occur with changes in food supply, temperature, predation pressures, and other environmental factors. The life cycles, reproductive characteristics, generation times, growth rates, feeding rates, feeding modes, and feeding preferences vary from species to species. Even within a species, zooplankton vary their feeding habits with food supply, and often change their feeding mode and food preferences throughout their life cycle. Both consumption and assimilation rates vary with the type, quality, size, and abundance of the food items, and these characteristics continually change as phytoplankton populations, bacterial populations, and detrital abundance change throughout the year. These factors make it difficult to quantify zooplankton production in the field.

Relation to Beneficial Uses

Zooplankton are important components of lake ecosystems since they are the primary consumers of phytoplankton, and therefore help keep phytoplankton populations under control. Zooplankton populations often increase during or immediately after phytoplankton blooms, exerting large grazing pressures on the phytoplankton. Zooplankton are also an important food source for many fish, including both planktivorous species and the larval or juvenile stages of predatory fish.

Measurement

Zooplankton productivity is somewhat difficult to measure. Production rates must generally be determined separately for each species or for groups of similar species since the reproduction rates, growth rates, and life cycles can be very different between species. Net zooplankton production is the sum of growth, gamete production, and molted exuviae minus maintenance losses from respiration and excretion. Zooplankton production is evaluated by estimating temporal changes in the numbers and biomass of individuals in a cohort over a life cycle. Since reproduction and recruitment are often fairly continuous, cohorts overlap, making it difficult to measure them separately (Wetzel 1983). This makes it necessary to estimate the birth rates, growth rates, and mortality rates of individuals over a full life cycle. Predation losses must also be estimated separately since predation can remove a significant portion of the production. Emigration and immigration effects on zooplankton populations are not generally important in lakes and can be ignored.

Data Availability

Zooplankton data are fairly sparse. Zooplankton are not typically measured in water quality sampling programs. They are generally measured only in ecological surveys. Zooplankton densities (numbers) or biomass are typically reported. Productivity is only measured in special studies.

Recommendation

Due to the paucity of data, zooplankton are not recommended to be included as an indicator to set nutrient criteria.

5.5.8 Macroinvertebrates

Macroinvertebrates are animals without backbones that are large enough to be seen with the naked eye. The lower size limit is arbitrary. The USGS has adopted a mesh size of 0.21 mm as the most suitable for sampling macroinvertebrates in flowing waters (Platte et al. 1983), while APHA (1989) defines macroinvertebrates as those invertebrates retained using a mesh size of 0.595 mm.

A wide variety of taxonomic groups are found in freshwater environments. These include annelids, crustaceans, nematodes, flatworms, mollusks, and insects. Benthic macroinvertebrates, which live either on or in the stream or lake bottom, are the group most amenable to systematic study.

Macroinvertebrates generally respond to nutrient enrichment indirectly since they are not very sensitive toxicologically to nutrient concentrations. Instead, macroinvertebrates respond to either the increase in food supply (algae) or to a decrease in dissolved oxygen levels caused by the decay of algal detritus.

Benthic macroinvertebrate communities are complex because they consist of many diverse species with different morphologies, feeding modes, reproductive characteristics, life cycles, and habitat requirements. Organisms from almost every animal phylum are represented (Wetzel 1983). Benthic animals are extremely heterogeneous in their distributions due to the heterogeneity of benthic substrates and food sources. In addition, many of the important benthic insects live in aquatic systems for only a portion of their life cycle. Benthic organisms consume many different types of food including organic detritus, bacteria, fungi, protozoans, phytoplankton, benthic algae, macrophytes, zooplankton, and other benthic invertebrates.

The benthic community can be used as an indicator of nutrient enrichment in several different ways. This includes changes in abundance and productivity, shifts in dominant species, changes in diversity, elimination of species less tolerant to low dissolved oxygen or high concentrations of decomposition products, and shifts in the locations of maximum abundance.

The productivity and abundance of most benthic organisms increases as nutrient enrichment increases due to the increase in food supply, which increases growth and reproduction rates. The productivity of benthic herbivores and detritivores is typically 5 to 10 times higher than benthic carnivores due to their greater growth efficiency (Wetzel 1983).

As lakes become more eutrophic, the composition of the dominant organisms in the benthic community changes, with decreases in chironomid larvae and increases in tubificid oligochaete worms. Tubificid oligochaetes are more tolerant to low oxygen concentrations and reach high densities when dissolved oxygen is very low since many other competing benthic animals and predators are excluded. Chironomid midge larvae are important predators of oligochaete worms (Wetzel 1983).

Benthic species diversity generally drops with increasing productivity as dissolved oxygen levels drop to levels that make portions of the water body uninhabitable by more sensitive species. In lakes, species diversity is typically higher in the littoral zone than in the offshore profundal zone due to the greater heterogeneity of the substrate (Wetzel 1983).

The location of the maximum number and biomass of benthic invertebrates may shift back and forth between the littoral and profundal zones as lakes become more productive. At low to moderate levels of productivity, benthic abundance is generally higher in the littoral zone. However, as productivity increases to the point that phytoplankton shading eliminates submerged macrovegetation, maximum benthic abundance may shift to the profundal zone. Further increases in productivity may reduce oxygen and increase organic decomposition products in the hypolimnion so that many of the profundal organisms are eliminated, shifting the zone of maximum abundance back to the littoral areas (Wetzel 1983).

Relation to Beneficial Uses

Macroinvertebrates play several major roles in aquatic ecosystems. They graze on periphyton and feed on terrestrial material that falls into the stream. Other invertebrates act as predators and filter feeders. Macroinvertebrates provide a major food source for most fish species (Gregory et al. 1987). Much of the ecological importance of macroinvertebrates stems from their position as an intermediate trophic level between microorganisms and fish (Hynes 1970).

Measurement

Benthic macroinvertebrates have several characteristics, which make them potentially useful as indicators of water quality:

- Many macroinvertebrates have either limited migration patterns or a sessile (attached) mode of life, and this makes them well suited for assessing site-specific impacts.
- Their life spans of several months to a few years allow them to be used as indicators of previous environmental conditions (Platts et al. 1983).
- Benthic macroinvertebrates are abundant in most streams and lakes;
- Sampling is relatively easy and inexpensive in terms of time and equipment (EPA 1989).
- The sensitivity of these organisms to habitat and water quality changes often makes them more effective indicators of stream impairment than chemical measurements (EPA 1990). In Ohio, for example, 36% of impaired stream segments detected with biosurveys could not be detected using chemical criteria alone (Ohio EPA 1988).

The disadvantages of monitoring macroinvertebrates include:

- A relatively high degree of variability within or between sites (Minshall and Andrews 1973). Much of the variability between samples is due to the highly heterogeneous distribution of invertebrates with depth, current speed, and substrate composition (Platts et al. 1993, Morin 1985).

- Local or regional variations in the organisms to stress (Winget and Mangum 1979).
- The need for specialized taxonomic expertise.
- The cost of processing samples containing numerous organisms.

A variety of sampling and data analysis techniques can be used to monitor macroinvertebrate communities. Some of the more common parameters include presence or absence data, functional feeding group analysis, and community parameters. Sample collection techniques can be equally as varied, ranging from the placement of uncolonized artificial substrates to kick nets, drift nets, and fixed-area substrate samplers.

Collection techniques can be classified as qualitative, semiquantitative, or quantitative (Platts et al. 1983). Qualitative techniques rely on indicator species or an evaluation of selected functional or taxonomic groups. Generally, the samples for qualitative evaluation are not collected on the basis of a specified area or collection effort, and this severely limits any numerical analysis.

Sampling techniques that use uniform substrates or a specified amount of collection effort (e.g., a 3-hour drift net sample or 50 sweeps with a dip net) are termed semiquantitative techniques. Data from these samples can be used for qualitative purposes such as the presence or absence of particular taxa, or for estimating population characteristics like diversity, total numbers, or biomass. The primary limitation of semiquantitative methods is that results are reported as a per sample basis rather than as a per unit area basis (Platts et al. 1983).

Quantitative techniques involve complete sampling in a specified area. The resulting density data are on an absolute basis (number of organisms per unit area), and this allows for a comparison of populations over space and time. These types of data can be used to estimate productivity as well as population characteristics.

Although qualitative techniques are typically quicker and easier than semiquantitative or quantitative procedures, they yield less specific information. This usually makes qualitative techniques less sensitive and less reliable and since a similar level of expertise is required to analyze the samples and interpret the results, it is logical that either semiquantitative or quantitative procedures should be used for most projects (Platts et al. 1983).

This range of sampling procedures indicates that a wide variety of sampling techniques have been developed to accommodate varying study objectives and locations. The composition of the substrate, water depth, and current velocity largely determines the most appropriate technique. The most common methods include various types of nets, substrate sampling techniques, and the placement and subsequent removal of artificial substrates (Greeson et al. 1977). Each technique has a different set of errors and bias, making comparisons of data from different sampling techniques difficult (Platts et al. 1983).

Artificial substrate samplers are useful in large rivers or wherever natural substrates cannot be effectively sampled (EPA 1989). The most common artificial substrate techniques make use of multiplate (Hester and Dendy 1962) or basket (Mason et al. 1973) samplers. Advantages and disadvantages of artificial substrates are discussed in Greeson et al. (1977), Rosenberg and Resh (1982), and EPA's Rapid Bioassessment Protocols report (1989). The most common criticism is that they do not provide a representative sample of the natural community. Advantages include lower sample variability and elimination of substrate differences between sites.

Drift nets are used to sample macroinvertebrates that have been dislodged or are migrating, and typically they are left in place for at least several hours. However, the nets can become clogged if they are not regularly cleared of debris and this can reduce the number of organisms captured in the nets. Drift net data are expressed as numbers and biomass of organisms per unit discharge (APHA 1989).

Dip nets are used to qualitatively collect organisms associated with backwater areas, nearshore areas, and deposits of organic debris. Collection techniques can be specified by area and effort in order to obtain semiquantitative data. In deep waters and in areas with fine substrates, a variety of grab samplers, such as Eckman or Peterson dredges, may prove most effective. In small streams, Surber (1937) and modified Hess (Waters and Knapp 1961, Jacobi 1978) samplers are most often used for quantitative sampling (Platts et al. 1983). Both of these samplers utilize a frame to delineate a specific area of stream bottom and a net to capture the benthic fauna as the substrate is disturbed to a depth of 5 or 10 cm.

Regardless of technique, sampling methods must take into account the time of year, number of samples per site, and habitat to be sampled. Significant changes in the invertebrate population occur during the year because of natural life-cycle processes (Minshall and Andrews 1973). Collecting samples in more than one season is preferable, but when this is not possible the optimal sampling seasonis the period when most macroinvertebrates are both large enough to be retained during sieving and sorting, and identifiable with the most confidence (EPA 1989).

A variety of community and population indices can be used to characterize benthic macroinvertebrates, although the choice will be somewhat constrained by the particular sampling method used to collect the sample. One useful approach is to divide the invertebrates into functional feeding groups such as shredders, collectors, scrapers, and predators (Cummins 1973). Changes in the relative abundance of different functional feeding groups can indicate habitat change. For example, an increase in the number of scrapers as compared to shredders suggests an increase in the production of attached algae due to a reduction in riparian canopy cover, an increase in stream width, or an increase in nutrient levels.

Some of the more commonly used community parameters include abundance, species richness, diversity indices, and biotic indices. Each of these parameters considers only a part of the overall invertebrate population characteristics, and each has certain drawbacks in terms of representing the complex assemblage of organisms present at any given site (Elliott 1977). It is therefore beneficial to use more than one community measure for assessing invertebrate populations.

Abundance can be expressed in absolute terms as the number of individuals per unit area present, or in relative terms as a percentage of total numbers. The absolute abundance is a useful indicator of the overall productivity at a site. Relative abundance values, such as percent contribution of the dominant taxon, indicate the community balance. Communities dominated by just a few taxa indicate environmental stress (EPA 1989).

Species richness generally refers to the total number of taxa present. The total number of taxa in specific orders (e.g., total number of mayflies, stoneflies, and caddisflies) is also an useful indicator (EPA 1989). Lent (1988) observed a high correlation between species richness and water quality in North Carolina. In Oregon, species richness showed good correlation with trout populations from high desert streams (MacDonald et al. 1991).

Diversity indices combine species richness and relative abundance. A variety of indices have been developed, with the Shannon-Wiener index probably being the most common (Platts, et al. 1983). The use of diversity indices for detecting environmental stress has been criticized because they:

- do not incorporate any trophic community structure
- exhibit considerable variation even in undisturbed sites
- may be insensitive to disturbance
- are insensitive to ecological differences between sites (Pielou 1975, Zand 1976)

Various biotic indices have been developed to capture the complexities of natural populations. The Biotic Condition Index (BCI) incorporates stream habitat, water quality, and environmental tolerances of aquatic insects (Winget and Mangum 1979). The BCI is based on the mean tolerance of the aquatic insects predicted for a site divided by the actual mean tolerances of the aquatic insects found on the site. This method has been used extensively by the Forest Service and Bureau of Land Management in the Western U.S.

EPA developed five rapid bioassessment protocols (RBPs) in 1989 in an effort to provide state governments with a cost effective integrated biological index (EPA 1989). Protocols I, II, and III use benthic macroinvertebrates to assess water quality impairment; protocols IV and V use fish.

RBP I relies upon the qualitative abundance of different macroinvertebrate taxa and professional judgment to determine whether water quality is impaired or unimpaired. It was designed to be used as a quick screening tool. RBP II is a more intensive and systematic procedure intended to distinguish among three categories of water quality (non-impaired, moderately impaired, and severely impaired). RBP III is a more detailed protocol for benthic macroinvertebrates, similar to RBP II, but requiring identification to the genus and species level.

In summary, aquatic macroinvertebrate monitoring is a useful tool for evaluating general water quality condition and the extent to which beneficial uses are impaired or supported. Biological measurements are often less expensive than detailed chemical analyses, as a trained entomologist can use aquatic invertebrate data to infer a great deal about the site under consideration. To be most effective and reliable, however, biological studies need to be integrated into a monitoring plan that includes both physical and chemical evaluations.

Data Availability

Macroinvertebrate data are fairly sparse. California is currently assessing several rivers and streams and quality data will be available in the future.

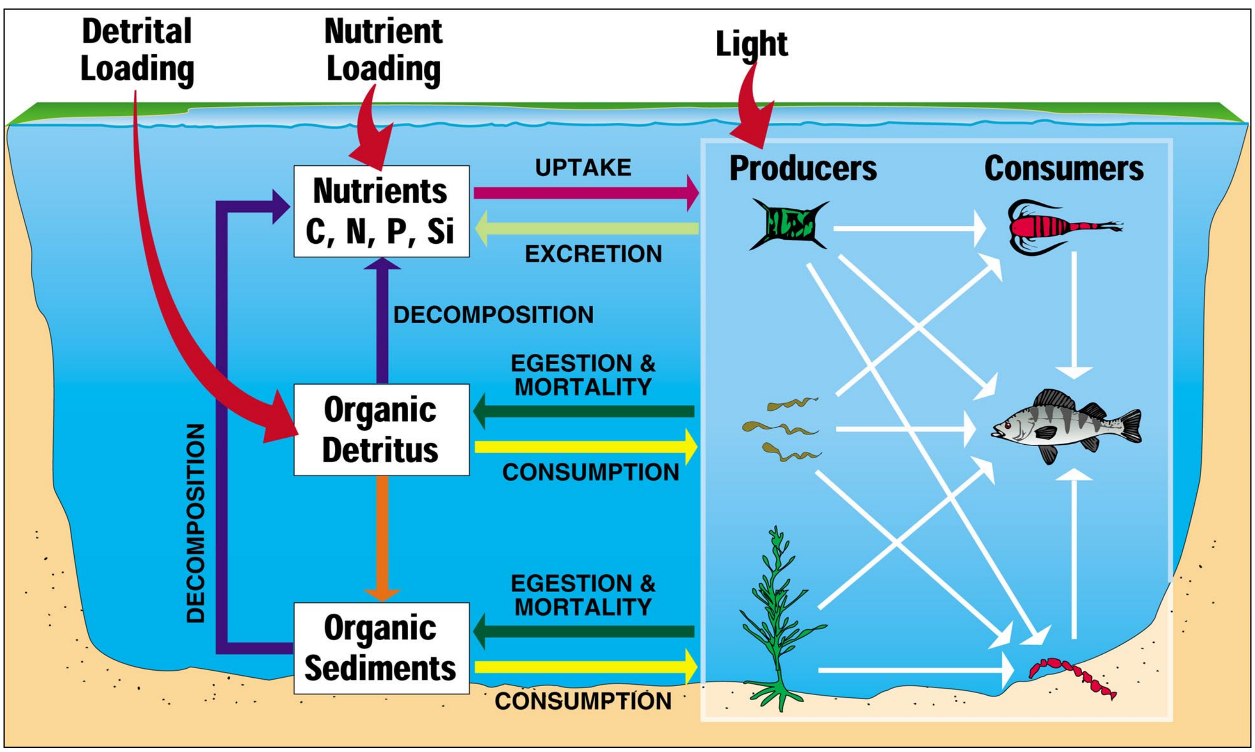
Recommendation

Due to the paucity of data, benthic macroinvertebrates are not recommended to be included as an indicator to set nutrient criteria.

5.5.9 Fish

Fish are useful indicators of water quality and the overall health of aquatic ecosystems since they represent the upper trophic levels and are therefore dependant on suitable water quality conditions for all trophic levels below them. Because of their long life spans, they integrate the effects of water quality, food supply, and habitat conditions over long time periods. Fish occupy different portions of the food web and have different food, water quality, and habitat requirements as they mature from larvae to adults. Certain life stages of fish are often the most sensitive organisms to environmental stresses. The relationship between nutrients and fish productivity is illustrated in Figure 5-6.

Figure 5-6. Relationship Between Nutrients and Fish Productivity



Nutrient enrichment influences fish through both its effects on food supply and its effects on fish habitat. Habitat effects involve many aspects of water quality including low dissolved oxygen, increased turbidity from phytoplankton, pH extremes, elevated levels of unionized ammonia and potentially ammonia toxicity, extensive periphyton growth, macrophyte infestation of lakes, and excessive accumulation of organic debris on the sediments. These effects may make the water uninhabitable for certain species, cause fish toxicity and fish kills, and interfere with reproduction by destroying suitable spawning habitat or increasing mortality of sensitive lower life stages.

Fish productivity generally increases along with lower trophic level productivity as lakes become more nutrient enriched. As with other consumer organisms, this results from increased growth and reproduction rates associated with the increased food supply. Production rates, expressed as ratios of annual production to fish biomass, are generally higher for fish with short life cycles than fish with longer life cycles (Wetzel, 1983). Production rates are typically highest in planktivorous fish and lowest in carnivorous fish, with benthic feeders in between (Wetzel, 1983).

However, some fish populations may start to decline as eutrophication becomes more severe and the habitat becomes unsuitable for sensitive species with more stringent water quality or habitat requirements. This produces changes in the composition of the fish community, with different types of species becoming dominant as lakes become more productive. For example, in northern temperate lakes, salmonids typically dominate in less productive lakes with oxygenated hypolimnions (Oglesby et al., 1987; EPA et al., 2000). However, when productivity increases enough to produce anoxic hypolimnions, salmonids may disappear and be replaced by percids (Oglesby et al., 1987; EPA et al., 2000). As eutrophication increases further, centrarchids may replace percids, and eventually rough fish such as carp and bullheads may be dominant at very high nutrient levels (Oglesby et al., 1987; EPA et al., 2000).

Fish are a useful surrogate or integrator of a variety of physical and biological factors. Some of the factors necessary to sustain a particular fish population include the following:

- adequate streamflow
- sufficient spawning habitat
- sufficient rearing habitat
- appropriate food sources at different life stages
- proper environmental conditions (temperature, dissolved oxygen, and turbidity)

The use of fish for monitoring presents many parallels to algae and macroinvertebrate monitoring. As such, monitoring can be based on the presence or absence of particular species, or community parameters such as productivity, density, and diversity (Hendricks et al. 1980). The conceptual advantages and disadvantages of these different parameters are briefly discussed in the following sections, as are the specific techniques, which pertain to the use of fish for water quality monitoring.

Relationship to Beneficial Uses

Fisheries are a very important designated use in fresh, estuarine and salt waters. Sport and commercial fishing are worth hundreds of millions of dollars each annually. In many rural areas, sport and commercial fishing are major components of the local economy. Fish also have important economic, cultural, and subsistence values for many native Americans.

Ecologically, fish are important because they represent the higher trophic levels in streams and lakes. Although fish are primarily predators of macroinvertebrates, their role in the food web varies by species and age. At certain times, fish are an important food source for terrestrial fauna such as bears, raptors, and raccoons. Because fish are high in the aquatic food web, they can serve as excellent indicators of the overall physical, chemical, and biological condition of streams.

Measurement

A wide variety of techniques have been employed to assess changes in number and condition of fish.

Fish population counts or estimates are probably the most common parameter. For anadromous fish, counts are most often made of the number of fish returning to spawn or the number of fish carcasses following spawning. One can also count the number of outmigrating juveniles (smolts) from a particular river, but this requires the regular use and maintenance of traps, nets, and weirs. Counts of outmigrating young provide a more specific indication of spawning and rearing habitat productivity than counts of resident fish or returning adults.

Transient or resident populations within a stream reach can be counted by a variety of means (Platts et al. 1983). Electrofishing is the most common field technique (EPA 1989), although it can be difficult to obtain the proper permits if rare or endangered salmonid species are expected to occur in the proposed stream reach. Another technique is direct observation by snorkeling (Platts et al. 1983).

Accurate counts of returning adult and departing young anadromous fish species can be obtained by placing nets or weir traps on the stream of interest. These capture all migrating fish, but complete counts may require several months. To prevent fish mortality, the captured fish must be regularly removed, and individuals are often counted and weighed at this time.

Estimates of the number of salmonid spawning pairs can be obtained by counting the spawning nests (redds). Groundbased counts are usually more accurate and less costly, and they are the only appropriate technique for smaller forested streams. Aerial surveys may be preferable on large rivers, but these are usually less accurate (Bevan 1961).

Species presence or absence, species richness, and diversity indices all have been used as relative or qualitative indicators of water quality (Warren 1971, Cairns et al. 1973, Langford and Howells 1977). The limitations of these parameters are the same as those for the algae and macroinvertebrate and have already been discussed. In evaluating these measurements, consideration must be given to the biogeographic region, season of measurement, and stream size (Karr 1981). Generally, fewer species of fish occur in undisturbed streams and lakes of the Pacific Northwest than in the Midwest or Southeast, and this hampers the use of diversity or richness measures as indicators of water quality. However, the number of native species may be a sensitive measure of the deterioration of pools and other habitats (Miller et al. 1988).

Over the last several years, there have been several attempts to develop more comprehensive and meaningful measures of fish communities. The index of well being (IWB) incorporates two diversity and two abundance estimates with approximately equal weight (Gammon 1980). The index of biotic integrity (IBI) is obtained by weighting and summing 12 individual measures or metrics (Karr 1981). The metrics were selected on the basis of experience in the Midwest, and they include parameters such as total number of species, the number of species tolerant and intolerant to poor water quality, several trophic measures, and several indicators of condition (Karr 1981; Angermeier and Karr 1986). Since this procedure was developed only for Midwestern streams, substantial alterations must be made in order to apply it to streams in the west (Hughes and Gammon 1987, Miller et al. 1988).

The IBI is the basis for EPA's RBP V. As with RBP II and RBP III, the habitat quality and IBI for the site under study is compared to the habitat quality and IBI for an unimpaired reference station. Concurrent collection of water quality data is also recommended (EPA 1989).

Some advantages of using fish for monitoring water quality are as follows:

- Their mobility and relatively long life span allows them to indicate broad-scale and long-term habitat conditions.
- Their higher trophic position means that they can be used as an integrator of changes in the lower trophic levels.
- They are relatively easy to collect and identify in the field.
- The habitat requirements of many species are relatively well known (EPA 1989)

The disadvantages include:

- The difficulty of obtaining a representative sample or an accurate estimate of the population.
- The variety of extraneous factors that can affect fish populations during different life history stages (fishing pressure, fish stocking, predation, disease) (Hellawell 1977, Hocutt 1981).
- The mobility and limited residence time of anadromous species in freshwater.

Fish represent an important beneficial use of most waters. Fish populations can be economically and culturally important, and they often have a high public profile. The most stringent constraints on water quality often stem from the need to protect coldwater fisheries. The relative absence of certain species from a suitable waterbody can be a quick and important indication of serious impairment. However, the quantitative monitoring of fish populations, although of critical importance for fisheries management, often is of limited or uncertain value for water quality monitoring.

The mobility, multi-year life span, ecological role, and numerous extraneous factors that can affect their population limit the value of fish monitoring. High mobility means that it is difficult to obtain an accurate population estimate, and this limits the likelihood of detecting a statistically significant change. Their multi-year life span may be an advantage in that the number of fish in a certain age group or size class integrates past conditions, but it is also a disadvantage because the number of fish may not provide useful data on current conditions. The position of fish at the top of the food web means that they are affected by any fluctuation at other trophic levels, and this may make it difficult to identify the cause of an observed change. Similarly, interspecific competition is often very important, and this may require an entire set of species to be monitored rather than a single population (MacDonald et al. 1991). Predation is particularly important for alevins and juveniles.

Data Availability

Data resources for stream fish populations include the California Department of Fish and Game (CDFG), U.S. Fish and Wildlife Service (USFWS), and USGS. While there are many sources of data there is little consistency in the data collected or the geographic areas and temporal periods covered by fish surveys.

Recommendation

Due to the paucity of data, fish are not recommended to be included as an indicator to set nutrient criteria.

5.6 ANALYTICAL METHODS

There is rarely only a single way to quantify a given parameter. In fact, oftentimes, there are several. It is always best and easiest, to use data that were analyzed using identical methods since methods differ in accuracy, precision, and detection limits. While we recommend concentrating on a single analytical method fro each parameter of interest, the selection of a particular “best” method may result in too few observations. In light of this, we propose using a step-wise approach in order of most preferred to least preferred:

- Use analytical method that provides the best level of accuracy, precision, and detection limits
- Select the most frequently used analytical method, and lastly
- Select data using similar methods

Table 5-1 provides a listing of several potential parameters and commonly used methods

Table 5-1. Potential Types of Environmental Data to be Collected

Data Type	Analytical Method(s)
Location using GPS or map (latitude/longitude or UTM coordinates)	EMAP, section 4.1.1, pages 57-60
Photodocumentation (visual record of sampling sites)	EMAP, section 4.1.4, page 61
EMAP habitat characterization (measurement and observations on transects)	EMAP, pages 107, 118, 123, 137-138, 141, 215-224; Rapid Bioassessment Protocol, A-7, A-9 (Barbour, et. al., 1999)
Flow	USGS gauging station or manual meter
Dissolved Oxygen	EPA 360.1
pH	EPA 9040/9045
Temperature	Meter
Conductivity	EPA 120.1
Ammonia	EPA 350.1/350.3
Nitrate + Nitrite	APHA 4500-NO3-E
Total Nitrogen	EPA 353.2
Ortho-Phosphate	EPA 365.2
Total Phosphorus	EPA 365.2
Turbidity	EPA 180.1
Copper (micro-nutrient)	EPA 6010
Selenium (micro-nutrient)	EPA 6010
Zinc (micro-nutrient)	EPA 6010
Periphyton	
– Chlorophyll, phaeophytin, and ash-free dry weight	Holm-Hansen and Riemann (1978); Peck, et al. (2000); APHA (1995)
– Algal C:N	American Soc. of Agron., Inc. (1996)
– Algal (organic) P	Solorzano and Sharp (1980)
– Community Composition	Peck, et al., (2000); USEPA (1997); Wetzel (1979); APHA (1991)
Benthic Macroinvertebrates	USEPA (1997); Barbour, et al. (1999)
Zooplankton	
Fish	Barbour, et al. (1999)

6.0 WORK PLAN TASK AREAS OUTLINES

The preferred approach to developing nutrient criteria for this region is to further refine regional and waterbody categories and to supplement the existing reference distribution approach with a work plan that evaluates effect-based criteria. The challenge over the next two years is to collect sufficient information to support analyses in three areas. These areas are briefly described below. These analyses will be undertaken in the context of a nutrient criteria development process that is defined by the following questions:

- **What nutrient criteria development approach will the work plan recommend?**

The work plan will describe an approach to develop numeric nutrient criteria based on evaluation of distributional and effects-based data. Criteria will be developed for all subecoregions and two waterbody types (i.e., lakes/reservoirs and rivers/streams). Nutrient criteria will be developed for the protection of the most sensitive beneficial uses. Physical classification and stratification criteria will be used to ensure that waterbodies within a given category respond in a similar manner to nutrient inputs.

- **What forum will be used to coordinate the nutrient criteria development activities?**

The RTAG will serve as the general coordinating forum for Region IX stakeholders. The STRTAG will closely collaborate with RTAG and will provide the primary direction for the development of nutrient criteria for California. Arizona will participate in the RTAG process and has its own nutrient criteria development initiative under way. Nevada will cooperate and contribute to the RTAG and STRTAG process. Contractors, acting on the direction of the RTAG and STRTAG, will be responsible for data collection and analysis activities. Contractors will develop technical findings and recommendations for RTA and STRTAG consideration. Hawaii is also participating in the RTAG process. However, since ecoregional guidance has yet to be developed for Hawaii, there will be no nutrient criteria developed for the state. The RTAG will continue to provide some support to Hawaii on data analysis tasks.

- **How will waterbodies within the region be grouped?**

The approach for grouping waterbodies within the region is described in Section 4 of this document. Different stratification criteria may be used for different ecoregions. Beneficial uses will be a primary stratification criteria used in all ecoregions.

- **How will waterbodies within the region be assigned a priority ranking for nutrient criteria development?**

RTAG and STRTAG participants will develop lists of ecoregions or waters that should be addressed first because of downstream loading concerns, pending nutrient management plans, anticipated TMDLs, available data, numbers of affected waterbodies, or other criteria. A priority ranking rationale will be provided with the schedule that will be included in the final work plan.

- **What data will be used in regional nutrient criteria development?**

The data collection strategy and potential sources for regional nutrient criteria development are described later in this section. The RTAG and STRTAG process will rely on existing data, data to be collected by the SWAMP (California), other participating agencies and organizations, site-specific studies that have been reviewed for methods consistency, and data generated from computer modeling scenarios.

- **How will data for the region be analyzed?**

Preliminary data analysis procedures are described in this report. The approach relies on development and refinement of regional distribution data (based on updated classification categories), computer model simulated background scenarios; modeling scenarios evaluating representative system responses; and a significant amount of verification using existing site-specific studies.

- **What parameters will be used in the development of regional nutrient criteria?**

Section 5 includes a detailed discussion of both causal and response candidate parameters. Different parameters may be selected for ecoregions. Initially the parameters may be limited to those for which data is currently available. However, the work plan could lay the foundation for refining nutrient criteria though recommended monitoring for parameters that are more clearly linked to waterbody response to nutrient inputs.

- **What administrative procedures will be used in the region to develop and adopt nutrient criteria?**

The RTAG and STRTAG has been the primary forum for development of criteria for California and Nevada. Arizona has a long-standing program for nutrient criteria development. The work plan will describe the degree to which there will be collaboration (e.g., sharing data and information) between these three states. Currently no guidance exists for Hawaii. The RTAG will continue to provide some support to Hawaii on data analysis tasks. Once the RTAG and STRTAG has approved the preliminary nutrient criteria recommendations (in 2004) each state will submit the recommended values to their individual states and regional boards for approval.

➢ What provisions have been made for public and stakeholder involvement in the regional criteria process?

The RTAG and STRTAG have provided a forum for the public, interested organizations, and the states to participate in the nutrient criteria development process. Key information and meeting summaries are posted on a publicly-accessible website maintained by a support contractor. The RTAG and STRTAG have defined roles for participants that allow for public and stakeholder input while preserving the regulatory integrity of the process. It is anticipated that this work group format will be continued through the duration of the nutrient criteria development process.

➢ Who will be involved in the decision-making?

Each state and regional board will make the final decision regarding the RTAG and STRTAG recommendations, using their own standard administrative process and guidelines for the formal adoption of nutrient criteria. During the development process, the public and stakeholders have input to the discussions and process. However, only state water quality agency representatives have voting rights on decisions regarding technical direction to the support contractor.

➢ How will outside expertise be used in the nutrient criteria development process?

University representatives are serving as technical advisors to RTAG and STRTAG. In addition several agencies and organizations that conduct research on nutrients and their environmental impact are participating in the process including (but not limited to) USGS, CDFG, Southern California Coastal Research Project, and individual municipalities.

➢ What are the major milestones and schedule of completion of the regional nutrient criteria?

The major milestones for the regional nutrient criteria development process have been identified in Section 1of this document. It is anticipated that interim findings and recommendations will be available by June 2004.

The following sections provide a preliminary description of the activities that will be undertaken when the regional work plan has received RTAG and STRTAG approval.

6.1 STATISTICAL TREE-BASED APPROACHES TO IDENTIFY FACTORS MOST SIGNIFICANT IN DETERMINING NUTRIENT CONCENTRATIONS IN WATER BODIES WITHIN A WATERSHED

The goal of the EPA nutrient criteria development process is to find the most appropriate nutrient levels for reference conditions in water bodies. It is generally understood that reference levels of nutrients will be different for different regions of a state or EPA region as well as for different water body characteristics. Therefore, when numeric nutrient criteria are defined, we must also define the best regionalization and water body classification that goes with the numeric criterion.

There are two ways to select the most appropriate stratification for water bodies in a state: the first is to do it based on the scientific judgment of the RTAG members, and the second is to use a group of statistical classification techniques to select the best stratification given the nutrient data. Possibly the best approach is to combine the statistical classification with the human element, i.e., first obtain the stratification using different statistical techniques, and then refine the stratification based on the knowledge of experts in the RTAG.

For the purpose of this study we propose to use a method called Classification and Regression Trees (CART), that can be used with both numeric and non-numeric classification data. Thus, we can associate each sampling station and its nutrient level with various land use and geology-related parameters. A station could be identified by the watershed it was in, the geology of the watershed, the slope of the streambed, the flow rate in the stream, the season, the land-use in the watershed, and so on. Note that all of these data may not be available for the reference streams that we have studied in California, and that significant effort may have to be made to make the data set complete. When finished, the complete data set would consist of the variable to be predicted (i.e., the nutrient species of concern), and a series of independent variables that we believe can be used to estimate the predicted variable. This is not unlike the data format that would be used for multiple regression, except that CART allows us to use a mixture of numeric and category-type variables, for example the use of descriptive terms for the geology of the watershed. When the CART algorithm is applied to the data, we can find out what values and ranges of the independent variables are best able to predict the dependent variable. The goal of the CART algorithm is to divide a large data set into smaller and smaller subsets of data, that are more similar to each other than the full data set, and that can be associated with certain ranges of the independent variables. To consider a hypothetical example, high phosphorus levels in reference streams may be associated with a certain type of geology, steep slope, and the absence of forest cover. Such a stratification does not replace the professional expertise of the RTAG members, but provides a subset of the criteria that can be refined further.

6.2 NUTRIENT MODELING SCENARIOS

Computer simulation models will be used to evaluate the nutrient response of streams and lakes to the key variables in the classification hierarchy described above. The modeling will be used to verify the results of the statistical analyses, and also to fill in data gaps and extend the analysis to a full range of conditions that may not be fully represented in the database. The modeling will allow us to systematically explore the effects of varying one lake or stream characteristic at a time while everything else remains constant. This will help determine which classification parameters have the most impact on nutrient and algal conditions, and how far the nutrient response can be expected to change with variations in the classification parameters. The effects of key watershed characteristics will also be evaluated with the models.

Three types of models may be used in the analyses: watershed models, stream models, and lake models. Watershed models will be used to estimate background nutrient loads to the streams and lakes. These models typically predict loads or concentrations at the downstream end of the watershed. Therefore, they also predict the nutrient concentrations in the streams at the watershed outlet. Nutrient levels at different reaches of a river network can be evaluated by dividing the overall watershed into several subwatersheds and calculating the results separately. Therefore, a separate stream model may not be necessary for the river and stream analyses. However, stream models will be evaluated for possible application to specific issues such as periphyton growth that are not typically included in watershed models. Separate lake models will be necessary for the lake analyses since lakes have much longer residence times than streams, making internal cycling processes more important. However, the nutrient loads calculated from the watershed model will be used as input to the lake models.

The models will be applied to generic watersheds, streams, and lakes representing each of the 13 ecoregions in California, Nevada, and Arizona. Since we are focusing on generic analyses, models will be selected that do not require a lot of site-specific data and calibration efforts. This will constrain us to models that have default parameters that have been established from the analysis of many watersheds, rivers, or lakes, preferably to conditions representative of California, Nevada, and Arizona. The major modeling options are described below.

6.2.1 Watershed Modeling

Watershed models for estimating nutrient loads to streams and lakes can range from simple empirical methods such as export coefficients to very complex simulation models with detailed process formulations and extensive data and calibration requirements. Several approaches will be evaluated for this study. An application plan will be developed that balances the desire for accurate predictions with the need to analyze numerous scenarios generically without requiring extensive data and calibration. The project resources will also be an important consideration, since the more detailed models require much more effort and would therefore be restricted to fewer scenarios.

A typical watershed scenario will first be established for each ecoregion. This will include the average distribution of land uses, vegetation covers, soil characteristics, topography, climates, and background nutrient concentrations in each ecoregion. A few additional scenarios may also be defined for each ecoregion to represent the typical range of watershed conditions that can be expected. For example, scenarios may be developed for different sizes of watersheds, different land uses, and to evaluate the differences between upland watersheds with low order streams and larger watersheds feeding downstream higher order streams. The major watershed modeling options are described below.

Approaches for Watershed Modeling

The *Compendium of Tools for Watershed Assessment and TMDL Development* (Shoemaker et al., 1997) reviews the available models and divides them into three major levels. These levels are distinguished by the complexity of the models and the corresponding amounts of data and resources necessary to apply them. The three levels are 1) simple methods, 2) mid-range models, and 3) detailed models. Within each level, watershed models vary in the particular types of water quality constituents they simulate, and whether they are applicable to rural undeveloped areas, urban areas, or both. For this study, since we are interested in simulating reference conditions in relatively unimpacted areas, the model must be capable of simulating nutrients in watersheds with primarily undeveloped landscapes, but also capable of including urbanized areas.

Simple methods provide rough estimates of long-term average pollutant loads from different land use types. They are empirically based, and include things like export coefficients and regression relationships. Export coefficients give the long-term average pollutant yield per unit area (e.g., lb N/acre/year) for a particular land use category based on information collected from specific studies in the literature. Regression relationships predict pollutant loads using empirical equations that depend on variables such as land use type, drainage area, impervious fraction, annual rainfall, and air temperature. Regression equation methods may give better estimates than simple export coefficients since they consider basic watershed and climatic characteristics, but the regression coefficients are based on data from specific areas that may be different than the study area. Although the simple methods are easy to apply and require minimal data, they are not as accurate as more detailed approaches for predicting site-specific nutrient loading, and they do not consider seasonal and year-to-year variability.

Mid-range models consider both spatial and temporal variability in pollutant loading processes, and may use mechanistic, but simplified, formulations for some of the hydrologic, sediment transport, and pollutant generation and transport processes. For example, they may simulate the water budget using daily meteorological data, but may not consider transport between model spatial segments or along waterways within the watershed. Sediment transport may be simulated using the Universal Soil Loss Equation and sediment delivery ratios, while more mechanistic detailed models may simulate soil detachment, transport, and deposition as functions of overland flow characteristics. Mid-range models predict dissolved and particulate nutrient loads using hydrologic information and simple nutrient accumulation and removal relationships for different land use types. These relationships consider factors such as vegetation and soil characteristics, and types and intensity of use. However, these models do not consider the different chemical forms of the nutrients, or transformations between forms that are simulated in some of the detailed models. Mid-range models are better than the simple approaches since they consider the effects of hydrologic variability, incorporate site-specific land characteristics in the load estimates, provide reasonably accurate estimates with readily available data, and do not require the extensive data and resources necessary for the detailed models.

Six mid-range models are identified in the *Compendium of Tools for Watershed Assessment and TMDL Development* (Shoemaker et al., 1997). These are GWLF, SITEMAP, P8-UCM, Auto-Ql, AGNPS, and SLAMM. Three of the models – P8-UCM, Auto-Ql, and SLAMM – were developed for urban areas only, so they are not appropriate for this study since our focus is on estimating background nutrient loads from undeveloped areas. AGNPS was developed for rural areas, but it predicts loads from a single event (single storm), rather than continuously throughout the year as with the other models. A new continuous version of the model, called AnnAGNPS, is currently under development, but it would fall into the detailed model category since it predicts spatial variability in transport processes (as does AGNPS) and includes many more detailed process formulations than the original model. GWLF and SITEMAP are the remaining candidates in the mid-level category, since they consider both rural and urban areas, as well as point sources. GWLF is preferred over SITEMAP since it predicts nutrient loads associated with both groundwater baseflows and surface runoff (SITEMAP does not consider baseflows), and since it considers both dissolved and particulate nutrient loads separately (SITEMAP only considers total nutrient loads and does not simulate sediment transport). In addition, GWLF uses daily rainfall data, while SITEMAP requires hourly data. GWLF includes nutrient build-up and washoff formulations for urban areas similar to those used in more detailed models such as SWMM.

Simple GIS models based on stormwater monitoring data can also be classified as mid-range models. These models calculate runoff volumes based on precipitation rates, land use areas, and runoff coefficients determined from the impervious fractions of each land use. The runoff volumes are combined with average pollutant concentrations (EMCs) measured from different land use categories during storms to calculate the pollutant loads. This approach does not consider variations in runoff due to soil moisture changes, and does not include groundwater baseflow contributions during the dry season.

Detailed models are the most complex and typically include state-of-the-art representations of watershed hydrology, sediment transport, and nutrient generation, cycling, and transport processes. They also consider both temporal and spatial variability in these processes throughout the watershed. This includes flow routing, and the transport and nutrient transformation processes that occur as nutrients are carried from the upland to downstream portions of the watershed. Different chemical forms of nitrogen (particulate N, organic N, ammonia, nitrate, nitrite) and phosphorus (particulate P, organic P, phosphate) may be simulated in soils and waterways, along with transformation processes between these forms such as decomposition, mineralization, nitrification, denitrification, adsorption/desorption, and plant uptake and release. However, since so many different processes are simulated, these models require extensive amounts of data to set up and calibrate, and a considerable amount of time, resources, and expertise to apply properly. Many of these models have options for simpler approaches to predicting pollutant loading processes during storms, for example the use of generic build-up and washoff functions instead of simulating detailed nutrient cycling processes in the soils and vegetation. The detailed model category includes models such as HSPF (and BASINS NPSM), SWAT/SWRRBWQ, AnnAGNPS, ANSWERS, SWMM, STORM, and DR3M-QUAL. SWMM, STORM, and DR3M-QUAL were developed primarily for urban areas, so they are not appropriate for estimating natural background loads. ANSWERS was developed only for rural areas, and is not appropriate for urbanized portions of watersheds. Only HSPF, SWAT, and AnnAGNPS are applicable to both rural and urban areas. All three of these are capable of both long-term continuous simulations and single storm events. All of these models have extensive data requirements and typically require major site-specific calibration efforts. However, SWAT includes several GIS databases that contain default model parameters for different vegetation types, soil types, landscape topography, and climatic regimes, so it is the most amenable to generic applications. The other two models would be much more difficult to apply generically.

Watershed Modeling Plan

A watershed modeling strategy will be selected after an assessment of the available data, the number and types of scenarios to be evaluated, and the available project resources. The approach may involve a simple method, a mid-range model, a detailed model, or some combination of these techniques. The recommended mid-level model would be GWLF, and the recommended detailed model would be SWAT. A GIS model based on runoff coefficients and nutrient EMCs for different land uses and vegetation types may also be considered. Even if a detailed or mid-level modeling approach is selected, it may also be useful to perform a comparison check using a simplified approach such as export coefficients. Annual average export rates of phosphorus and nitrogen would be obtained for each land cover in each of the ecoregions using information obtained from the literature. The nutrient export rates, together with land use distribution and watershed size, would be used to calculate a series of loading scenarios for watersheds in each ecoregion.

6.2.2 River/Stream Modeling

As mentioned above, separate river and stream modeling are often not necessary to predict nutrient concentrations in streams since they can be estimated reasonably well from the results of the watershed modeling, particularly when the travel time through the reach is relatively short. Dynamic river models are useful for predicting nutrient transformations, biological uptake and cycling, and sediment exchange along major river systems, but these models typically require site-specific data and calibration and are therefore less useful for generic applications. River models are also useful for predicting things like periphyton and dissolved oxygen that are not included in watershed models.

River models will be evaluated for their applicability to generic applications for issues such as periphyton growth or diurnal dissolved oxygen depletion that are not addressed by the watershed models. Periphyton are not standard components of most river water quality models. Some models include simplistic representations of periphyton or periphyton effects on dissolved oxygen and nutrients, but they require site-specific calibration and are not capable of accurately estimating periphyton abundance for generic applications. More mechanistic biologically-based models are necessary for this purpose.

A few periphyton models have recently been developed, either as stand-alone models or as components of larger models. These include a periphyton model developed by Jim Brock and his colleagues as part of the Dynamic Stream Simulation and Assessment Model (DSSAMt), periphyton enhancements to the AQUATOX model developed by Richard Park, and the addition of periphyton to the WASP5 model. Statistically based models using regression analysis and other techniques have also been developed to predict periphyton abundance from nutrient concentrations and other stream characteristics. These include the models of Heiskary and Markus (2001), Niewenhuyse and Jones (1996), Biggs (2000), Dodds et al. (1997), and Welch et al. (1989). These and any other appropriate models will be evaluated for their applicability to generic river analyses.

6.2.3 Lake/Reservoir Modeling

A steady-state lake model will be used to predict the lake response to each load scenario established from the watershed modeling. For a given loading situation, the model will predict the average concentrations of total phosphorus, total nitrogen, and chlorophyll a. These analyses will be repeated for lakes of different size, shape, and hydrology using the classification variables derived from the database. Several analyses will be conducted for each combination of variables covering the range of conditions in the database. The steady-state model BATHTUB will be used for the lake modeling. The rationale for selecting this model is described below, followed by a brief description of the model.

The model will be run using default values for the nutrient sedimentation and algal parameters. These defaults are based on the statistical analysis of data from many different lakes from EPA’s National Eutrophication Survey and from Corps of Engineers reservoirs. The default parameters will allow the generic analysis of many different lakes with different size and hydrologic characteristics without requiring site-specific calibration to a specific lake. The model will be used to estimate the average nutrient and phytoplankton concentrations that could be expected for a given lake situation in each ecoregion. These results will show how the nutrient and phytoplankton levels can be expected to vary with size, hydrology, and watershed characteristics in each ecoregion.

General Approaches for Lake Modeling

Lake water quality models can be divided into two major categories—steady-state models and dynamic models. Steady-state models predict water quality conditions using average nutrient loads and flows. They are useful for predicting both long-term average conditions, and conditions under critical nutrient loads or flow regimes. They can be used repeatedly to predict the effects of different nutrient loads or flow rates, but they cannot predict temporal variability in water quality response resulting from to day-to-day variations in loads and flow conditions. Dynamic models are used for this purpose. Dynamic models predict temporal variations in water quality conditions due to variations in loads, transport processes, and chemical and biological processes within the lake. Lake models are further distinguished by their spatial resolution (single mixed compartment, 1-d, 2-d, or 3-d spatial grid), the water quality constituents simulated, the types and complexity of the processes modeled, the approach used to simulate sediment-water interactions, and the approach used for hydrodynamics and transport processes.

Dynamic models require much more data to set up and calibrate than steady-state models. Dynamic models typically use much more detailed process formulations that involve many model parameters that must be adjusted during calibration to match the temporal variations in the field data. This requires extensive time series of monitoring data, including simultaneous measurements of lake water quality, stream inflow water quality, and lake inflow and outflow rates. These data are required for calibration to characterize the dynamic response of the lake under different conditions. In contrast, steady-state models can be applied with much less data. They also typically include fewer processes and simpler process formulations, so they are easier to calibrate and apply in situations with limited field data.

A steady-state approach was selected for the lake modeling for several reasons. First, since we are applying the model to hypothetical lakes of different size and hydrology, there are no field data with which to setup and calibrate a detailed dynamic model. Second, we are interested in predicting the long-term average nutrient and phytoplankton concentrations in the lakes. In order to simulate seasonal dynamics, both detailed dynamic watershed and lake models would be required, both requiring fairly extensive field data. The steady-state lake model is consistent with simplified approaches for estimating nutrient loads from the watershed, but can also be used with more complex watershed modeling approaches. Nutrient targets for lakes are usually expressed in terms of either annual averages or growing season or summer averages, both of which can be determined with steady-state models.

Lake Model Selection

The *Compendium of Tools for Watershed Assessment and TMDL Development* (Shoemaker et al. 1997) identifies four steady-state models that are applicable to nutrient problems in lakes – BATHTUB, EUTROMOD, PHOSMOD, and the EPA Screening Procedures.

The EPA Screening Procedures are hand calculations that are useful for simplified analyses, but they do not include many of the capabilities found in the other computer-based models, so they were not considered for this study.

PHOSMOD is actually a simple long-term dynamic model rather than a steady-state model, but loading and flow conditions are assumed to remain constant over a given year (but can vary between years). The model predicts water column and sediment total phosphorus concentrations based on external loadings, outflows, and internal loss (settling) processes and exchange processes with the sediments. The model is used for whole lake analyses, but considers seasonal stratification and hypolimnetic oxygen effects on sediment phosphorus exchange. This model was not selected since it only simulates phosphorus, and does not consider phytoplankton, nitrogen, or any other water quality variables.

BATHTUB and EUTROMOD are both steady-state models that predict average growing season concentrations of phosphorus, nitrogen, phytoplankton (chlorophyll-a), and Secchi depth for different loading rates, outflow (flushing) rates, and reservoir morphometries.

EUTROMOD uses the simplest mass balance approach, which calculates phosphorus and nitrogen concentrations based on external load rates, lake outflow rates, and a first-order loss rate to represent all loss processes from the water column. Sediment recycling processes are not considered separately, but are incorporated in the overall loss rate. The first-order loss rates are estimated using regression relationships calculated from regional data (for several states) as functions of lake retention time, mean depth, and nutrient concentrations in inflows. Chlorophyll-a concentrations and Secchi depth are calculated from regional regression relationships as functions of the predicted nutrient concentrations, mean depth, and residence times. The significant regression variables vary between regions, and may include as few as one of the above variables. For example, the chlorophyll-a relationship for western states is based only on the predicted phosphorus concentration. Nitrogen limitation of algal growth is therefore not represented in this regional empirical model. EUTROMOD also has no way of representing macrophyte decomposition effects on nutrient concentrations.

BATHTUB is the most complex of the steady-state models and was selected over the other models. BATHTUB also uses mass balance models to predict phosphorus and nitrogen concentrations in the water column as functions of loading rates, outflow (flushing) loss rates, and internal loss rates. However, several options are provided to allow first-order, second-order, and other loss rate formulations that have been proposed from various nutrient loading models in the literature. Phytoplankton concentrations are estimated from more 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. Nutrient releases from sediments, from decomposing macrophytes, and atmospheric loads can be included in the model. Empirical relationships are provided to estimate Secchi depth and hypolimnetic oxygen depletion rates. Default values are provided for most of the model parameters based on extensive statistical analyses of data from EPA's National Eutrophication Survey and from data from many Corps of Engineers reservoirs. Spatial variability and transport processes such as advection and dispersion can also be simulated for large complex lakes, but may not be necessary for this study.

BATHTUB Description. BATHTUB (Walker, 1996) is a steady-state model developed for the Army Corps of Engineers 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.

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 larger lakes, particularly lakes with multiple side arms and tributary inflows, that have substantially different water quality in different portions of the lake. However, horizontal segmentation is not necessary, and the model can be applied as a whole lake model to predict average concentrations in large lakes. For small or moderately sized lakes, it is not appropriate to segment the model.

6.3 SYNTHESIS OF SITE-SPECIFIC STUDIES

Data from site-specific studies will be used to provide detailed information to supplement the distribution information. This type of data will provide a sense of "ground truthing" by answering questions such as "Given this range of potential nutrient criteria, can we be certain that Beneficial Uses would be supported?" A synthesis of several site-specific studies that compare causal and response variables to their impact on beneficial uses is provided in Appendix E of this report. This summary includes site-specific data for waterbodies both within and without EPA Region IX. The work plan will include a task that supplements this table with data from special studies performed on waterbodies inside EPA Region IX.

There are several sources of site-specific data available to us. These include, but are not limited to, completed and ongoing nutrient TMDLs; university studies; sanitary surveys; studies performed by local interest groups (e.g., Friends of the River); SWAMP's reference water study; and the California Bioassessment program.

Technical advisors will be used as resources to identify potential site-specific studies as well as providing technical guidance in developing the data synthesis.

6.3 DATA COLLECTION STRATEGY

Data will be collected using a 3-tiered, hierarchical approach and incorporate two types of data ("hard" as well as data acquired via modeling scenarios). These three tiers are (1) existing data, (2) data from on-going projects, and (3) data from special studies. Additionally, GIS will be used to acquire topographic data. Appendix F provides a list of potential sources of data. Hard data, in this study are defined, as those data that contains parameter values that have been, or will be, actually measured *in situ.* This is in contrast with those data that are generated via model simulations. GIS will be used extensively during this process to identify those physical topographical watershed parameters (e.g., gradient) that will be used to classify and categorize the waterbody. Each of these data collecting approaches is discussed in the following sections.

6.3.1 Existing Data

The time constraints of the nutrient criteria development program dictate that the majority of the data that will be used to set nutrient criteria will originate from data sets that are already in existence. This will require Tetra Tech staff contacting state and private sources of water, biological, and habitat quality data. Appendix F presents an extensive, yet not exhaustive, list of potential sources of data. (This list includes federal, tribal, state, and local government sources, as well academia, environmental, and private groups.)

The process of acquiring the data will include contacting the sources via personal and phone interviews that are followed up with site visits to collect the data. Additionally, on-line database searches as well as other, as of yet, unidentified resources will be actively pursued.

6.3.2 Ongoing Projects

The database generated from existing data will be supplemented with data currently being collected and compiled by other agencies. This would include, but is not limited to, data being collected by the State Water Resources Control Board's Surface Water Ambient Monitoring Program (SWAMP), the California Department of Fish & Game's Bioassessment Program, and USGS's NAWQA program. Other ongoing projects that are collecting water, biological, and habitat quality data will be actively pursued and, if possible, their datasets incorporated into the nutrient database.

6.3.3 Special Studies

Data from special studies will be used to fill in critical data gaps that are identified in the database. They may include, but are not limited to, collection of specific data types (e.g., benthic chlorophyll a or % periphyton coverage) or intense data collection from a specific waterbody type, that happens to be under-represented in the database as a whole.

7.0 REFERENCES

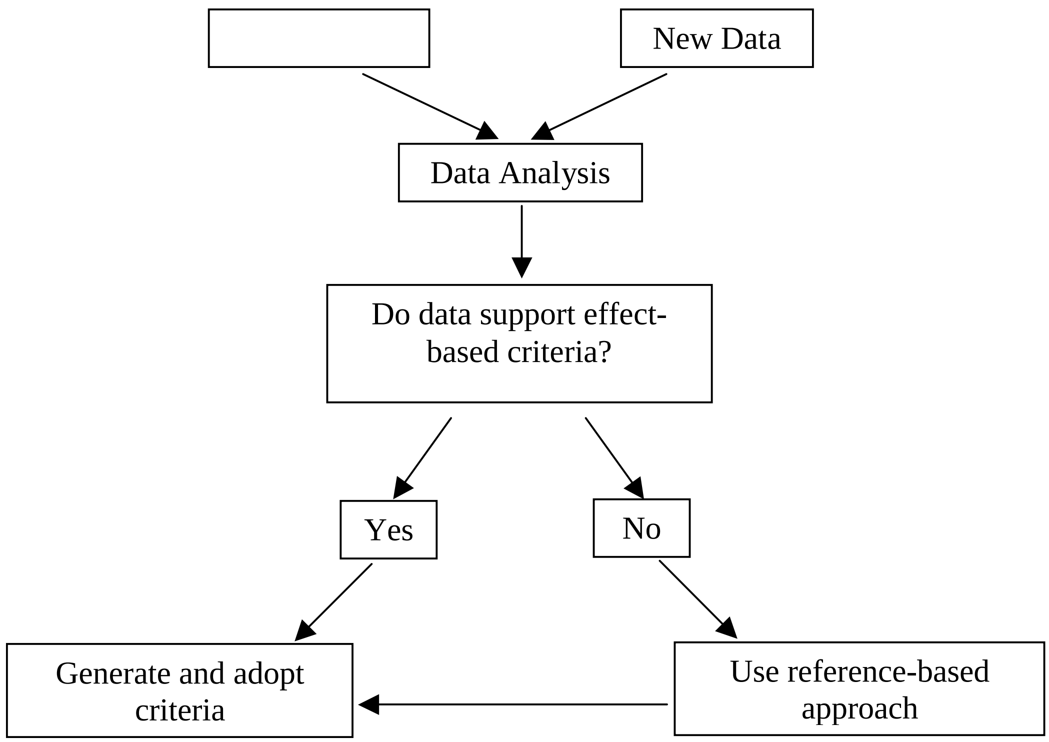
(to be provided at a later date)

APPENDIX A.

STATE X NUTRIENT CRITERIA DEVELOPMENT PLAN REVIEW

<u>State X Nutrient Criteria Development Plan Review:</u>

General Schematic of State X Proposed Plan:



Approach:

Preferred: Effect-based criteria derived by finding correlations between nutrient enrichment and negative changes in biological variables

Fall-back: Reference condition-based criteria

Form:

Not discussed. ***State X*** *needs to think about whether or not it will develop and adopt criteria for all causal variables. If not, the plan should direct some effort to generating data needed to support a decision not to adopt criteria for N.*

Process:

Regionalization: Not discussed. *Does* ***State X*** *have a regionalization system in place for interpreting biological data? Will this system be used for purposes of establishing nutrient reference conditions, if such* *reference conditions are needed or will **State X** use some other system?*

Classification: Not discussed. *Is **State X's** intent to rely on the water body classification system proposed by EPA (i.e., lakes and reservoirs; rivers and streams)? How will **State X** develop criteria for water body types that lack biological criteria and assessment data?*

Prioritization & Coverage: Not discussed. *Does **State X** have a plan for sequencing work on the different types of water bodies? Does **State X** intend to prioritize criteria development in some way and if so, how? What are the expected timeframes?*

Inventory of Existing Data: Summary provided, see attached Table.

Planned Data Collection: Summary provided, see attached Table.

Data Needs: General data needs were discussed. *Are any of the identified needs being addressed by **State X**, other States, or regional efforts?*

Assessing Progress: Not discussed. *At what intervals and/or points in the process will **State X** evaluate its progress towards the goal of adopting nutrient criteria against the milestones contained in its plan? What are the significant milestones?*

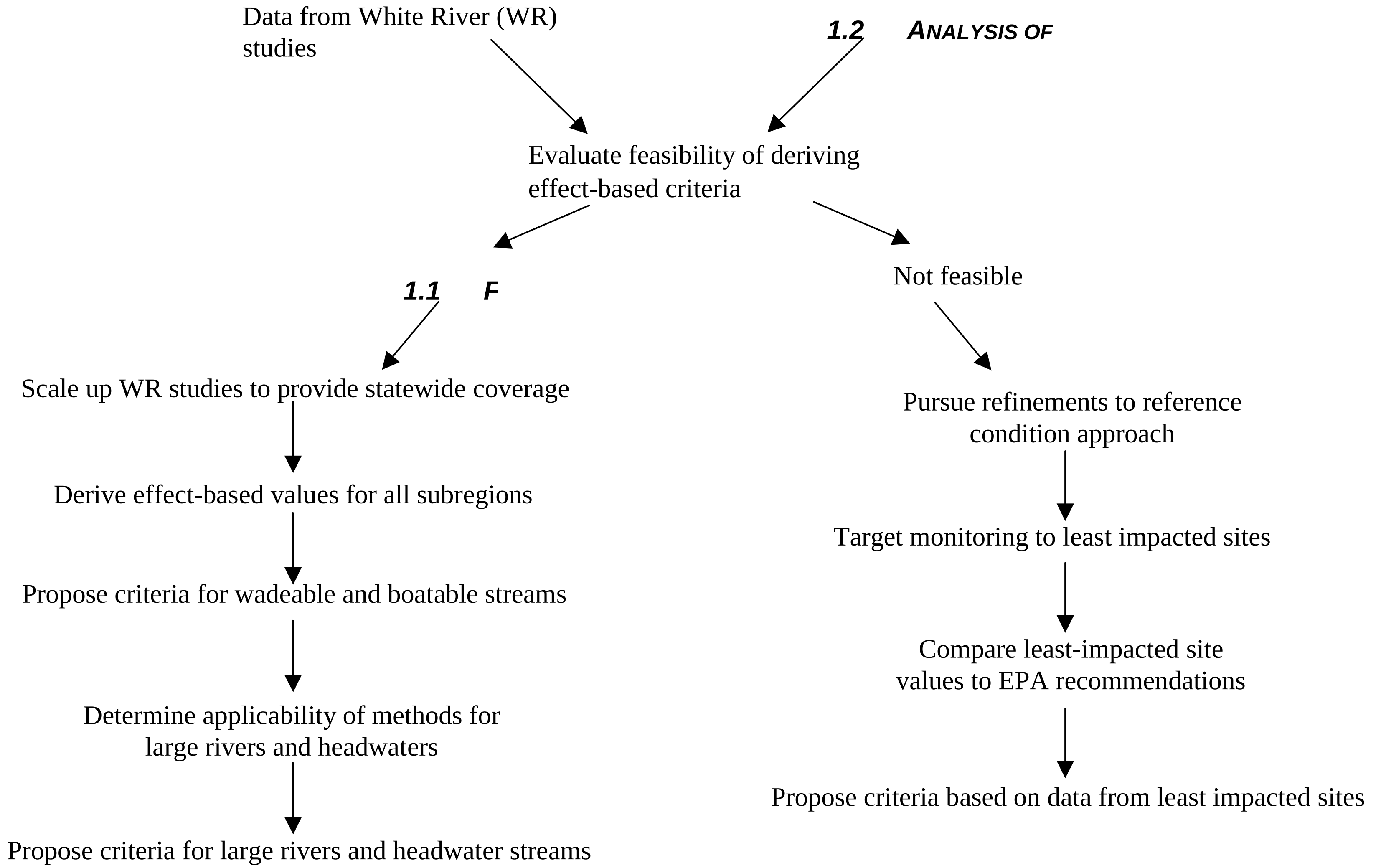
Plan Revisions: Not discussed. *How will **State X** revise the plan and how will notification of revisions be provided to EPA?*

Table 1. Data Assessment

Water Body Type	Available data	General Data Set Information	Causal variables		Response variables				
			Total N	Total P	Chl *a*	Turbidity	DO	Biology	
								fish	Inverts
Wadeable Streams	Existing Data	100 - 150 Fixed station datasets	X	X		X	X	1125 sites	704 sites
		Targeted stream surveys	?	?		X	X		
		Probabilistic stream survey sites	?	?		X	X		
		USGS Data (limited data set)	X	X	X	?	?	X (subset)	
		COE data (limited data set)	X	X					
	Planned Data Collection	USGS/ ***State X*** W fork of the White River Study (July, '02)	O	O	O	O (?)	O	O	
		USGS/ ***State X*** E fork of the White River Study (2003)	O	O	O	O (?)	O	O	O
Lakes and Reservoirs	Existing Data	***State X*** Lake monitoring program	X	X	X	X	X		Few
		Volunteer lake monitoring		X	X	X	X		
		COE (limited data set for reservoirs)	X	X					
	Planned Data Collection	Rigorous user perception study							
Headwater Streams	?	?	?	?	?	?	?	?	?
Large Rivers	?	?	?	?	?	?	?	?	?

<u>Recommendations for *State X* on the proposed nutrient plan based on EPA review:</u>

Rivers and Streams:



Possible activities and milestones for development of nutrient criteria for rivers and streams:

Year	Activities	
2002	1.	Conduct sampling on E. fork of the White River
	2.	Complete analysis of existing data for correlations between nutrient levels and biological response
2003	1.	Conduct sampling on W. fork of the White River and the Whitewater River
	2.	Complete analysis of data from the E. fork of the White River
2004	1.	Complete analysis of data from the W. fork of the White River and Whitewater River
	2.	Make a preliminary decision of whether P, N or both are needed as components of the criterion.
	3.	Assess utility and feasibility of developing effect-based criteria both for the White River and Statewide.
	4.	If effect-based criteria are useful and feasible, develop plan and schedule for obtaining the necessary data Statewide. If not, develop a plan to strengthen the reference-based approach.
	5.	Review plan and revise as necessary based on the data analyses.
2005	1.	Proceed according to revised plan.

Recommended process based on available data for lakes and reservoirs

Lakes and Reservoirs:

Possible activities and milestones for development of nutrient criteria for lakes and reservoirs:

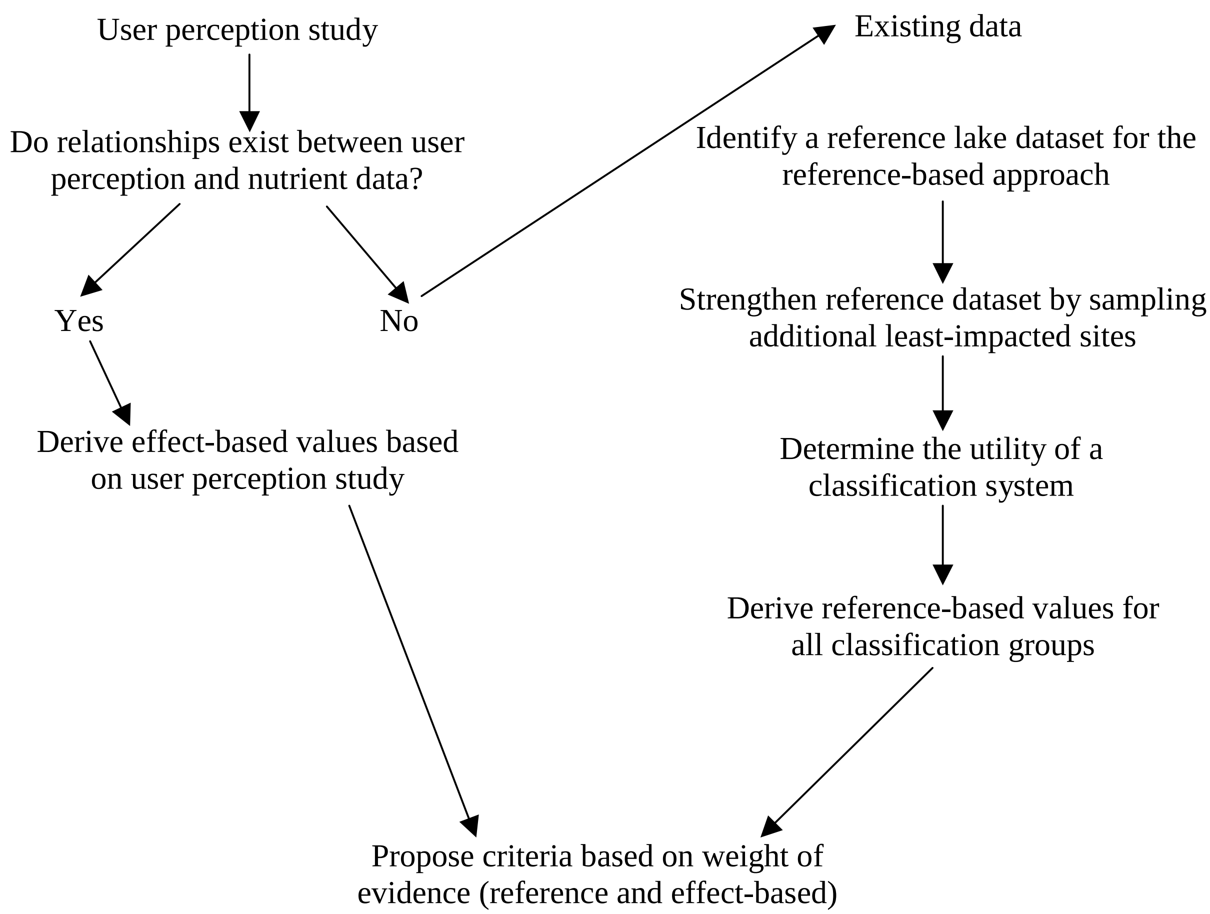
2002
1. Design and implement user perception study
2. Evaluate feasibility of refining EPA's reference condition by creating a reference lake data set of least impacted lakes and deriving values based on the nutrient levels found in these lakes. Implement sampling plan if determined to be useful.
3. Evaluate the need for subdividing lakes into different classes

2004
1. Compare values based on reference lake data, user perception studies, literature values, and EPA's criteria recommendations. Propose criteria that as accurately as possible reflect least impacted conditions and protect recreational uses.

Suggestions, Wetlands:

None at this time, pending completion of the wetland guidance document.

NOTE TO A.M.: WHAT DO YOU MEAN BY "WEIGHT OF EVIDENCE" IN THE CONTEXT OF THIS TABLE???



APPENDIX B.

NUTRIENT CRITERIA DEVELOPMENT PLAN OUTLINE

NUTRIENT CRITERIA DEVELOPMENT PLAN OUTLINE

In the January Federal Register notice, EPA recommends that states and authorized tribes develop a plan which will outline their process for how and when states and authorized tribes would adopt nutrient criteria into their water quality standards (66 FR 1671).

EPA encourages states to critically consider how best to achieve the goal of quantified nutrient criteria in water quality standards and provide EPA with the best possible information about the process and schedule for completion. This information will aid EPA in setting their expectations.

The purpose of this outline is to provide guidance to states and authorized tribes on developing a Nutrient Criteria Development Plan that will be a useful tool for developing and implementing nutrient criteria.

I. Criteria Development Process
   A. Conceptual Approach (The What and The How)
      1. Use EPA's Approach to Criteria Development as outlined in the appropriate EPA Technical Guidance Manual or
      2. Use EPA's 304(a) Nutrient Criteria Recommendations or
      3. Use Another Scientifically Defensible Method
         a. Empirical approaches
         b. Loading models
         c. Cause and effect based studies or relationships
         d. Other methods
   B. Relation to State/Tribe Use Classifications
      1. General Applicability to All Uses
      2. Applicability Tailored to Specific Categories
         a. General aquatic life uses
         b. Specialized aquatic life uses (e.g., designated beneficial use)
      3. Development of refined use classifications
   C. Relation to Physical Classification
      1. Lake Type (e.g., size and depth)
      2. Stream Type (e.g., Rosgen classification)
      3. Ecoregional sub-classifications
      4. Land-Use classifications
      5. Other natural geographic boundaries
   D. Inventory of Existing Data
      1. National Nutrient Data Base
      2. Other Data
      3. Identification of Data Gaps
      4. Identification of Data Base Management Needs

E. Requirements for New Data Collection
    1. Physical, Chemical, and Biological Measurement Variables (see attached list)
    2. Sampling and Analysis Plan (use consistent methods)
    3. Data Quality Objectives

II. Staffing and Resources (The Who)
- A. Internal
    1. Scientific Analysis Expertise
    2. Monitoring and Modeling Expertise
    3. Data Base Management Expertise
- B. External
    1. Consultants
    2. Universities
    3. Citizen and Watersheds Groups
    4. Shared Information from Other States/Regions/Federal Agencies
    5. Funding/Resources (e.g., 106, 104(b), 319)
        - a. Grants
        - b. Base Budget
        - c. Special Budget
        - d. Other

III. Schedule for Development and Adoption (The When)
- A. Products
    1. Milestones
    2. Intermediate Products
- B. Items to Consider
    1. Rationale for Criteria Decisions
    2. Administrative Procedures and Process
    3. Stakeholder Input and Public Participation
    4. RTAG Coordination
    5. Scientific Review
- C. Implementation Issues/Procedures
    1. Relationship of Criteria to Use Classifications
    2. Appropriate Uses of Criteria for Stressor and Response Parameters, and How they are Used in Combination
    3. Assessment Decisions
    4. Listing Decisions
    5. TMDL Development
    6. Permitting Decisions (application to low/zero flow systems)
    7. Application to Interstate Waters
    8. Protection of Downstream Waters (Lakes/Estuaries)

APPENDIX C.

DRAFT TEMPLATE FOR RTAG REVIEW OF STATE/TRIBAL NUTRIENT CRITERIA PROPOSALS

Draft Template for RTAG Review of State/Tribal Nutrient Criteria Proposals
George Gibson - 2/27/2002

Based in part on information in Lakes and Reservoirs Tech Guidance Manual.

1. State accepts Ecoregional Reference Condition as their Criteria.
   No further action required ... except we worry about why they took our numbers.

2. State submits alternative reference condition values.
   Review their rationale for the alternative numbers.
   - A. Significantly different from ecoregional reference condition values? If not can recommend EPA approval. If significantly different and less stringent, review for the following:
   - B. Subregionalization.
     No gerrymandering. No land-use designated regions.
   - C. Physical classification.
     Not based on trophic conditions or designated uses. But can use similar classes of waters based on retention times, max depth, color, inorganic turbidity, watershed characteristics -geology, slope, size, soils, native vegetation, but no factors related to cultural activities.
   - C. Data base.
     Larger N
     Purpose of the data collection indicated; screened for degraded sites.

   If RTAG concludes that a good scientific, objective case can be made for the alternative values, may then recommend this to the Regional EPA Office. If RTAG sees the potential for classification, etc which, if documented, would support such criteria, should discuss same with the State so they can enhance and resubmit their proposed criteria.

   When such cases are made and endorsed by RTAG, Regional EPA, and HQ EPA, the affected Ecoregional Criteria Documents should be amended accordingly.

3. State submits alternative reference condition and criteria variables.
   EPA expects the States/Tribes to include the four primary variables of TN, TP, Chlor-*a*, and SD or similar measure of clarity.

   However, additional variables can be included, usually DO and some form of algal or higher plant biomass and /or taxa composition; invertebrates and fish may be possible

4. State should always include a description of how the four or more variables will be incorporated in the criteria.
   Either a protocol for addressing variations on TN, TP, Chl-*a*, SD,... or an index relating the different variables as described in the technical manuals, esp lakes and reservoirs.

5. Temporal variation.
   Usually States are expected to establish nutrient criteria to be measured during the growing season, roughly June to September, but wet-dry season criteria or warm weather-cold weather seasonality may also be developed.

   Mean or median values for the entire year usually are not precise enough, present criteria documents not-with-standing, and seasonality or temporal indexing are preferred.

6. Monitoring procedure.
   Should be the same method as the sampling technique used to gather the criteria data including sampling at the same time period. One gab sample is not adequate to determine compliance. If State or Tribe chooses to use multiple sampling events and allow a certain minimal frequency of criteria non-compliances to account for variability, should **not** collect all samples during one short time period.

7. Sampling design.
   Stratified random sampling is preferred over completely random or fixed station sampling unless a good case can be made for fixed station sampling.

8. Analysis.
   State or EPA certified labs and EPA or "Standard Methods" procedures should always be used.

9. Criteria development.
   State or EPA reference condition values alone are not sufficient to serve as nutrient criteria. The other factors of history, data projection, attention to downstream effects, and regional expert consensus are also required before the criterion can be determined.

   Evaluate the rationale presented for the determinations made, if concerned ask questions to insure objective, disinterested approach was taken.

   Any guiding assumptions of the determining body should be clearly listed. Logistical cost considerations may be considered such as the amount of sampling which can be accomplished per year, but not any costs or social concerns about the consequence of criteria value determinations such as the number of sites that may be listed as failing to meet criteria or expense of remedial management. These questions can and should be addressed when subsequent standards determinations and/or water quality management options are deliberated.

10. Supporting Material:
   A. List of reference sites incl lat/long , water body name and county, dates and times of sampling and indication of efforts to preserve them from degradation

such as location on public lands, easements, and agreements to preserve sites and access.

B. List of test sites incl lat/long, water body name and county, dates and times of sampling.

C. List of State/Tribal experts participating in the determinations; incl title, affiliation, address, phone number.

D. Statement of consensus on the action signed by all participants or description of dissenting (alternative) opinions on the action.

Draft guidelines for EPA review of States' nutrient criteria development plans

A State's plan should provide sufficient detail to permit EPA to make an objective evaluation of the State's progress relative to the plan. The plan should also provide an estimate of the total time required to go from where the State is now to final adoption of quantitative nutrient criteria, including all administrative processes such as public hearings, legislative review and any others that might be appropriate. To accomplish this, each States' development plan should answer the following questions:

1. **Approaches** What approach or approaches will be pursued (effect-based criteria tied to specific designated uses, reference-based criteria or both or some other scientifically defensible approach)?
2. **Form** What parameters will be included in the criteria? If this will be determined later, what data will be used to determine it? If parameters are to be excluded (such as N), what is the basis for excluding them and are the data available or will they be generated to support this decision?
3. **Process** What will be the overall process (identify all expected tasks including technical and administrative, key decision point, possible outcomes, contingencies and the preliminary schedule)?
    a. **Regionalization** Will criteria be applied Statewide, to ecoregions, subecoregions, or some other system of regionalization.
    b. **Classification** How will the criteria be related to waterbody types? Will the State identify more classes of waterbodies than the rivers and streams, lakes and reservoirs, wetlands and estuaries that EPA's criteria recommendations recognize? If so, what different classes of waterbodies will be recognized?
    c. **Prioritization and Coverage** If the state intends to prioritize its waters for purposes of nutrient criteria development, what method will it use to prioritize waters, how will it ensure that all waters are ultimately covered and over what time frame? If the State believes that nutrient criteria are not needed for a specific water body or class of water bodies, how does the State intend to demonstrate that the waters are neither impacted nor threatened by nutrient overenrichment?
    d. **Inventory Existing Data** What are the available data that could be used as a basis for nutrient criteria and what are the limitations and gaps in the data?

e. **Data Collection** What data are needed based on the selected process for criteria development and the available data and how will those data be obtained?

f. **Assessing Progress** How often will the State assess its of progress and review its plan?

g. **Deviations and Revisions** How will EPA be notified of significant deviations from the plan? How will revisions be made and agreed to by the State and EPA?

4. **Specific Near-term Objectives** What is the workplan for the first twelve to eighteen months (What questions will be answered, what data are required to answer the questions, how will the data be gathered, how will the data be analyzed, how will it be accomplished, by who, how long should it take and what are the major milestones in accomplishing the first task)?

To accomplish this, we would expect that each state will submit the following information (or something equivalent):

- A schematic, process diagram describing how the State will go from where it is now (with respect to data and criteria) to final adoption of a complete set of criteria that will protect waters of the State from nutrient over-enrichment with estimated dates of completion of key tasks;

- Answers to any of the questions above that cannot be illustrated through the process diagram; and,

- A detailed workplan describing the activities the state will undertake during the first twelve to eighteen months including questions to be answered, data to be collected, data to be analyzed, methods of analysis and expected next steps based on the results of the analysis.

<u>Overview of information that should be included in a State's plan:</u>

I. General Objectives/Goals
   A. Data Status/Inventory
      1. Available data (time frame, parameters measured, no. of water bodies)
      2. Data needs
   B. Projects planned to address data needs
      1. Objectives
      2. Sampling design
      3. Time frame
   C. Criteria Development
      1. Selected parameters
         a. TN, TP, chl a, and turbidity
         b. Others–must provide rationale and supporting documentation (e.g. data illustrating a relationship between another parameter, like DO or low IBI scores, and nutrient concentrations
      2. Type
         a. Quantitative

b. Qualitative with a translator

3. Approach
   a. Effect-based
   b. Reference-based
   c. Multiple approaches–explain in detail
   d. Other–explain in detail
4. Classification used? (provide rationale)
5. Will waters be prioritized? If yes,
   a. How?
   b. Provide time frame
6. Will N & P criteria be set for all water bodies? If no,
   a. Only developing criteria for one pollutant for all water bodies
      (1) Must provide rationales (including quantitative evidence)
      (2) For example, in P-limited systems, states may not want to adopt N criteria. However, if N is a problem further downstream then States will need to have a mechanism for controlling N if necessary to ensure that downstream uses are not impaired.
   b. Not developing any criteria for a set of water bodies
      (1) Must provide detailed rationale
      (2) For example, States must demonstrate that :
         (a) the waterbodies are not exhibiting signs of nutrient overenrichment, AND
         (b) no significant point and non-point source discharges of nutrients to the waterbodies at the present time and that none are expected in the foreseeable future
            i) e.g., vast areas/watersheds within federal nature reserves that are protected from land use alterations

II. 2002-2003
   A. Objectives
   B. Strategy
      1. Sampling design & methods
      2. Data to be collected
      3. Data analyses
      4. Data sharing
   C. Milestones

III. 2003-2004
   A. Objectives
   B. Strategy
   C. Milestones

IV. 2004-beyond
   A.

Proposed Process for Reaching Agreement Between States and EPA on State Plans:

1. EPA provides comments on State plans as presented at last meeting

1. State and EPA meeting/conference call to discuss/clarify EPA comments
2. State revises plan (if necessary)

1. EPA reviews and concurs with revised plan (Staff level)

1. Concurrence letter from EPA WD Director to State Water Director documenting agreement on State plan

(The process for agreement on significant revisions to the plan would be essentially identical to that described above for initial agreement on the plan).

APPENDIX D.

ARIZONA STRATGEY FOR DEVELOPMENT OF NUTRIENT CRITERIA

Arizona Strategy for Development of Nutrient Criteria

A. Background

On January 9, 2001, EPA announced the publication of 17 nutrient water quality criteria documents for lakes and reservoirs, rivers and streams, and wetlands within specific ecoregions of the United States (*See* 66 Federal Register 1671). According to EPA, the ecoregional criteria recommendations are intended to serve as a starting point for states to develop more refined nutrient criteria, as appropriate, using published EPA water body-specific technical guidance manuals or other scientifically defensible approaches. EPA set a goal for each state to complete a plan for the development and adoption of nutrient criteria into their state water quality standards by the end of 2001. EPA expects states to adopt nutrient criteria into their state water quality standards by the end of 2004.

B. Purpose of EPA's Ecoregional Nutrient Criteria

EPA states that its recommended nutrient criteria are intended to protect against the adverse affects of cultural eutrophication [Id]. The intent of EPA's recommended ecoregional nutrient criteria is "to represent water quality conditions of surface water that are minimally affected by human development activities and to provide for the protection and propagation of aquatic life and recreation." [See 66 FR 1672]. EPA's approach can be described as an attempt to describe minimally impacted or reference conditions in each ecoregion for different water body types.

EPA aggregated available nutrient data for total phosphorus, total nitrogen, chlorophyll a, and turbidity for different water body types in each ecoregion. EPA's nutrient criteria recommendations are based on the 25th percentiles of an entire water body type population within an ecoregion. EPA recommends that these 25th percentile values be used as the starting points for each state to develop more refined nutrient criteria. EPA presumes that numeric nutrient criteria based on 25th percentile values will adequately protect all designated uses.

Arizona does not fully support EPA's reference condition approach as a basis for developing numeric nutrient criteria. In particular, ADEQ questions the use of recommended criteria based on 25th percentile values as starting points to develop of numeric nutrient criteria at more refined spatial scales in Arizona. The statistical representation of reference conditions and the use of 25th percentile values by EPA is understandable in light of the complexity and variability of nutrient dynamics at different spatial and temporal scales. However, ADEQ is concerned that implementation of EPA's 25th percentile approach may be overly conservative and result in the misidentification of surface waters as water quality-limited for nutrients that do <u>not</u> have cultural eutrophication problems.

C. Available Options

EPA states in the Federal Register that states have several options available to them in developing and adopting water quality criteria for nutrients. EPA recommends the following options, in order of preference:

Option 1: Wherever possible, develop nutrient criteria that fully reflect localized conditions and that protect specific designated uses, using the process outlined in EPA's technical guidance manuals for nutrient criteria development. Nutrient criteria may be expressed either as numeric criteria or as procedures to translate a state narrative standard into a quantified endpoint in state water quality standards. (Preferred Approach)

Option 2: Adopt EPA's recommended ecoregional criteria either as numeric criteria or as a translator for the state's narrative nutrient standard into a quantified endpoint.

Option 3: Use other scientifically defensible methods and appropriate water quality data to develop nutrient criteria protective of designated uses.

As a long-term nutrient criteria development strategy, ADEQ favors EPA's preferred approach. That is, ADEQ prefers the development of numeric nutrient criteria that reflect localized conditions and that protect specific designated uses. However, ADEQ does not believe that a fully refined set of numeric nutrient criteria can be developed for all of the state's surface waters by the end of 2004 given current data gaps and ADEQ resource constraints. Instead, the state proposes a tiered approach based initially on water-body type.

ADEQ notes that EPA has given states the option to develop: 1) numeric nutrient criteria **or** 2) translator procedures for a state-adopted narrative nutrient standard that can be used to translate the narrative standard into quantified nutrient endpoints on a case-by-case basis. As an interim approach, Arizona favors the development of translator procedures for the state's existing narrative nutrient standard to address nutrient over-enrichment.

The state currently has a narrative water quality standard that prohibits pollutants in amounts or concentrations that "[cause the growth of algae or aquatic plants that inhibit or prohibit the habitation, growth, or propagation of other aquatic life or that impair recreational uses" [*See* R18-11-108(A)(6).] ADEQ has developed a set of implementation guidelines for this narrative nutrient standard [*See* attachment, "Implementation Guidelines for the Narrative Nutrient Standard," Water Quality Assessment Unit of the Arizona Department of Environmental Quality (January 16, 1996)]. ADEQ which it proposes to amend to include procedures for translating the narrative nutrient standard into a quantified endpoints. This effort will commence in the spring of 2002.

D. Nutrient Criteria Development Workplan

EPA expects that each state will write a workplan describing how and when nutrient criteria will be adopted as part of a triennial review, or by some other process. EPA guidance outlined that the workplan should address items such as the criteria development process, staffing, and a schedule to complete the nutrient criteria adoption process. The nutrient criteria development plan should address the following questions:

What criteria development approach will Arizona use?
How will Arizona coordinate with the Regional Technical Assistance Group?
How will nutrient criteria relate to designated uses?

How will Arizona group surface waters? By geographic area? By physical or biological characteristics? By water body type? By designated uses? Some other classification scheme?
How will Arizona prioritize surface waters for nutrient criteria development?
What data will Arizona rely on? Will Arizona collect additional data?
How will Arizona analyze data?
What parameters will Arizona set criteria for?
What administrative procedures will Arizona use to develop nutrient criteria?
How will Arizona solicit public participation and stakeholder involvement?
Who will be involved in critical decision-making?
How will Arizona utilize outside expertise for data collection or analysis or peer review?
How will Arizona integrate its nutrient criteria development plan with plans from adjacent states and Tribes?
What are the major milestones and the schedule for completion?

1. Criteria Development

Arizona supports the development of nutrient criteria using a water body-type approach. Arizona's approach will be to focus first on the development of translator procedures for the narrative nutrient standard for lakes and reservoirs followed by development of translator procedures to implement the narrative nutrient standard for rivers and streams.

Arizona does not have separate surface water quality standards for wetlands and does not plan to develop translator procedures or numeric nutrient criteria for them. As an inland state, Arizona does not have estuaries or coastal marine waters. ADEQ has no plans to develop nutrient criteria for these water body-types.

2. How will Arizona coordinate with the Regional Technical Assistance Group?

ADEQ will participate with the Regional Technical Assistance Group for EPA Region IX to pool expertise and share resources with other states and Tribes in Region IX.

The key parameters that EPA addresses in the ecoregional nutrient criteria documents and that EPA expects states to address in their nutrient criteria development plans are total phosphorus, total nitrogen, chlorophyll a, and turbidity. EPA considers the first two parameters, total phosphorus and total nitrogen, to be causal agents of over-enrichment in surface waters. The latter two parameters, chlorophyll a and turbidity, are considered to be response variables or early indicators of over-enrichment. EPA encourages states to consider the development of additional criteria for other parameters such as dissolved oxygen, algal biomass, and biological integrity indices.

Arizona agrees that the key indicators identified by EPA in the Federal Register, with the possible exception of turbidity, are appropriate indicators to investigate for nutrient criteria development. Arizona also will investigate pH as a response variable and as a key indicator.

3. Relationship of Nutrient Criteria to Designated Uses

Water quality criteria are established for the purpose of protecting designated uses of surface waters. In fact, Arizona law requires that water quality standards be expressed in terms of uses to be protected [*See* A.R.S. §49-221(D)]. Arizona also has authority to adopt narrative standards that it deems appropriate. As noted previously, Arizona has a narrative water quality standard addressing over-enrichment at Arizona Administrative Code R18-11-108(A).

Arizona has adopted surface water quality standards that are expressed in terms of maintaining and protecting water quality for the following designated uses: domestic water source (DWS), fish consumption (FC), full-body contact recreation (FBC), partial-body contact recreation (PBC), aquatic and wildlife-cold water (A&Wc), aquatic and wildlife-warm water (A&Ww), aquatic and wildlife-effluent dependent water (A&Wedw), aquatic and wildlife-ephemeral water (A&We), agricultural irrigation (AgL), and agricultural livestock watering (AgL).

Arizona's will investigate the development of water quality standards to maintain and protect water quality for the state's aquatic life designated uses, specifically A&Wc, A&Ww, and A&Wedw. The relationship between nutrient over-enrichment, water quality impairment, and the attainment of aquatic life designated uses in surface waters is well-documented. For example, adverse effects to aquatic life from cultural eutrophication include low dissolved oxygen, fish kills, harmful algal blooms, excessive growth of macrophytes, and undesirable shifts in community structure and function. High algal and macrophyte biomass may be associated with diurnal swings in dissolved oxygen and pH in surface waters.

Arizona does <u>not</u> intend to pursue development of numeric nutrient criteria to maintain and protect water quality for the domestic water source, full body contact recreation, partial body contact recreation, fish consumption, agricultural irrigation, or agricultural livestock watering designated uses.

4. Classification of Surface Waters for Nutrient Criteria Development

EPA asked states to indicate how it will group surface waters for nutrient criteria development. There are various ways to group waters:

- by geographic area
- by physical or biological characteristics
- by water body-type
- by designated uses
- some other classification scheme

ADEQ intends to classify surface waters for nutrient criteria development using a layered approach of water body type, designated use, geographic area, and by physical, biological and chemical characteristics.

As noted in previous subsections, ADEQ intends to classify surface waters for nutrient criteria development by water body-type: 1) lakes and reservoirs, and 2) rivers and streams.

On bottom up approach recognizing lakes and reservoirs are the ultimate receiving waters into which rivers/streams may flow. To that end, ADEQ will target lakes/reservoirs first for criteria develop following up with streams/rivers.

ADEQ intends to classify surface waters for nutrient criteria by designated use. ADEQ will investigate the development of numeric nutrient criteria for three aquatic life designated uses: A&Wc, A&Ww, and A&Wedw.

4a. Lake/Reservoir Classification

Appendix A contains an outline that describes ADEQ's recommended approach and a tentative schedule for the lake classification project. The outline on the following page describes the geographical, physical, biological, and chemical characteristics that ADEQ will investigate for purposes of lake/reservoir classification. Appendix B outlines a proposed classification system for purposes of 305(b) and 303(d).

ADEQ plans to take a holistic approach in setting quantifiable endpoints for lakes and reservoirs describing least-impacted conditions in each lake class using a matrix of factors including: a) critical season (highest productivity), b) depth-integrated dissolved oxygen concentration, c) pH, and d) chlorophyll a / macrophytes on volumetric basis. ADEQ will rank lakes by respective class against least-impacted conditions using the same matrix of factors and establish criteria according to most critical or sensitive designated use at the 75th percentile of the least-impacted condition unless site-specific considerations or limitations inform otherwise. ADEQ will provide a translator for endpoint conditions when needed to establish nutrient limitations for upstream tributaries.

We believe that Arizona has distinctive regions, subregions, and types of lakes. For example, most of the lakes in Arizona are man-made impendent. Also, EPA has not defined ecoregional criteria for lakes and reservoirs in an arid desert environment below 5000 feet. These criteria need to be defined by Arizona.

Finally, a matrix approach is the most integrative and scientifically defensible approach because it does not rely solely on any one parameter to determine water quality impairment caused by over-enrichment. A matrix approach integrates natural variability and promotes the idea of whole-lake functionality within the context of designated use support.

4b. Stream Classification

The purpose of stream classification is to identify groups of streams or rivers in Arizona that have comparable biological, chemical, physical characteristics for nutrient criteria development. Stream classification reduces the variability of stream-related measures within classes and maximizes variability among classes. EPA recommends that states classify their streams first by type and then by trophic status. Initially, stream type classification is based primarily on physical parameters including climate, parent geology, substrate features, slope, canopy cover, discharge and flow continuity, stream order, size, and channel morphology. Trophic state classification focuses primarily on chemical and biological parameters, including concentrations of nutrients, algal biomass as chlorophyll a, and turbidity.

5. Prioritization of Surface Waters for Nutrient Criteria Development

ADEQ will prioritize for nutrient criteria development by water body type. In general, ADEQ will give first priority to the development of nutrient criteria for lakes and reservoirs. The development of nutrient criteria for rivers and streams will be given second priority.

Arizona lakes and reservoirs will be prioritized for nutrient criteria development based on data availability, vulnerability, and critical designated uses. ADEQ will investigate the development of numeric nutrient criteria for lakes and reservoirs in the following order of priority:

Major drinking water reservoirs
Vulnerable lakes with existing or potential eutrophication problems.
Lakes and reservoirs in targeted watersheds

Arizona rivers and streams will be prioritized for nutrient criteria development by flow regime and by degree of possible impact. ADEQ will investigate the development of nutrient criteria for rivers and streams in the following order of priority:

Effluent-dependent waters (A&Wedw)
Wadeable, perennial streams (A&Wc, A&Ww)
Large rivers

ADEQ will not develop nutrient criteria for intermittent waters or ephemeral streams. Ephemeral waters will be addressed only in the context of a conversion to an effluent-dependent water by a new discharge or through the TMDL process.

6. What data will Arizona rely on? Will Arizona collect additional data?

Initially ADEQ will rely on existing data collected by the Clean Lakes Program (1991 - present), Arizona Game and Fish Department, and a limited data set obtained by the U.S. Fish & Wildlife Service to develop nutrient criteria for lakes and reservoirs. ADEQ has existing data on nutrient species, dissolved oxygen, pH, and turbidity (as defined by chlorophyll, organic carbon, total suspended solids, and secchi depth) and trophic state (as defined by nutrient (or light) limitation / chlorophyll-biovolume / macrophyte % by volume).

Initially ADEQ will rely on existing water quality data on rivers and streams obtained by the WQD as part of its ambient surface water monitoring programs and other federal state parties existing data obtained by the U.S. Geological Survey through the ADEQ / USGS Cooperative Fixed Station Network Monitoring Program. Existing data is housed in the ADEQ Surface Water Quality Database. Existing data on nutrient species includes data on total phosphorus, total nitrogen, total Kjehldahl nitrogen, ammonia nitrogen, nitrate, and nitrite. ADEQ and its cooperators also collect field data on dissolved oxygen concentration, dissolved oxygen percent saturation, pH, turbidity, and stream flow at sampling sites. ADEQ also has existing bioassessment data on wadeable, perennial streams that has been collected each water year since 1992.

The ADEQ Surface Water Monitoring Program and Clean Lakes Program plan to collect additional data. Upon identifying data gaps and will continue to fill data gaps under ADEQ's targeted watershed monitoring schedule. ADEQ staff conduct surface water quality monitoring in targeted watersheds each fiscal year. ADEQ has delineated 10 major watersheds in Arizona. ADEQ staff target monitoring resources in two watersheds each water year. All 10 major watersheds are monitored over a 5-year period. ADEQ will continue to collect additional water chemistry and bioassessment data at sampling sites located in the targeted watersheds.

As resources allow, ADEQ intends to collect additional data on periphyton (algae) communities in wadeable, perennial streams and effluent-dependent streams to start building a statewide dataset that may be used to define reference conditions for algal communities. In 2001, ADEQ initiated a periphyton pilot project in the Santa Cruz River basin in an attempt to develop the use of diatom communities as a bioassessment tool. The primary objective of the Santa Cruz Periphyton Pilot Project was to test several metrics using artificial substrates to determine which metrics best distinguished between diatom communities at potential reference sites (i.e., least-impacted surface waters) and diatom communities in effluent-dependent waters in the Santa Cruz River basin. A secondary objective of the periphyton pilot project was to determine if there were similarities in the diatom communities at the selected reference sites. ADEQ will collect data on algal community composition and biomass in wadeable, perennial rivers and streams to evaluate its potential use as a bioassessment tool and as a key indicator of nutrient over-enrichment.

7. Data Analysis

ADEQ proposes to use the following methods of data analysis: a) descriptive statistics for ranges with frequency distribution, b) non-parametric rank sum significant difference - by season, by source water, etc., c) spatial or temporal trends where there is sufficient data, and d) multi-variate statistical analyses to compare against EPA-derived ecoregional criteria to refine endpoints as a matrix of factors, expressed in ranges that represent degrees of impairment.

8. For what parameters will Arizona establish criteria?

ADEQ has not determined which parameters to establish nutrient criteria at this point in time. ADEQ will investigate the feasibility of establishing nutrient criteria for key indicators including total phosphorus, total nitrogen, and chlorophyll a. ADEQ also will investigate the feasibility of establishing numeric criteria or thresholds for pH and dissolved oxygen concentrations. Per EPA guidance and as resources allow, algal communities and biological data will be incorporated into the process.

9. Administrative, public participation and stakeholder involvement procedures

ADEQ is required by state law to adopt surface water quality standards through rule making [See A.R.S. §49-221(A)]. Consequently, the adoption of numeric nutrient criteria into the surface water quality standards rules must be done through administrative rule making. In general, ADEQ addresses the adoption of new or revised water quality standards in the triennial review process.

There are several opportunities for public participation in the triennial review process. ADEQ typically holds both informal public hearings to identify issues and to solicit informal public comment on surface water quality standards before initiating the formal rule making process to revise the surface water quality standards. Federal and state law require that ADEQ hold public hearings during the formal rule making process to take public comment on proposed revisions to the surface water quality standards.

A recently-enacted state law in Arizona requires that ADEQ adopt implementation procedures for narrative standards after providing an opportunity for public notice and comment. A.R.S. §49-232 governs the listing of impaired surface waters under §303(d) of the Clean Water Act. A.R.S. §49-232(F) states, in relevant part:

Before listing a navigable water as impaired based on a violation of a narrative or biological surface water quality standard *and after providing an opportunity for public notice and comment*, the department shall adopt implementation procedures that specifically identify the objective basis for determining that a violation of the narrative or biological criterion exists. A total maximum daily load designed to achieve compliance with a narrative or biological surface water quality standard shall not be adopted until the implementation procedure for the narrative or biological standard has been adopted [emphasis added].

A.R.S. §49-232(F) mandates that public notice and comment procedures be used to develop the implementation procedures for the narrative nutrient standard. Thus, the public must be given an opportunity to comment on the development of implementation procedures that translate the narrative nutrient standard into quantifiable endpoints. Moreover, ADEQ has a successful stakeholder model that it uses in the development of rules, policy and guidelines. The WQD will involve the stakeholders throughout the development process.

10. Who will be involved in critical decision-making?

11. What are the major milestones and the schedule for completion?

Clean Lakes Program Milestones:	Schedule:
• Data collection	01/91 - 09/04
• Data analysis	01/02 - 03/04
• Watershed delineations (GIS)	09/01 - 03/02
• GIS database for multivariate analysis	01/04
• Initial data pull / reformatting for analysis	09/01 - 01/02 (and ongoing)
• Web site information - updated quarterly	03/02
• Public participation	Ongoing

- Peer review of proposed plan 06/02
- Peer review of proposed criteria 06/04
- Final criteria and implementation 12/04 (promulgation 2005)

12. How will Arizona utilize outside expertise for data collection or analysis or peer review?

ADEQ is evaluating whether to seek the support of outside contractors for portions of this project. A decision is still forthcoming.

13. How will Arizona integrate its nutrient criteria development plan with plans from adjacent states and Tribes?

ADEQ will work with adjacent states (California, Nevada, Utah and New Mexico) and Tribes in Arizona with authorized water quality standards programs to integrate the state's nutrient criteria development plans with theirs. ADEQ will work to ensure consistency in the nutrient criteria development processes that are used to develop nutrient criteria for interstate waters, particularly the Colorado River and its major impoundments (e.g., Lake Powell, Lake Mead, Lake Havasu, and Lake Mohave), the Gila River, Virgin River, and the San Francisco River.

Appendix A

Lake/Reservoir Classification Project
Regional Criteria Development
Phase I

GIS Data Review

Watershed characteristics for each lake/reservoir

1. Ecoregion

Size in acres

3. Geology type
4. Soil type
5. Average slope
6. Dominant vegetation
7. Vegetation cover
8. % dominant land use(s)
9. Impervious cover
10. Annual precipitation
11. Annual rainfall
12. Mean winter temperature
13. Mean summer temperature

Physical lake/reservoir characteristics

1. Natural vs. manmade
2. Age of lake or reservoir
3. Sediment depth
4. Basic shape (simple, complex, riverine)
5. Elevation @ maximum pool
3. Surface acres
4. Maximum depth
5. Average depth (approximated from topo)
6. Volume (estimated)
7. Number of tributaries
8. Water retention time
9. Average winter temperature
10. Range in summer temperature (mean?)
11. Evaporation rate
12. Summer secchi depth

Appendix A

Lake/Reservoir Classification Project
Regional Criteria Development
Phase I

II. Database Review

A. Chemical lake characteristics
1. Alkalinity
2. Hardness (carbonate/bicarbonate, sulfate, other)
3. TDS / EC
4. TSS / VSS
5. TOC / DOC
6. Winter & summer: total N & total P in water
7. Winter & summer: total N & total P in sediment
8. Winter & summer DO (volume-avg)
9. Winter & summer pH (volume-avg)

B. Biological lake characteristics
1. Peak biomass (summer chlorophyll a)
2. % macrophyte cover
3. Algal / plant diversity (% dominant species : total species)

III. Phase Ia: Statistical Analyses - identify available metals / multiple correlation / regression / multivariate

Watershed & lake physical factors
Watershed physical factors & lake chemical factors
C. Watershed physical factors & lake biological factors
D. Lake physical factors & lake chemical factors
E. Lake physical factors and lake biological factors

IV. Define reference 'nutrient' condition for following a priori classes:
A. Reservoirs - trends...look for no significant change over time.
B. High elevation lakes > 7500' - least impacted by grazing / recreation.
C. Volcanic lakes - least impacted by grazing / recreation.
D. Mid-elevation (AZNM) lakes < 7500' least impacted.
< 4 m mean depth (macrophyte-dominated)
> 4 m mean depth (algae-dominated)
Karst lakes

Southern desert lakes 4000-5000' least impacted.
Sky island lakes > 6000' least impacted.
Urban lakes, least impacted.
Others?

Appendix A
Lake/Reservoir Classification Project
Phase I

V. Data gap analysis

VI. Phase Ib: Statistical Analyses

Run all lakes against each reference class to test % departure for nutrients

1. Numeric values and trophic parameters

Lakewatch

Observed vs predicted nutrient loading from streams (Eutromod?)

VII. Design Fiscal Year 2003 and 2004 sample plans to address data gaps and to obtain additional data for nutrient criteria development

VIII. Status report on Lake Classification Project for internal review due December 31, 2002 and December 31, 2003.

Appendix A
Lake/Reservoir Classification Project
Regional Criteria Development
Phase II

Test criteria and gather additional data in 2003 / 2004;

Prepare Recommended Nutrient Criteria for Lakes and Reservoirs in 2005 Triennial Review

Appendix B
Lake/Reservoir Classification Project
Lake Classification for Purposes of 305(b) and 303(d) (Proposed)

Designated Uses: A&Wc, A&Ww, A&Wedw

I. *A&Wc Use*:

A. Lakes that AGFD has determined to support salmonid populations *year-round*
B. Lakes with an average summer temperature of 20 degrees C or less
C. Lakes with an average DO concentration of 6.0 mg/L or greater
D. Lakes above 7500 ft elevation
E. Lakes with a mean depth of 7 meters or more

<u>Assessment for narrative nutrients: see general lakes assessment criteria</u>

II. *A&Ww General*:
A. Lakes that AGFD has determined will not support salmonids *year-round*
B. Lakes with an average summer temperature of 20 degrees C or more
C. Lakes with an average DO concentration of 5.0 or greater
D. Retention time variable (treat as IIa if more than 1 month <u>and</u> mean depth less than 4 meters)
E. Lakes below 7500 ft elevation
F. Not in urban landscape
G. Trout stocked only when mean water temperature falls below 20 degrees C

<u>Assessment for narrative nutrients: see general lakes assessment criteria</u>

IIa. *A&Ww Shallow*:
A. Lakes that generally follow Type II, but have a retention time greater than 1 month and/or a mean depth of 4 meters or less
B. Lakes that have a tendency to be macrophyte-dominated, not algal dominated
C. Macrophytes will shade out algae; expect greater water clarity
D. Expect greater swings in DO and pH diurnally
E. Expect very high biomass present as aquatic macrophytes

<u>Expectations in regard to narrative nutrients attainment: will need to consider recreation support differently</u>

Expect very high biomass will need to be treated/controlled/reduced - to meet DO and pH expectations and/or to meet other recreational expectations
First line of attack is to mechanically harvest (and remove) vegetation on a regular basis
May consider selective herbicides or biocontrols for invasive species

4. May consider dredging on rotational basis
5. Fish stocking of tolerant species only; trout not allowed
6. DO and pH must be maintained to ensure 50% habitat fully supported for existing species

III. *Urban A&Ww General Recreational Use:*

A. Manmade lakes within urban environment
B. Mean depth at least 4 meters
C. Receive no treated effluent
D. Retention times greater than 1 month
E. Regularly stocked with fish, trout included but not in warm months; carry FC
F. May receive chemical treatment; requires periodic testing for priority pollutants
G. If carry FBC; requires bi-weekly bacteria testing, otherwise PBC

<u>Expectations regarding attainment of narrative nutrient standard in type IIa waters</u>: will allow some flexibility in recognition of design limitations, climatology, source water, and storm water influences as long as all BMPs are in place:

1. Expect up to 50% higher nutrient/organic loading type II A&W lakes
2. Expect need to treat/control/compensate for excess biomass: chemically and/or mechanically in already established urban lakes
3. Expect design considerations to minimize need for #2 above in new lakes

4. Trout stocked only when mean water temperature falls below 20 degrees C
5. Urban A&Ww composite TSI may be up to 150% non-urban A&Ww TSI
6. DO and pH must be maintained such that a refuge of 66% of the appropriate habitat is provided to support each aquatic organism present in any one season
7. Biomass (algae and/or aquatic plants) must be managed such that #6 above is fully supported
8. Expect 1-2 fish kills may be encountered until each lake has carrying capacity established; a goal of no fish kills after 3 years is reasonably expected

III. *Urban A&Wedw Limited Fishery Use*:

A. Manmade lakes that receive treated effluent as major source water
B. Retention time greater than 1 month
C. Mean depth at least 3 meters
D. May be regularly stocked with fish (*no trout)*; catch and release only
E. Carry only PBC; requires monthly bacteria testing
F. May receive chemical and/or mechanical treatment for algae and/or plants

Appendix B
Lake/Reservoir Classification Project
Lake Classification for Purposes of 305(b) and 303(d) (Proposed)

<u>Expectations with regard to attainment of narrative nutrient standards in Type III waters</u>:

1. Expect much higher (33-150%) nutrient loading than non-effluent dominated lakes (minimum of extended aeration secondary treatment)
2. TSI may be up to 200% of Type II TSI, not to exceed 1.33 x hypereutrophic threshold
3. Expect in-lake BMPs to include aeration/circulation, chemical or mechanical treatment of algae and/or aquatic vegetation
4. Expect stocking with tolerant species only
5. Expect habitat for tolerant species to be maintained in 33% of water column and 50% of benthic surface
6. Expect periodic fish kills may occur; analysis of carrying capacity will minimize these events

IV. *Urban A&Wedw, non-fishery:*

A. Receive treated effluent for storage purposes
B. May be stocked with fish for vector control; fishing not allowed
C. Fenced off from public access if no PBC
D. May receive any chemical and/or mechanical treatment for algae/plants that is acceptable for end use of water

Expectations for narrative nutrient standard: none

Priority for development of nutrient criteria

APPENDIX E.

SITE SPECIFIC STUDIES

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)	Geographic Location	Source(s)	Additional Comments
Drinking water/ Aquatic life, growth of musty odor-producing algae supported			>30 μg/L TP	>10 filaments of O*scillatoria 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 & 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					EPA's water quality criterion	
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
STREAMS/ RIVERS								
Aquatic life, decreased biotic integrity using fish and invert indices (inc in tolerant and omnivore spp, dec in EPT taxa)	Streams, headwater & 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"	Streams, New			>13-20 mg/m^2 (mean		New Zealand; 21 streams	Biggs 2000, Ministry of	Based on diatom/cyanobacteria assemblages. Low biomass does not suggest only EPT will

EPT invertebrates (and increase in rel abundance of chironomids and oligochaetes)	Zealand			monthly)			Environment	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.
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 & 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 & Hickey 1990	Trophic status determined by percent of catchment developed for agriculture: oligo-<1%, meso-1-30%, and eutrophic->30 %.
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 >20µg/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			100-150 mg/m^2		National?	Welch et al. 1988, Horner et al. 1983	Max biomass to avoid recreational and aesthetic use impairment; Articles state that levels are based on 19 enrichment studies, but specific studies are not cited.
Recreation/Aesthetics, nuisance levels	Rivers, streams	300-350 µg/L	>30 µg/L			Clark Fork River, MT (some data from NA, Europe, & 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).
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.
Recreation/Aesthetics, nuisance level	Rivers/ Streams			150-200 mg/m^2		Washington, Spokane River	Welch et al. 1989	Nuisance level based on perceived impairment.
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; cyanobact 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)
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 & 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 versus TP for 21 reservoirs. LogFSS=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.
Aquatic life, max biomass of sport fish	Reservoirs /Impoundments (>200 ha)		>100 µg/L			Virginia, Arkansas, & Nevada reservoirs	Ney 1996;	Suggests that sport fish biomass does not peak at less than 100 µg/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 & piscivorous fishes	Reservoirs (>200 ha)		**see comments			Smith Mountain Lake, Virginia	Yurk & 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 & Jones 1991	Levels noted by the authors solely based on their experiences (no data shown to support values).
Aquatic Life, peak in sportfish yield	Lakes (depth <6 m; Area 6-24000 ha)			>70 mg/m3		Missouri and Iowa	Jones & 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
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 comm structure should lead to shift in herbivore community.

Recreation	Lakes				1.2 m	New York	Effler et al. 1984 (cited in Heiskary and Walker 1988)	State standard for beaches. Basis unclear.
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	Lakes				1.2 m	Mass.	MDPH 1969 (cited in Heiskary and Walker 1988)	State standard for beaches. Basis unclear.
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.
Aesthetics, water clarity	Lakes		>20 µg/L	>10 mg/m^3	<1.5 m SD		Bachmann & 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 & Davies-Colley 1999	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 Secchi depths are noted in ( ). Basis for category "labels" is unclear.
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 & 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/m^2		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.
	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 & 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 + 4.285log d_a – 0.929(log $d_a)^2$ + 0.504log SIN (R^2=0.74) Log chl_{max}=-2.714 + 4.716log d_a – 1.076(log

								$d_a)^2$ + 0.494log SRP (R^2=0.72) (d_a=mean days of accrual)
	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 & 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.

APPENDIX F.

POTENTIAL DATA SOURCES

First Name	Last Name	Agency
Patti	Armison	Tahoe Research Group (TRG)
Jeffrey	Armstrong	Orange County Sanitation District
Jeanique	Artiola	University of Arizona
Brenda	Begay	White Mtn. Apache Tribe
Bryan	Bennon	Gila River Indian Community
Michael	Carlan	City of San Francisco Public Water Utilities
Jay	Cass	CA RWQCB-6
Robert	Gearheart	HSU, Env. Resource Engineering Dept.
Nancy	Grimm	AZ. State University
Matt	Hegemann	US Park Service
Terry	Knight	NV Nature Conservancy
Kevin	Kratt	Ameritech (Performing nutrient TMDL's)
John Paul	Kyle	Tahoe Regional Planning Agency
Liz	Lewis	Marin County Flood Control Dist.
Chris	Maxwell	CA RWQCB-6
Glenn	Miller	UNR, Environmental Resources Program
Brian	Niewinski	Pyramid Lake Fisheries
MJ	Oliveri	City of Santa Rosa, Public Works
Patti	Orozco	City of Santa Rosa, water quality
John	Reuter	UC Davis
Glenn	Stark	Gila River Indian Community
Lynette	Stevens	Navajo EPA
Marc	Sylvester	USGS NAWQA Menlo Park
Karen	Thomas	USGS
Dean	Tucker	US Park Service
Roland	Williams	AZ Dept. of Environmental Quality
Iris	Yamagata	CDEC- DWR Fresno
Victor	Baker	University of Arizona
Marie	Barry	Washoe Tribe of NV and CA
Judy	Bloom	EPA Region IX
Val	Connor	CA RWQCB-5
Mike	Deas	UC Davis
Terry	Flemming	US EPA Region IX
John	Johnston	Calif. State University, Sacramento
Cindy	Larkin	City of Eureka

First Name	Last Name	Agency
Jack	Lewis	Redwood Science Lab
Geoff	Powers	County of Sonoma, Stormwater
Tina	Rhom	US EPA
Larry	Roundtree	Bureau of Health Protection
Stewart	Schillenger	City of Tucson, Dept. of Water Quality
Nancy	Vacinich	Pyramid Lake Fisheries
Sean	White	Sonoma Co. Water Agency
Mike	Young	Prescott Water Treatment, City of Prescott, AZ
		Arizona Water Resources Research Center
		Natural Resources, Division of Water Planning
		Carson River Advisory Committee
Dave	Bogner	CA DWR Central Valley Region
Gale	Cordy	USGS NAWQA
Jennifer	Davis	Scott River CRMP
Marie	deAngelis	HSU Oceanography Dept.
Niel	Dubrovsky	USGS NAWQA
Tom	Galier	City of Tempe
Gregory	Gearheart	CA EPA, CA RWQCB-2
Bob	Hollander	City of Phoenix
Bob	Klamt	CA RWQCB-1
Mark	Larkin	Friends of Santa Cruz River
Mike	Lico	USGS NAWQA
Alan	Martindale	City of Mesa
Gene	Michael	City of Glendale
Barbara	Oliveri	City of Scottsdale
Carol	Rische	Humboldt Bay Municipal Water Supply
Kathleen	Ruttenberg	Woods Hole Oceanographic Institution
Pat	Sampson	City of Chandler
Jeffrey	Stoner	USGS
William	Taylor	City of Gilbert
Ken	Velutz	USGS NAWQA
Stan	Wiemeyer	USFW Reno
Adele	Basham	Nevada Department of Env. Protection
Bob	Berger	EBMUD
Martha	Conklin	University of Arizona
Scott	Dawson	Santa Ana RWQCB

First Name	Last Name	Agency
Richard	Engel	Humboldt Water Resources
Marilyn	Ethelbah	Ft. McDowell Indian Community
Theresa	Foglesong	USGS
Jill	Geist	City of Arcata
Chris	Hepe	US EPA Region IX
David	Herbst	Sierra Nevada Aquatic Research
Hans	Krock	University of Hawaii
Michael	Lyons	CA RWQCB-4
Mary	Madison	UC ICE
Pat	Mariella	Gila River Indian Community
Diana	Marsh	Arizona Dept. of Environmental Quality
Alan	Miller	CA RWQCB-6
Al	Olsen	USFS
Bernadette	Reed	CA RWQCB-1
Lynn	Small	City of Santa Rosa
Gordon	Smith	Hawaii DOH
Debbie	Smith	CA RWQCB-4
Hope	Smyth	CA RWQCB-8
Jeff	Stuck	ADEQ, Drinking Water Division
Evelyn	Thompkins	DWR Southern District
Judith	Unsicker	CA RWQCB-6
Erwin	Van Niewenhuyse	USF&WS Stockton
Dave	Webb	Shasta RCD
Rita	Whitney	Tahoe Regional Planning Agency
Mike	Wilson	Humboldt Water Resources
Robert	Ziemer	USFS PSW Redwood Sciences Laboratory
Shirley	Birosik	CA RWQCB-4
Jerry	Boles	CA Department of Water Resources, N. District
Lorrie	Bundy	Siskiyou RCD
James	Carter	USGS Menlo Park
Greg	Crawford	HSU Oceanography Dept.
Randy	Dahlgren	UC Davis, Dept. of Land, Air and Water Resources
Larry	Dugan	Bureau of Reclamation
Greg	Elliott	Salt River Project
Susan	Fitch	AZ DEQ, Clean Lakes Program

First Name	Last Name	Agency
Sid	Fong	Bright Chemical Laboratories
Gary	Gilbreath	DWR Southern District
Bruce	Gwynn	CA RWQCB-1
Robert K.	Hall	US EPA Region IX
Mark	Harvey	CA RWQCB-5
John	Heggeness	NV Dept. Env. Protection
Rodney	Jung	EBMUD
Perry	LeBeouf	CA DWR
Alan	McKay	Desert Research Institute
John	Munn	US Forest Service
Mike	Napolitano	CA RWQCB-2
Sam	Rector	AZ Dept. of Env. Quality
Amanda	Ryan	AZ Dept. of Environmental Quality
Tom	Scott	Lake Merry Water Treatment Plant
Patti	Spindler	AZ Dept. of Environmental Quality
Ron	Stillwell	City of Williams
Richard	Svetich	Tahoe Truckee Sanitation Agency
Judith	Unsicker	CA RWQCB-6
Brian	White	Los Angeles Dept of Power and Water
Rich	Breuer	DWR Central District
Kevin	McKernan	Hoopa Tribe
		Irrigation Districts (various)
		Public Utilities Districts (various)
		AZ game and Fish Dept.
		Resources Conservation Service
		Army Corps of Engineers
		Bureau of Land Management
		EPA Region IX RTAG
Craig	Wilson	SWAMP
Peter	Otis	CA RWQCB-1
Karen	Taberski	CA RWQCB-2
Steve	Moore	CA RWQCB-2
Karen	Worcester	CA RWQCB-3
Tracy	Patterson	CA RWQCB-4
Jonathan	Bishop	CA RWQCB-4
Jeanne	Chilkott	CA RWQCB-5

First Name	Last Name	Agency
Karen	Larsen	CA RWQCB-5
Betty	Yee	CA RWQCB-5
Tom	Suk	CA RWQCB-6
Joan	Stormo	CA RWQCB-7
Pavlova	Vitale	CA RWQCB-8
Linda	Pardy	CA RWQCB-9
Dave	Gibson	CA RWQCB-9
Bruce	Posthumus	CA RWQCB-9