APPENDIX A

Background Information for EPA Region IX RTAG / STRTAG Nutrient Criteria Development Process

2003 Progress Report

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
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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 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 the 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.

A concept paper providing information on issues critical to the development of nutrient criteria in Region IX was distributed to the RTAG / STRTAG in March 2002. This concept paper is known as the White Paper and is available at the EPA Region IX website (rd.tetratech.com/epa username: epauser - password: Region IX). The concept paper compiled information on topics that were to be considered during the EPA Region IX RTAG / STRTAG Nutrient Criteria Development Work Plan Workshop. The three-day workshop was held in April 2002 in San Diego, California. Over fifty members of the RTAG / STRTAG attended the workshop. Presentations and discussion sessions were included for each of the topical areas to be included in the work plan. Each session closed with a summary of majority position on each topic to provide direction for the work plan. A workshop summary report was issued on March 15, 2002 to the RTAG / STRTAG to verify the workshop discussions and direction provided. One set of comments were received on the meeting summary and they were addressed in a memorandum sent to the RTAG / STRTAG in June, 2001. With this direction the technical support team began developing the EPA Region IX Work Plan. The work statement for the development of nutrient criteria for ecoregions within EPA Region IX was distributed to the RTAG / STRTAG in mid-July 2002.

As plans were being made to implement the work plan the process took a six-month hiatus due to project funding and contract administration issues. When the issues were resolved in February 2003 the projected funding had been reduced by 66%. The delay and reduced funding necessitated modifications to the work plan to accommodate a compressed schedule and reduced funding. The solution was to undertake the Southern California Oak and Chaparral Ecoregion Pilot Study. This adjustment is a minor concession since the original work statement had called for addressing one ecoregion at a time. The major change to the work plan was to consider all waterbodies within the region on a comprehensive watershed basis rather than sequentially evaluating individual waterbody types. These modifications were presented to the RTAG / STRTAG in a background information memorandum and discussed during an April 2003 conference call. Additional information was provided to the RTAG / STRTAG in a project update memorandum that was distributed in June 2003. This report is the result of the tasks undertaken as part of the Pilot Project. The next steps in the adaptive management process will include a workshop to review the results of the pilot project and to develop next steps for nutrient criteria development.

Figure 1-1.

Table 1-1. Key Milestones and Draft Schedule for Nutrient Criteria Development

Milestone	1998	1999	2000	2001	2002	2003	2004
1. National Strategy for the Development of Regional Nuttrient Criteria		◆					
2. EPA Region IX Technical Advisory Group Formed		◆					
3. EPA Guidance Document Published							
Rivers & Streams				◆			
Lakes & Reservoirs			◆				
Wetlands							
Estuaries				◆			
4. EPA Region IX Demonstration Project Ecoregion II Rivers & Streams			▬ ◆				
5. EPA Ecoregional Nutrient Criteria Documents							
Rivers & Streams				◆			
Lakes & Reservoirs					◆		
Wetlands							
Estuaries							
6. California State Regional Technical Advisory Group Formed				◆			
7. RTAG / STRTAG resolution to pursue development of alternative nutrient criteria				◆			
8. Nutrient Criteia White Paper and Workshop					◆		
9. Nutrient Criteria Development Work Statement					◆		
10. Project Hiatus					▬	▬	
11. Pilot Project Background Information						◆	
12. Pilot Project Implementation						▬	
13. Pilot Project Report						◆	
14. RTAG / STRTAG Pilot Project Next Steps Workshop						◆ ●	● ● ● ● ● ● ● ●
15. Final Report for Ecoregions 6, 5, 1, and 78						▬	▬ ◆

◆ **Milestone**
▬ **Ongoing Activity**
● **RTAG / STRTAG Conference Call**

Table 1-1. 2000 Pilot Study Nutrient Data Summary

Ecoregion*	Total Phosphorus (approx. mg/L)					Total Kjeldahl 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

1.2 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 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

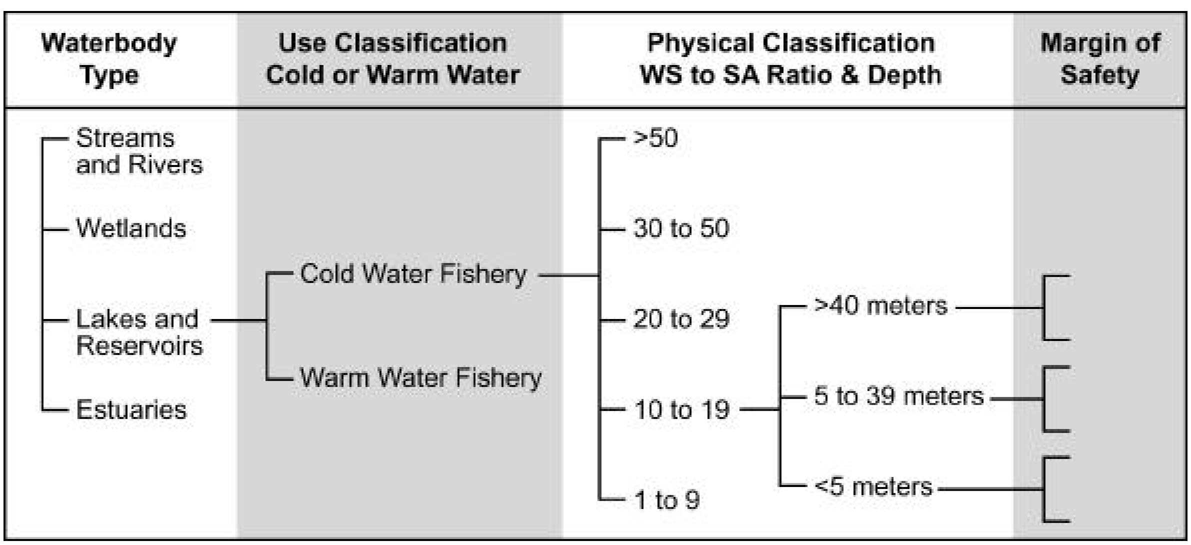
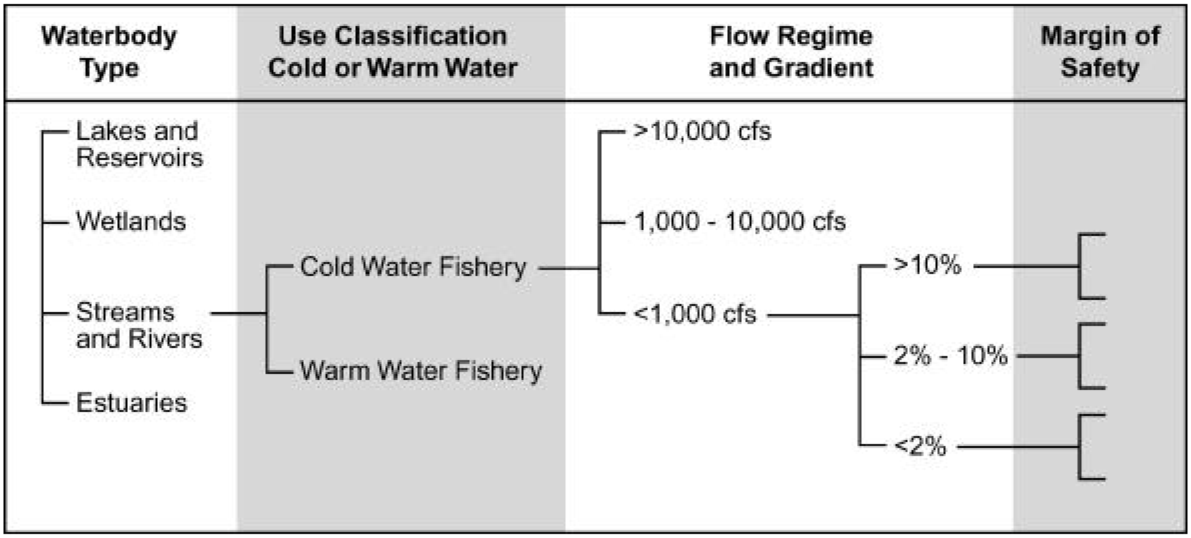


Figure 1-3. Decision Diagram for Lakes and Reservoirs



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 classification criteria. The framework eventually branches to the final decision point where a protective range of the criteria parameter(s) is assigned.

The approach 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.

1.3 ORGANIZATION OF THIS REPORT

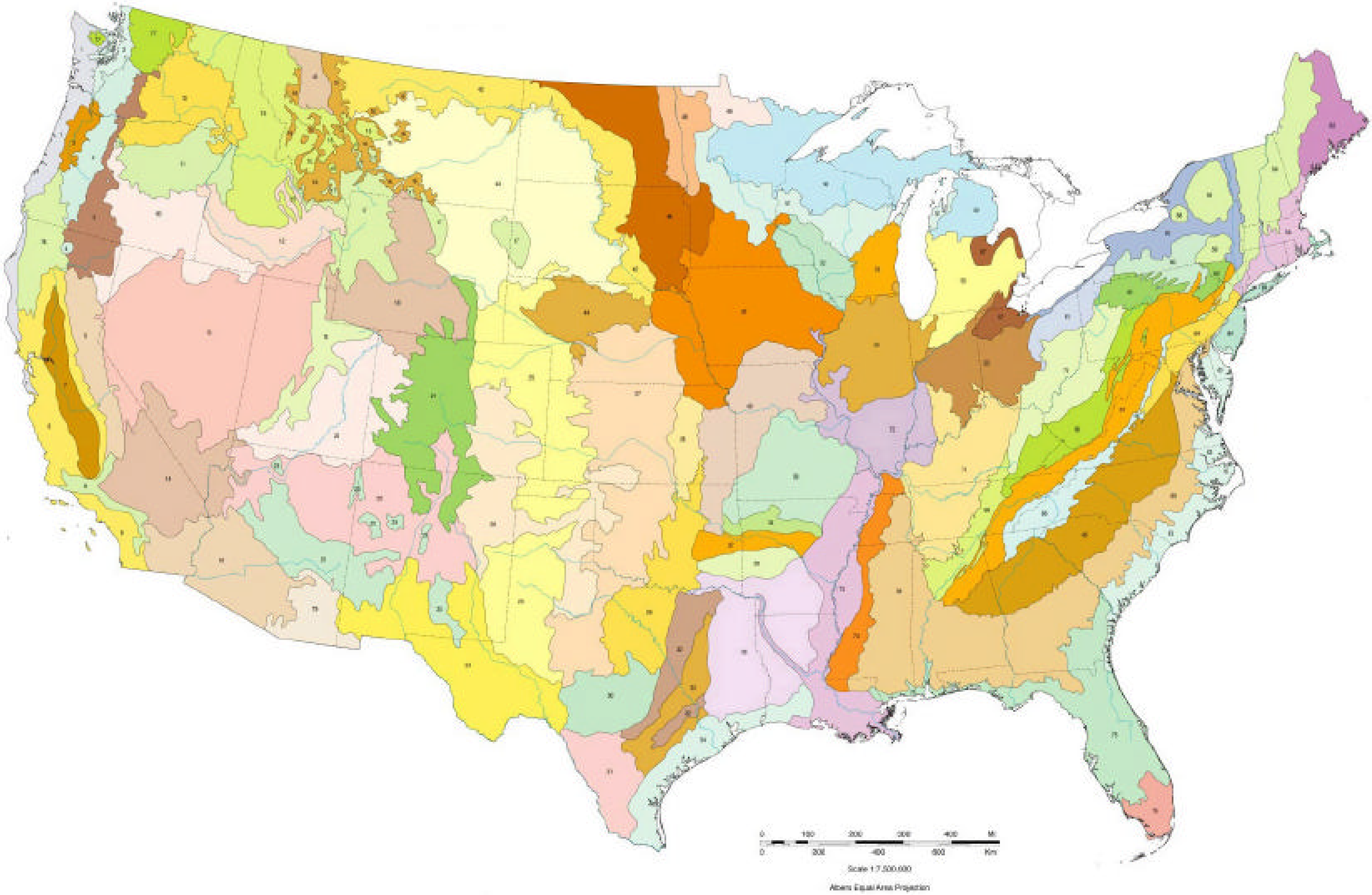
Section 1 of this document provides an overview of the nutrient criteria process including both past and future milestones. Section 2 explains the basis of the regional approach and identifies potential confounding factors that will need to be addressed outside the nutrient criteria framework Section 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. EPA Default 304(a) Ambient Water Quality Criteria Recommendations for Rivers and Streams

No.	Ecoregion Name	Total Phosphorus (mg/L)			Total Nitrogen (μg/L)			Chlorophyll a (mg/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 Chaparral 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 Archipelago	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. EPA Default 304(a) 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	25th	Min	Max	25th	Min	Max	25th	Min	Max	25th
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 Chaparral 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 Archipelago	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)

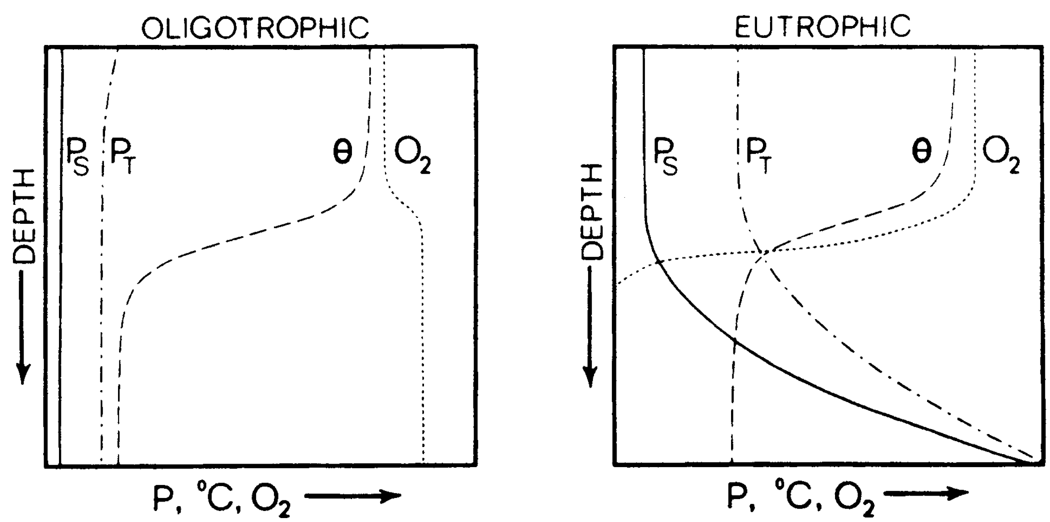


Figure 3-2. Generalized vertical distribution of ammonia and nitrate-nitrogen in stratified lakes of very low (oligotrophic) and very high (eutrophic) productivity (Source: Wetzel 1983)

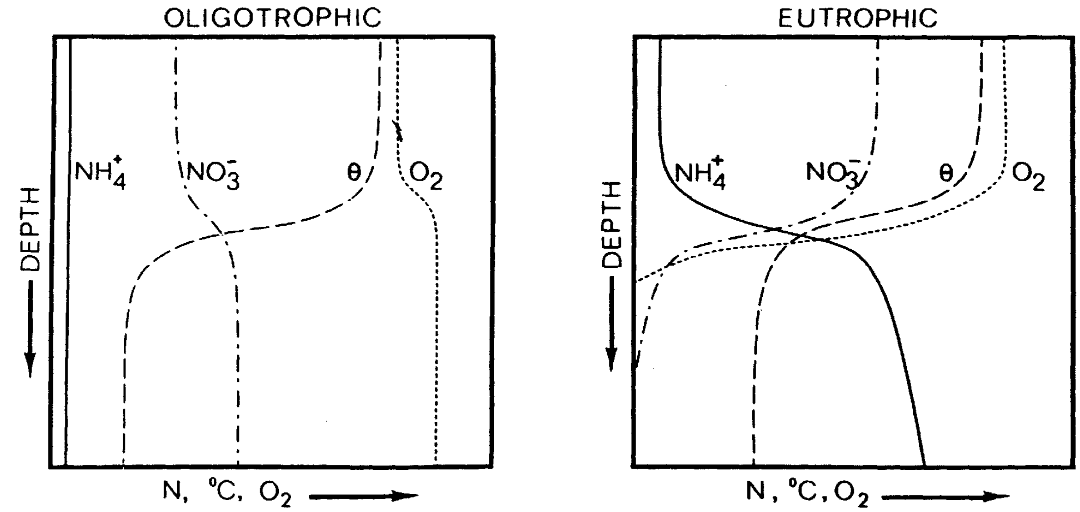
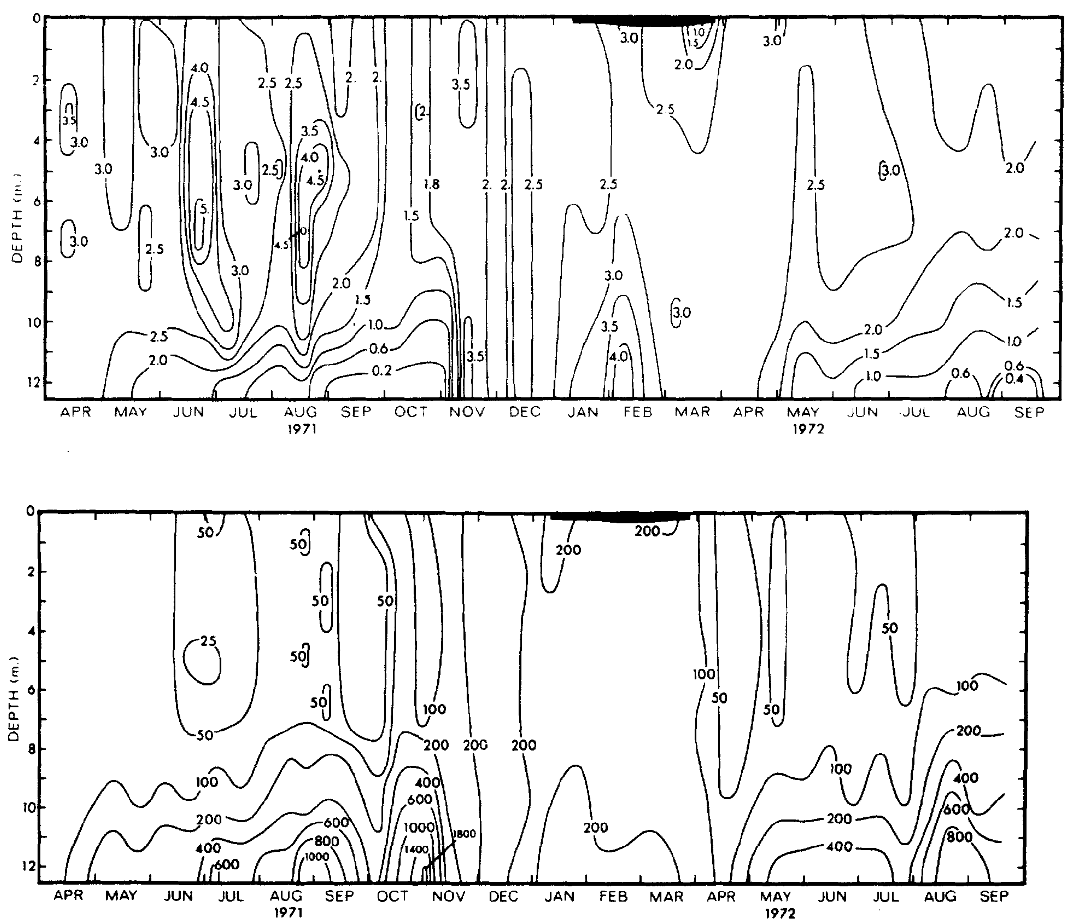


Figure 3-3. Depth-time diagrams of seasonal concentrations of NO3-N + NO2-N in mg/l (upper) and NH4-N in mg/l (lower) in Lawrence Lake, Michigan, 1971-72. Opaque areas represent ice cover (Source: Wetzel 1983)



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 productive) 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 CLASSIFICATION 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 scale, 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 1986). 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 (Appendix E). 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.

4.3 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 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.

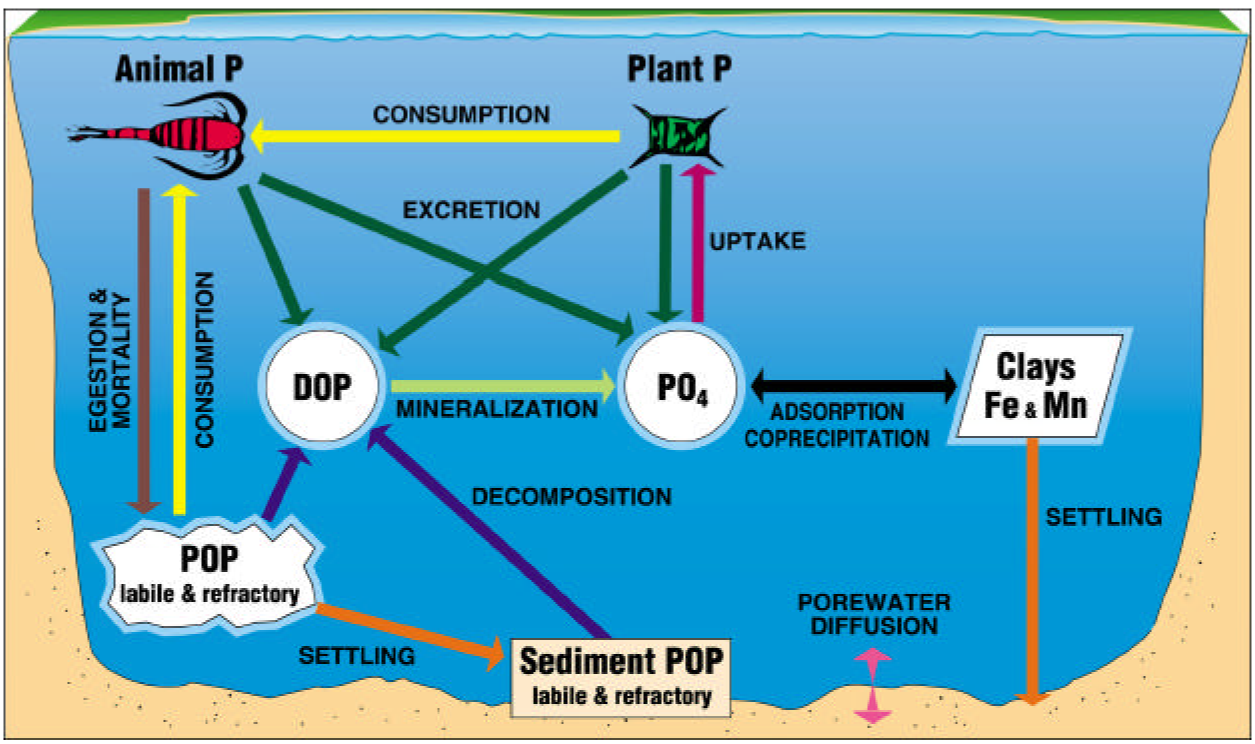
Although only dissolved inorganic nutrients are generally available for algal growth, most lake nutrient criteria are based on total phosphorus and total nitrogen. This is because during periods of high productivity, dissolved nutrients such as phosphorus and nitrogen are often depleted or present at 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 negligible 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 granitic 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.

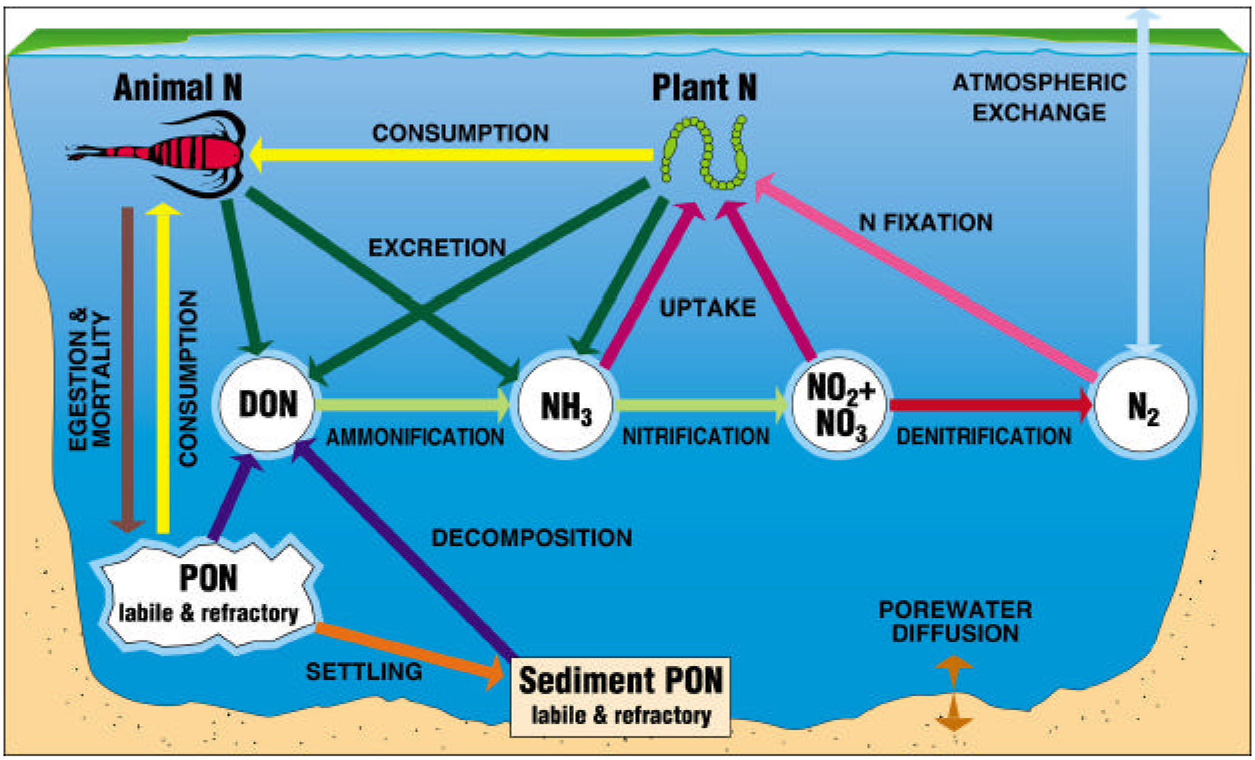
5.1.4 Recommendation

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.

5.2.5 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) considered 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 monograph for use to predict periphyton biomass (chlorophyll 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.

5.2.6 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.

5.3 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

5.4 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.

5.5 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.

5.6 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.

5.6.1 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.

5.6.2 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.

5.6.3 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.

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