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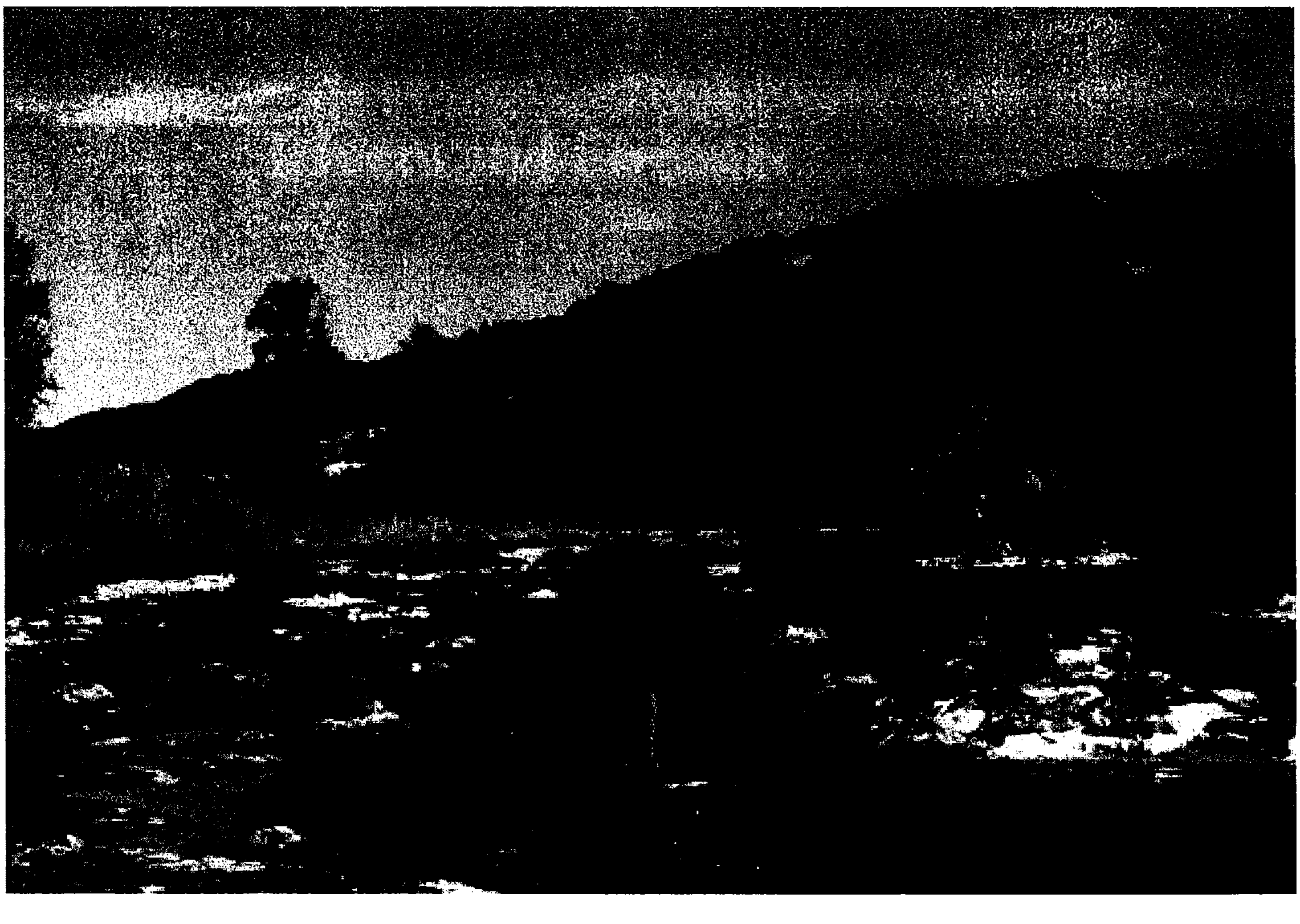
Nutrient Criteria Technical Guidance Manual

Rivers and Streams



DISCLAIMER

This manual provides technical guidance to States, Tribes, and other authorized jurisdictions to establish water quality criteria and standards under the Clean Water Act (CWA), in order to protect aquatic life from acute and chronic effects of nutrient overenrichment. Under the CWA, States and Tribes are required to establish water quality criteria to protect designated uses. State and Tribal decisionmakers retain the discretion to adopt approaches on a case-by-case basis that differ from this guidance when appropriate and scientifically defensible. While this manual constitutes EPA's scientific recommendations regarding ambient concentrations of nutrients that protect resource quality and aquatic life, it does not substitute for the CWA or EPA's regulations; nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, States, Tribes, or the regulated community, and might not apply to a particular situation or circumstance. EPA may change this guidance in the future.

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Cover Photograph: South Umpqua River, Oregon. Photograph courtesy of Dr. E. B. Welch, University of Washington.

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EXECUTIVE SUMMARY

The purpose of this document is to provide scientifically defensible technical guidance to assist States and Tribes in developing regionally-based numeric nutrient and algal criteria for river and stream systems. The Clean Water Action Plan, a presidential initiative released in February 1998, includes an initiative to address the nutrient enrichment problem. Building on this initiative, the EPA developed a report entitled *National Strategy for the Development of Regional Nutrient Criteria* (USEPA 1998). The report outlines a framework for development of waterbody-specific technical guidance that can be used to assess nutrient status and develop regional-specific numeric nutrient criteria. This technical guidance manual builds on the strategy and provides specific guidance for rivers and streams. Similar documents are being prepared for lakes and reservoirs, estuaries and coastal marine waters, and wetlands.

A directly prescriptive approach to nutrient criteria development is not appropriate due to regional differences that exist and the lack of a clear technical understanding of the relationship between nutrients, algal growth, and other factors (e.g., flow, light, substrata). The approach chosen for criteria development must be tailored to meet the specific needs of each State or Tribe. The criteria development process described in this guidance can be divided into the following iterative steps.

1. Identify water quality needs and goals with regard to managing nutrient enrichment problems.
2. Classify rivers and streams first by type, and then by trophic status.
3. Select variables for monitoring nutrients, algae, macrophytes, and their impacts.
4. Design sampling program for monitoring nutrients and algal biomass in rivers and streams.
5. Collect data and build database.
6. Analyze data.
7. Develop criteria based on reference condition and data analyses.
8. Implement nutrient control strategies.
9. Monitor effectiveness of nutrient control strategies and reassess the validity of nutrient criteria.

The components of each step is explained in detail in succeeding chapters of the document.

Chapter 1 addresses the necessity of defining water quality needs and goals for rivers and streams, and gives a general overview of nutrient criteria development. Well-defined needs and goals help to assess the applicability of the criteria development process and identify attainable water quality goals. This step will be revisited throughout the criteria development process to assure defined needs and goals are met.

Chapter 2 discusses classification of streams for water quality assessment and nutrient criteria development. The intent of classification is to identify groups of rivers or streams that have comparable characteristics (i.e., similar biological, ecological, physical, and/or chemical features). Classifying rivers and streams reduces the variability of river-related measures (e.g., physical, biological, or water quality attributes) within classes, maximizes variability among classes, and allows criteria to be identified on a broader rather than site-specific scale. Hence, classification of stream systems will assist in setting appropriate criteria for specific regions and stream system types and provide information used in developing management and restoration strategies.

Chapter 3 describes the candidate variables that can be used to evaluate or predict the condition or degree of eutrophication in a water body. Variables that are required for nutrient criteria development are water column nutrient concentrations (total nitrogen [TN] and total phosphorus [TP]), algal biomass (measured as chlorophyll *a* [chl *a*]), and a measure of turbidity. Measurement of these variables provides a means to evaluate nutrient enrichment and can form the basis for establishing regional and waterbody-specific nutrient criteria. This chapter provides an overview of the required variables and additional variables that can be considered when setting criteria.

Chapter 4 provides technical guidance on designing effective sampling programs. Appropriate data describing stream nutrient and algal conditions are lacking in many areas. Where available data are not sufficient to derive criteria, it will be necessary to collect new data through existing or new monitoring programs. New monitoring programs should be designed to assess nutrient and algal conditions with statistical rigor while maximizing available management resources.

Chapter 5 describes how to build a database of nutrient and algal information. A database of relevant water quality information can be an invaluable tool to States and Tribes as they develop nutrient criteria. Databases can be used to organize existing information, store newly gathered monitoring data, and manipulate data as criteria are being developed. This chapter discusses the role of databases in nutrient criteria development and provides a brief review of existing data sources for nutrient-related water quality information.

Data analysis, described in Chapter 6, is critical to nutrient criteria development. Proper analysis and interpretation of data determines the scientific defensibility and effectiveness of the criteria. The purpose of this chapter is to explore methods for analyzing data that can be used to derive nutrient criteria. Included in this chapter are techniques that link cause and effect relationships between nutrient loading and algal growth, statistical analyses to evaluate compiled data, and use of computer models. Methods of statistical analyses and a review of relevant computer simulation models are provided in appendices.

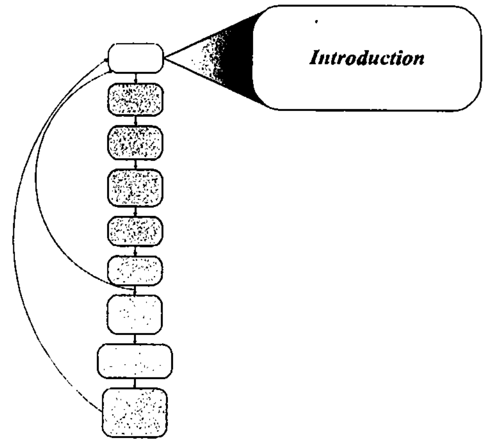
Chapter 7 presents several approaches that water quality managers can use to select numeric criteria for the rivers and streams in their State/Tribal ecoregions. The approaches that are presented include: the use of reference streams, applying predictive relationships to select nutrient concentrations that will result in desirable levels of aquatic growth, and deriving criteria from thresholds established in the literature. Considerations are also presented for those situations in which development of applicable river and stream nutrient criteria might be driven by conditions that are deemed acceptable for downstream receiving waters (i.e., the lake, reservoir, or estuary to which the river drains).

Chapter 8 provides information on regulatory and non-regulatory programs that may be affected by or utilize nutrient criteria. This chapter is intended to serve as an informational resource for water quality managers and foster potential links among regulatory and non-regulatory watershed programs. Information on other agency programs that may assist in implementing criteria and maintaining water quality is included.

Chapter 9 discusses the continued monitoring of river and stream systems to reassess goals and established nutrient criteria. This step should (1) evaluate the appropriateness of the nutrient criteria, (2) ensure that river and stream systems are responding to management action, and (3) assess whether water quality goals established by the resource manager are being met.

Appended to the guidance document are case studies; technical discussions of analytical methods, statistical analyses, and computer modeling; a list of acronyms; and a glossary.

Chapter 1. Introduction



1.1 PURPOSE OF THE DOCUMENT

The purpose of this document is to provide scientifically defensible technical guidance to assist States and Tribes in developing regionally-based numeric nutrient, algal, and macrophyte criteria for river and stream systems. Criteria are "elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use" (USEPA 1994). Water quality criteria are based on scientifically-derived relationships among water constituents and biological condition. "Water quality standards (WQS) are provisions of State or Federal law which consist of a designated use or uses for the waters of the United States, water quality criteria for such waters based upon such uses. Water quality standards are to protect public health or welfare, enhance the quality of the water, and serve the purposes of the Act (40 CFR 131.3)" (USEPA 1994). Water quality standards are comprised of three main components: criteria, which are scientifically based; designated uses, which involve economic, social and political considerations including effects on downstream receiving waters; and an anti-degradation policy, which protects the level of water quality necessary to maintain existing uses (Figure 1).

Water quality can be affected when watersheds are modified by alterations in vegetation, sediment balance, or fertilizer use from industrialization, urbanization, or conversion of forests and grasslands to agriculture and silviculture (Turner and Rabalais 1991; Vitousek et al. 1997; Carpenter et al. 1998). Cultural eutrophication (human-caused inputs of excess nutrients in waterbodies) is one of the primary factors resulting in impairment of U.S. surface waters (USEPA 1996). Both point and nonpoint sources of nutrients contribute to impairment of water quality. Point source discharges of nutrients are fairly constant and are controlled by USEPA National Pollutant Discharge Elimination System (NPDES) permitting (see Section 8.3) [Source: http://www.epa.gov/owm/gen2.htm]. Nonpoint pollutant inputs have increased in recent decades and have degraded water quality in many aquatic systems (Carpenter et al. 1998). Nonpoint sources of nutrients are most commonly intermittent and are usually linked to seasonal agricultural activity or other irregularly-occurring events such as construction or storm events.

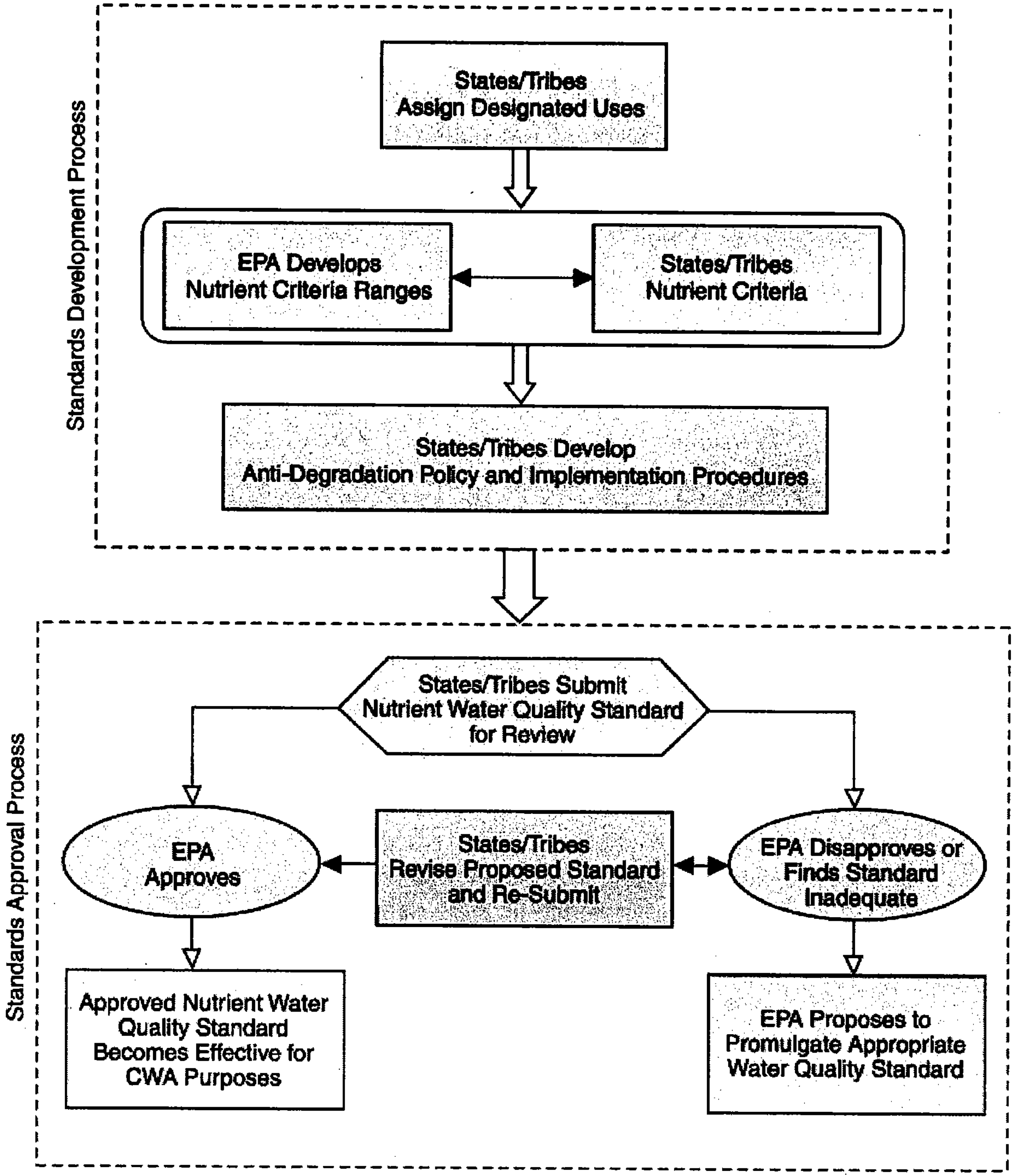


Figure 1. Developing water quality standards for nutrients.

Control of nonpoint source pollutants focuses on land management activities and regulation of pollutants released to the atmosphere (Carpenter et al. 1998).

Control of nutrients is further complicated by the cycling of nitrogen (N) and phosphorus (P) in aquatic systems. Nutrients can be re-introduced into a waterbody from the sediment, or by microbial transformation, potentially resulting in a long recovery period even after pollutant sources have been reduced. In flowing systems, nutrients may be rapidly transported downstream and the effects of nutrient inputs may be uncoupled from the nutrient source, further complicating nutrient source control (Turner and Rabalais 1991; Wetzel 1992; Vitousek et al. 1997; Carpenter et al. 1998). Recognizing cause-and-effect relationships between nutrient input and general waterbody response is the first step in mitigating the effects of cultural eutrophication. Once relationships are established, nutrient criteria can be developed to protect waterbodies. This document describes the process of developing numeric nutrient criteria, a new initiative by the USEPA to address the problem of cultural eutrophication (USEPA 1998a).

The Clean Water Action Plan, a presidential initiative released in February 1998, provides a blueprint for Federal agencies to work with States, Tribes and other stakeholders to protect and restore the Nation's water resources. The Clean Water Action Plan includes an initiative to address the nutrient enrichment problem. Building on this initiative, the USEPA developed a report entitled *National Strategy for the Development of Regional Nutrient Criteria* (USEPA 1998a). The report outlines a framework for development of waterbody-specific technical guidance that can be used to assess nutrient status and develop regional-specific numeric nutrient criteria. This technical guidance manual builds on the strategy and provides specific guidance for rivers and streams. Similar documents are being prepared for lakes and reservoirs, estuaries and coastal marine waters, and wetlands.

For the purposes of this document, river and stream systems are identified collectively as streams or stream systems, unless otherwise noted. Information presented here will provide water quality managers with an overview of the current state of the science, guidance on establishing and compiling a database, and suggested methods for data analyses. The process for setting stream nutrient and algal criteria ranges and a summary of appropriate regulatory and technical considerations are discussed. Diverse geomorphic and climatologic conditions throughout the nation require nutrient and algal criteria development to occur at the ecoregional, State, Tribal, or individual waterbody level to be scientifically valid. The framework for nutrient and algal criteria development follows a logical iterative process that begins with defining goals and needs for State and Tribal water quality. The steps of the process are described in this chapter and detailed in succeeding chapters.

1.2 NUTRIENT ENRICHMENT PROBLEMS IN RIVERS AND STREAMS

Nutrient enrichment frequently ranks as one of the top causes of water resource impairment. Systems are impaired when water quality fails to meet designated use criteria. The USEPA reported to Congress that of the systems surveyed and reported impaired, 40 percent of rivers, 51 percent of lakes, and 57 percent of estuaries listed and nutrients as a primary cause of impairment (USEPA 1996). The nutrient enrichment issue is not new; however, traditional efforts at nutrient control have been only moderately successful. Specifically, efforts to control nutrients in waterbodies that have multiple nutrient sources (point and nonpoint sources) have been less effective in providing satisfactory, timely remedies for enrichment-related problems. The development of numeric criteria should aid control efforts by providing clear numeric goals for nutrient and algal/macrophyte levels. Furthermore, numeric nutrient criteria provide specific water quality goals that will assist researchers in designing improved best management practices.

Nutrient impaired waters can cause problems that range from annoyances to serious health concerns (Dodds and Welch 2000). Nuisance levels of algae and other aquatic vegetation (macrophytes) can develop rapidly in response to nutrient enrichment when other factors (i.e., light, temperature, substrate, etc.) are not limiting. High macrophyte growth can interfere with aesthetic and recreational uses of stream systems (Welch 1992). Algae in particular can grow rapidly when the nutrients N and P (primary nutrients that most frequently limit algal growth, see Section 6.2 Defining the Limiting Nutrient) are abundant, often developing into single or multiple species blooms. Algal bloom development involves complex relationships that are not always well understood. However, the relationship between nuisance algal growth and nutrient enrichment in stream systems has been well-documented in the literature (Welch 1992; Van Nieuwenhuyse and Jones 1996; Dodds et al. 1997; Chetelat et al. 1999). Taste and odor problems in drinking water supplies are usually caused by algal blooms and actinomycete (nitrogen-fixing filamentous bacteria) occurrence and other bacterial blooms that frequently follow (Silvey and Watt 1971; Dorin 1981; Taylor et al. 1981). Algal blooms of certain cyanobacterial species produce toxins that can affect animal and human health. Reports of livestock, waterfowl, and occasionally human poisonings after drinking from waterbodies with blue-green algal blooms are not uncommon (Darley 1982; Carmichael 1986, 1994).

Human health problems can be attributed to nutrient enrichment. One serious human health problem associated with nutrient enrichment is the formation of trihalomethanes (THMs). Trihalomethanes are carcinogenic compounds that are produced when certain organic compounds are chlorinated and bromated as part of the disinfection process in a drinking water treatment facility. Trihalomethanes and associated compounds can be formed from a variety of organic compounds including humic substances, algal metabolites, and algal decomposition products. The density of algae and the level of eutrophication in the raw water supply has been correlated with the production of THMs (Oliver and Schindler 1980; Hoehn et al. 1982).

Effects directly related to nutrients can also result in human health problems. A study of nitrate in groundwater (the primary source of drinking water in the US) indicated that nitrate contamination generally increased with high nitrogen input, greater proportions of well-drained soils, and low woodland to cropland ratios (Nolan et al. 1997). The USEPA has an established maximum contaminant level of 10 mg/L because nitrates in drinking water can cause potentially fatal low oxygen levels in the blood when ingested by infants (USEPA 1995). Nitrate concentrations as low as 4 mg/L in drinking water supplies from rural areas have also been linked to an increased risk of non-Hodgkin lymphoma (Ward et al. 1996). A more detailed discussion of human health concerns related to eutrophication can be found in Suess (1981).

Nutrient impairment can cause problems other than those related to human health. One of the most expensive problems caused by nutrient enrichment is the increased treatment required for drinking water. Nutrient enriched waters commonly cause drinking water treatment plant filters to clog with algae or macrophytes (Welch 1992) and can contribute to the corrosion of intake pipes (Nordin 1985). High algal biomass in drinking water sources require greater volumes of water treatment chemicals, increased back-flushing of filters, and additional settling times to attain acceptable drinking water quality (Nordin 1985).

Adverse ecological effects associated with nutrient enrichment include reductions in dissolved oxygen (DO) and the occurrence of HABs (harmful algal blooms). High algal and macrophyte biomass may be associated with severe diurnal swings in DO and pH in some waterbodies (Wong et al. 1979; Welch 1992; Edmonson 1994; Correll 1998). Low DO can release toxic metals from sediments (Brick and Moore 1996) contaminating habitats of local aquatic organisms. In addition, low DO can cause increased availability of toxic substances like ammonia and hydrogen sulfide, reducing acceptable habitat for most aquatic organisms, including valuable game fish. Decreased water clarity (increased turbidity) can cause loss of macrophytes and creation of dense algal mats. Loss of macrophytes and increased algal biomass may also reduce habitat availability for aquatic organisms. Thus, nutrient enrichment may alter the native composition and species diversity of aquatic communities (Nordin 1985; Welch 1992; Smith 1998; Carpenter et al. 1998; Smith et al. 1999).

A large area (6,000 to 7,000 square miles) of hypoxia–water which contains less than 2 parts per million of DO–located off the Gulf of Mexico Texas-Louisiana Shelf is believed to be caused by a complicated interaction of excessive nutrients transported to the Gulf of Mexico from the Mississippi River drainage; physical changes to the river (e.g., channelization and loss of natural wetlands and vegetation along riverbanks); and the interaction of riverine freshwater with Gulf marine waters (Turner and Rabalais 1994; Rabalais et al. 1996; Brezonik et al. 1999). Hypoxia can cause stress or death in bottom dwelling organisms that cannot move out of the hypoxic zone. Abundant nutrients trigger excessive algal growth which results in reduced sunlight, loss of aquatic habitat, and a decrease in DO. Depletion of DO for the water column has resulted in virtually no biological activity in the hypoxic zone. Reductions in DO have also been implicated in fish kills leading to significant economic impacts on local recreational and commercial fisheries.

Harmful algal blooms (e.g., brown tides, toxic *Pfiesteria piscicida* outbreaks, and some types of red tides) are also associated with excess nutrients. Evidence suggests that nutrients may directly stimulate the growth of the toxic form of *Pfiesteria*, although more research is required to prove this conclusively (Burkholder et al. 1992; Glasgow et al. 1995). *Pfiesteria* has been implicated as a cause of major fish kills at many sites along the North Carolina coast and in several Eastern Shore tributaries of the Chesapeake Bay.

The primary limiting nutrients in freshwaters are phosphorus and nitrogen. Phosphorus is a mineral nutrient, i.e., it is introduced into the biological components of the environment by the breakdown of rock and soil minerals. The breakdown of mineral phosphorus produces inorganic phosphate ions (PO_4^{3-}) that can be absorbed by plants from the soil or water. Phosphorus moves through the food web primarily as organic phosphorus (after it has been incorporated into plant or algal tissue), where it may be released as phosphate in urine or other waste (by heterotrophic consumers) and reabsorbed by plants or algae to start another cycle (Figure 2a) (Nebel and Wright 2000).

The primary reservoir of nitrogen is the air. Plants and animals cannot utilize nitrogen directly from the air, but require nitrogen in mineral form such as ammonium ions (NH_4^+) or nitrate ions (NO_3^-) for uptake. However, a number of bacteria and cyanobacteria (blue-green algae) can convert nitrogen gas to the

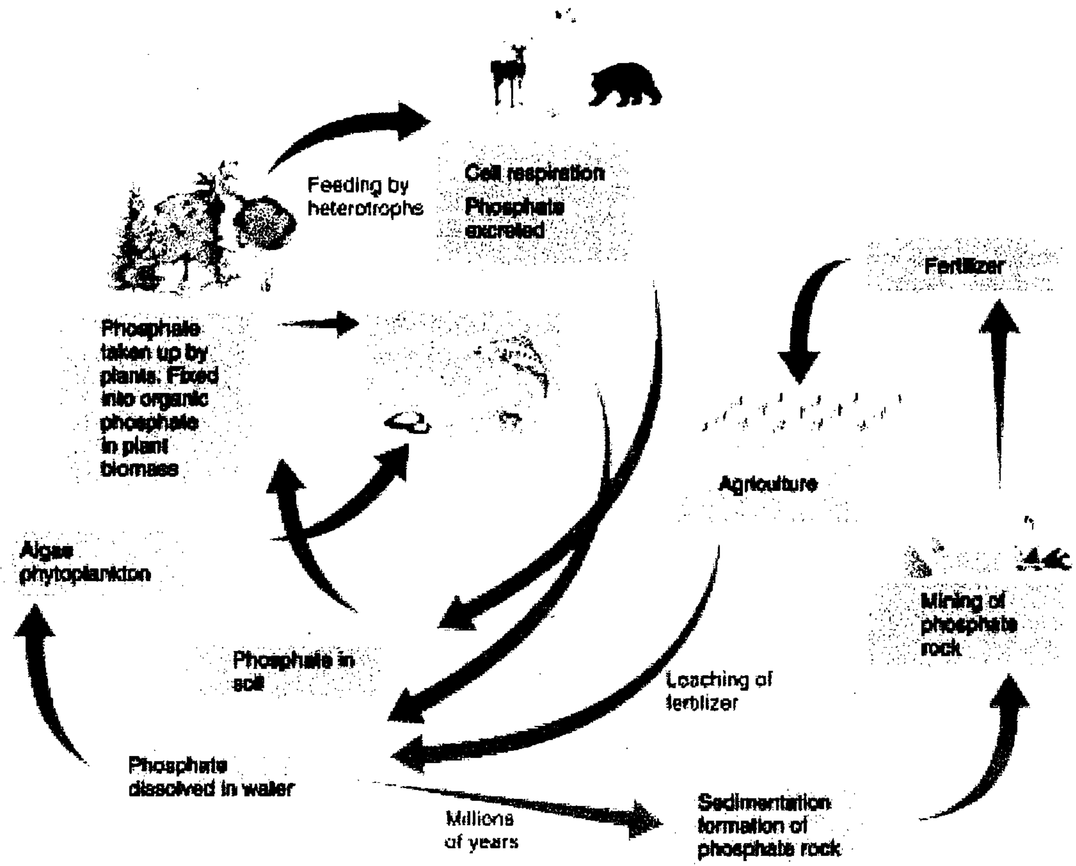


Figure 2a. The phosphorus cycle.

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ammonium form through a process called biological nitrogen fixation. Mineral forms of nitrogen can be taken up by plants and algae, and incorporated into plant or algal tissue. Nitrogen follows the same pattern of food web incorporation as phosphorus, and is released in waste primarily as ammonium compounds. The ammonium compounds are usually converted to nitrates by nitrifying bacteria, making it available again for uptake, starting the cycle anew (Figure 2b) (Nebel and Wright 2000).

Nitrogen and P are transported to receiving waterbodies from rain, overland runoff, groundwater, drainage networks, and industrial and residential waste effluents. Once nutrients have been received in a waterbody they can be taken up by algae, macrophytes and micro-organisms (either in the water column or in the benthos); sorbed to organic or inorganic particles in the water and sediment; accumulated or recycled in the sediment; or transformed and released as a gas from the waterbody (denitrification).

Nitrogen and P have different chemical properties and therefore are involved in different chemical processes. Nitrogen gas dissolved in the water column may be converted to ammonia (a usable form of N) by nitrogen-fixing bacteria and algae when nitrate or ammonia are not readily available. However, receiving waters can lose N through denitrification–anaerobic transformation of nitrate or nitrite into gaseous N oxides (which are released into the air)–mediated by denitrifying bacteria (Atlas and Bartha 1993). Phosphorus is found primarily in two forms, organic and inorganic, in freshwater. The biologically available form of inorganic P in water is orthophosphate (PO_4^{-3}). Most P in surface water is bound organically, and much of the organic P fraction is in the particulate phase of living cells, primarily algae (Wetzel and Likens 1991). The remainder of the organic fraction is present as dissolved and colloidal organic P. Phosphorus readily sorbs to clay particles in the water column reducing availability for uptake by algae, bacteria and macrophytes. The exchange of P between the sediments and overlying water involves net movement of P into the sediments. Exchanges across the sediment interface are regulated by mechanisms associated with mineral-water equilibria, sorption processes, redox interactions, and the activities of bacteria, fungi, algae, and invertebrates. Therefore, P in the sediment is slow to recycle into the water column. Detailed discussions of N and P cycling in freshwater can be found in Wetzel (1983); Goldman and Horne (1983); Atlas and Bartha (1993); and other limnology texts.

Many lakes have been successfully treated for nutrient enrichment problems by an assortment of techniques (Cooke et al. 1993). Lake Washington is a well-recognized example of nutrient diversion. Nutrients were diverted from Lake Washington by eliminating direct discharge from wastewater treatment plants and other dischargers, effectively reducing nuisance algal blooms and improving water clarity (Edmonson 1994). Although many cases have been documented for controlling organic waste inputs to rivers (e.g., the Thames River, England [Goldman and Horne 1983]), nutrient control efforts to correct algal and/or macrophyte problems in streams and rivers have been either minimal or undocumented in the peer-reviewed, published literature. Two well-documented cases are described in detail in Appendix A: the Clark Fork River, MT, and the Bow River, Alberta. Despite these and other efforts, a greater percentage of stream systems surveyed are reported as being nutrient impaired (USEPA 1994; USEPA 1996).

Many States, Tribes, and Territories have adopted some form of nutrient criteria related to maintaining natural conditions and avoiding nutrient enrichment. Most States and Tribes have narrative criteria with no specific numeric criteria. Established criteria most commonly pertain to P concentrations in lakes. Nitrogen criteria, where they have been established, are usually in response to the toxic effects of

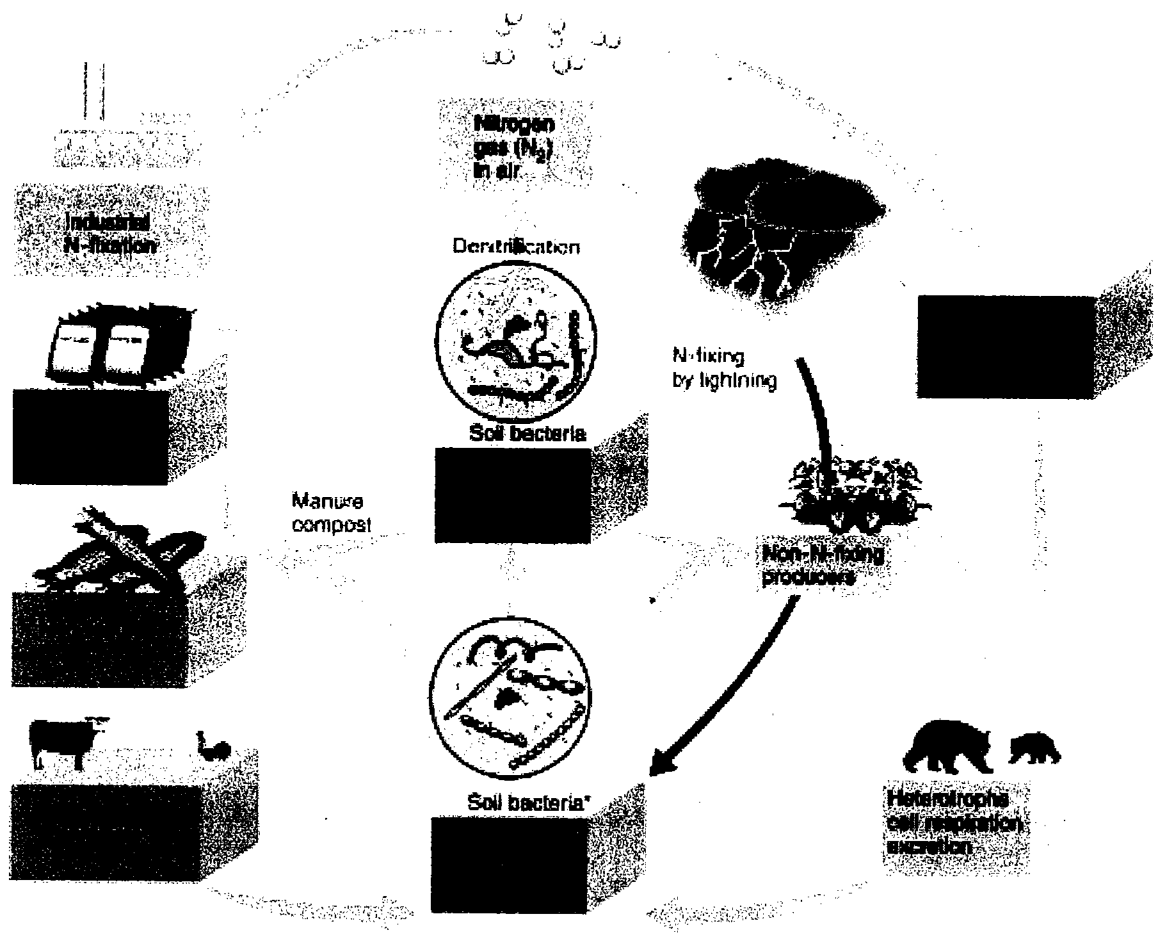

Figure 2b. The nitrogen cycle.

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ammonia and nitrates. In general, levels of nitrates (10 ppm for drinking water) and ammonia high enough to be toxic (1.24 mg N/L at pH = 8 and 25°C) will also cause problems of enhanced algal growth (USEPA 1986).

1.3 WATER QUALITY STANDARDS AND CRITERIA

States and authorized Tribes are responsible for setting water quality standards to protect the physical, biological, and chemical integrity of their waters (Figure 1). "Water quality standards (WQS) are provisions of State or Federal law which consist of a designated use or uses for the waters of the United States, water quality criteria for such waters based upon such uses. Water quality standards are to protect public health or welfare, enhance the quality of the water, and serve the purposes of the Act (40 CFR 131.3)" (USEPA 1994). A water quality standard defines the goals for a waterbody by designating its specific uses, setting criteria to protect those uses, and establishing an antidegradation policy to protect existing water quality. The three main components of water quality standards are based on different concerns: criteria are scientifically based; specific uses involve economic, social and political considerations including the protection of downstream receiving waters; and the anti-degradation policy protects the level of water quality necessary to maintain designated uses (Figure 1). A waterbody can be defined by an existing use (a use actually attained in the waterbody on or after November 28, 1975—the date of the promulgation by USEPA of the first water quality standards regulations) or designated use (a use specified in a water quality standard for each waterbody or segment, regardless of whether it is being attained). An established use cannot be removed unless it is being replaced by one requiring more stringent (protective) criteria. At a minimum, the uses must include recreation in and on the water, and propagation of fish and wildlife (Clean Water Act, Section 101[a] and 303[c]). Other uses, such as boating, cold water fisheries, or drinking water supply, may also be adopted.

Once designated uses of a waterbody have been established, the State or Tribe must adopt numeric or narrative criteria to protect and support the specified uses. Narrative criteria are verbal expressions of desired water quality conditions that are meant to describe the unimpaired condition of a waterbody. A narrative criterion from Vermont is shown below:

> There shall be no increase, in any waters, of total phosphorus above background conditions that may contribute to the acceleration of eutrophication or the stimulation of the growth of aquatic biota in a manner that has an undue adverse effect on any beneficial values or uses of any adjacent or downstream waters.
> (Source: http://www.state.vt.us/wtrboard/rules/vwqs.htm#C1S1)

Numeric criteria, on the other hand, attempt to quantify this ideal by building on and refining narrative criteria. Numeric criteria are values assigned to measurable components of water quality, such as the concentration of a specific constituent that is present in the water column (e.g., average total phosphorus [TP] concentration in a recreational stream shall not exceed 20 µg/L during the growing season). In addition to narrative and numeric criteria, some States and Tribes use numeric goals or assessment levels, an intermediate step between numeric criteria and water quality standards, that are not written into State or Tribal laws but are used internally by the State or Tribal agency for assessment and management purposes.

Numeric criteria can be more useful than narrative criteria in a number of ways. Numeric criteria provide distinct interpretations of acceptable and unacceptable conditions, form the foundation for responsible measurement of environmental quality, and reduce ambiguity for management and enforcement decisions. Despite these advantages, however, most of the Nation's waterbodies do not have numeric nutrient criteria. The lack of numeric criteria makes it difficult to assess the condition of rivers and streams and develop protective water quality standards, hampering the water quality manager's ability to implement management strategies.

Setting numeric nutrient criteria can provide a variety of benefits. For example, information obtained from compiling existing data and conducting new surveys can provide water quality managers and the public a better perspective on the condition of State and Tribal waters. The compiled waterbody information can be used to most effectively budget personnel and financial resources for the protection and restoration of river and stream systems. In a similar manner, data collected in the criteria development and implementation process can be compared before, during, and after specific management actions. Analyses of these data can determine the response of the waterbody and the effectiveness of management endeavors.

Nutrient criteria also support watershed-protection activities. Nutrient criteria can be used in conjunction with State/Tribal and Federal biocriteria surveys, National Estuary Program and Clean Lakes projects, and in development of TMDLs (Total Maximum Daily Loads) to improve resource management at local, State, Tribal, and national levels.

1.4 OVERVIEW OF THE CRITERIA DEVELOPMENT PROCESS

This section describes the five general elements of nutrient criteria development outlined in the National Strategy (USEPA 1998a) and is followed by a detailed overview of the steps taken to derive nutrient criteria for river and stream systems. A prescriptive approach is not appropriate due to regional differences that exist and the scientific community's limited technical understanding of the relationship between nutrients, algal growth, and other factors (e.g., flow, light, substrata). The approach chosen for criteria development must be tailored to meet the specific needs of each State or Tribe.

The USEPA has adopted the following principal elements as part of its *National Strategy for the Development of Regional Nutrient Criteria* (USEPA 1998a). This document can be downloaded in PDF format at the following website: www.epa.gov/OST/standards/nutrient.html.

1. Ecoregional nutrient criteria will be developed to account for the natural variation existing within various parts of the country. Different waterbody processes and responses dictate that nutrient criteria be specific to the waterbody type. No single criterion will be sufficient for each waterbody, therefore we anticipate system classification within waterbody type for appropriate criteria derivation (see Section 1.5, item 2).

2. Guidance documents for nutrient criteria will provide methodologies for developing nutrient criteria for four primary variables (total nitrogen [TN], TP, chlorophyll *a* [chl *a*], and a measure of turbidity) by ecoregion and waterbody type.

3. Regional Nutrient Coordinators will lead State/Tribal technical and financial support operations used to compile data and conduct environmental investigations. A team of agency specialists from USEPA Headquarters will provide technical and financial support to the Regions, and will establish and maintain communications between the Regions and Headquarters.

4. Nutrient criteria numeric ranges, developed at the national level from existing databases and additional environmental investigations, will be used to derive specific criterion values. Criteria values will be implemented into water quality standards by States and Tribes within three years of criteria publication. Ecoregional nutrient criteria will be used by States and Tribes either as a point of departure for the development of more refined criteria, or as numeric criteria. The USEPA will promulgate nutrient criteria in the absence of State or Tribal criteria development initiatives.

5. Nutrient and algal criteria will serve as benchmarks for evaluating the relative success of any nutrient management effort, whether protection or remediation. Criteria will be re-evaluated periodically to assess whether refinements or other improvements are needed.

Nutrient criteria will form the basis for regulatory values such as standards, NPDES permit limits, and TMDL values. Nutrient criteria will also be valuable as decision making benchmarks for management planning and assessment. The development of TMDLs may serve as an intermediate step between criteria development and watershed-based management planning.

The USEPA Strategy envisions a process by which State/Tribal waters are initially measured, reference conditions are established, individual waterbodies are compared to reference waterbodies, and appropriate management measures are implemented. This process is outlined in detail below.

1.5 THE CRITERIA DEVELOPMENT PROCESS

Figure 3 presents a flow chart of the nine key steps involved in the criteria development process. A brief discussion of each of the steps involved, and what ideally is accomplished at each stage, is given below:

1. Identify water quality needs and goals with regard to managing nutrient enrichment problems. State and Tribal water quality managers should define the water quality needs and goals for their rivers and streams. Well-defined needs and goals will help in assessing the success of the criteria development process, and will identify attainable water quality goals. This step should be revisited throughout the criteria development process to assure defined needs and goals are addressed.

2. Classify rivers and streams first by type, and then by trophic status. The intent of classification is to identify groups of stream systems that have comparable characteristics (i.e., biological, ecological, physical, chemical features). Classifying rivers and streams reduces the variability of stream-related measures (e.g., physical, biological, or water quality attributes) within classes and maximizes variability among classes. Classification will allow criteria to be identified on a broader rather than site-specific scale.

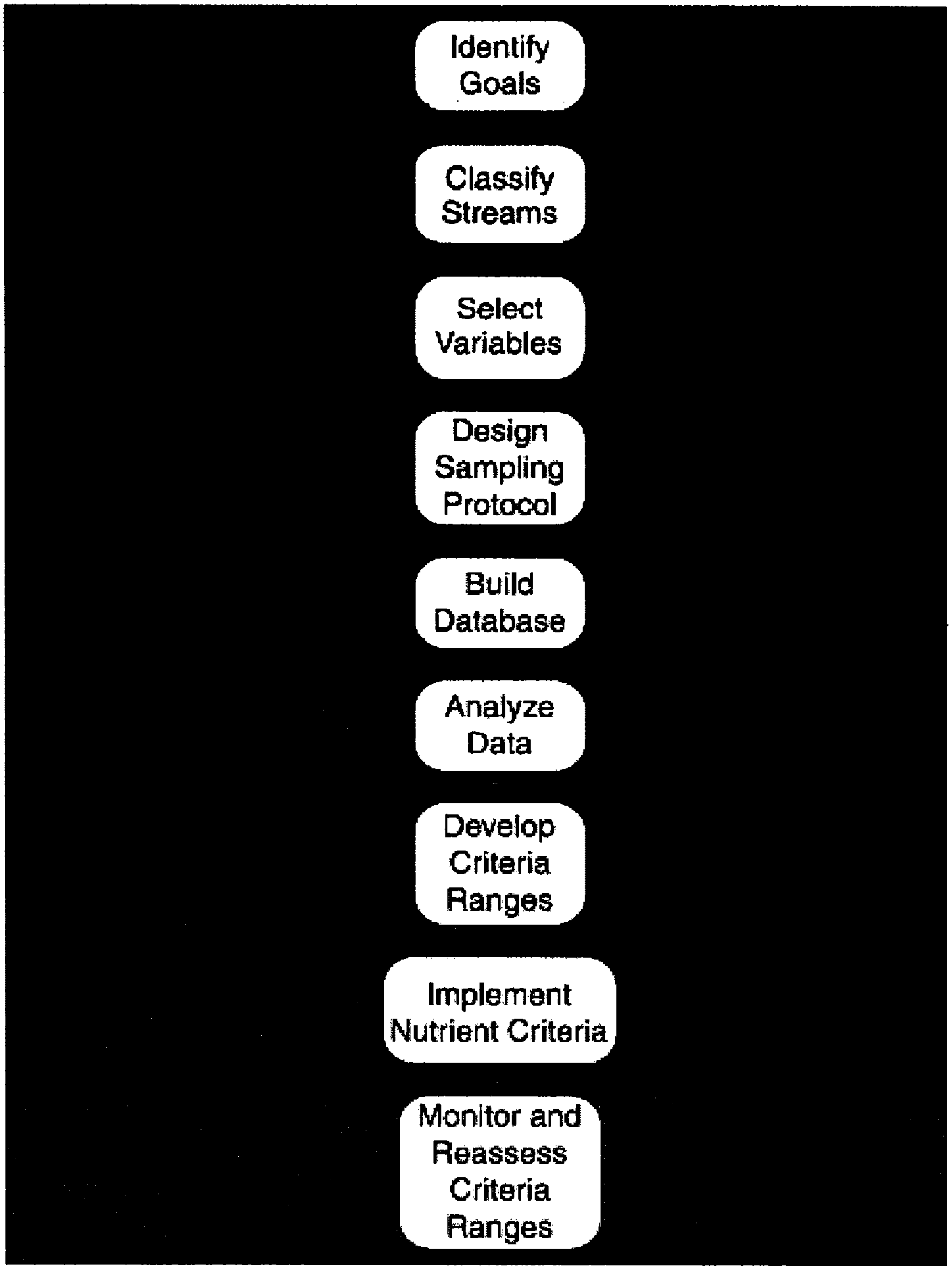


Figure 3. Criteria development flow chart.

3. Select variables for monitoring nutrients. Variables, in the context of this document, are measurable attributes that can be used to evaluate or predict the condition or degree of eutrophication in a water body. Four primary water quality variables that must be addressed are TN, TP, chl *a* as an estimate of algal biomass, and turbidity (see Section 3.2). Measurement of these variables provides a means to evaluate nutrient enrichment and can form the basis for establishing regional and waterbody-specific nutrient criteria. Additional secondary variables may also be monitored.

4. Design a sampling program for monitoring nutrients and algal biomass in rivers and streams. New monitoring programs should be designed to identify statistically significant differences in nutrient and algal conditions while maximizing available management resources (see Section 4.2). Initial monitoring efforts should focus on targeting reference stream reaches that can be used to assess impairment by nutrients and algae.

5. Collect data and build database. Potential sources of additional data for nutrient criteria development include current and historical water quality monitoring data from Federal, State, and local water quality agencies; university studies; and volunteer monitoring information. Databases can be used to organize existing data, store newly gathered monitoring data, and manipulate data as criteria are being developed. The USEPA is developing a national relational database for State/Tribal users to store, screen, and manipulate nutrient-related data.

6. Analyze data. Statistical analyses are used to interpret monitoring data for criteria development. Nutrient criteria development should relate nutrient concentrations in streams, algal biomass, and changes in ecological condition (e.g., nuisance algal accrual rate and deoxygenation). In addition, the relative magnitude of an enrichment problem can be determined by examining total nutrient concentration and chl *a* frequency distributions for stream classes. These analyses provide water quality managers with a tool for measuring the potential extent of overenrichment.

7. Develop criteria based on reference conditions and data analyses. Criteria selected must first meet the optimal nutrient condition for that stream class and second be reviewed to ensure that the level proposed does not result in adverse nutrient loadings to downstream waterbodies.

Three general approaches for criteria setting are discussed in this manual: (1) identification of reference reaches for each stream class based on best professional judgement (BPJ) or percentile selections of data plotted as frequency distributions, (2) use of predictive relationships (e.g., trophic state classifications, models, biocriteria), and (3) application and/or modification of established nutrient/algal thresholds (e.g., nutrient concentration thresholds or algal limits from published literature).

Initial criteria should be verified and calibrated by comparing criteria in the system of study to nutrients, chl *a*, and turbidity values in waterbodies of known condition to ensure that the system of interest operates as expected. **A weight of evidence approach that combines any or all of the three approaches above will produce criteria of greater scientific validity.** Selected criteria and the data analyzed to identify these criteria will be comprehensively reviewed by a panel of specialists in each USEPA Region. Calibration and review of criteria may lead to refinements of either derivation techniques or the criteria themselves. In some instances empirical and simulation modeling, or data sets from adjacent States/Tribes with similar systems may assist in criteria derivation and calibration.

8. Implement nutrient control strategies. Much of the management work done by USEPA and the States and Tribes is regulatory. Nutrient criteria can be incorporated into State/Tribal standards as enforceable tools to preserve water quality. As an example, nutrient criteria values can be included as limits in NPDES permits for point source discharges. The permit limits for N, P and other trace nutrients emitted from wastewater treatment plants, factories, food processors and other dischargers can be appropriately adjusted and enforced in accordance with the criteria.

In addition, watershed source reduction responsibilities, especially with respect to nonpoint sources, can be established on the basis of nutrient criteria. Resource managers can use nutrient criteria to help define source load allocations for a watershed. Once sources have been identified, resource managers can begin land use improvements and other activities necessary to maintain or improve the system. System improvements from a watershed perspective must proceed on a reasonable scale so that protection and restoration can be achieved.

9. Monitor effectiveness of nutrient control strategies and reassess the validity of nutrient criteria. Nutrient criteria can be applied to evaluate the relative success of management activities. Measurements of nutrient enrichment variables in the receiving waters preceding, during, and following specific management activities, when compared to criteria, provide an objective and direct assessment of the success of the management project.

Throughout the continuing process of problem identification, response and remediation, and evaluation to protect and enhance our water resources, States, Tribes, and the USEPA are called upon by the U.S. Congress to periodically report on the status of the Nation's waters (Section 305 [b] of the Clean Water Act as amended). Establishment of nutrient criteria will add two causal and two response parameters (see Sections 3.2 - 3.3) to the measurement process required for the biannual Report to Congress. These measurements can be used to document change and monitor the progress of nutrient reduction activities.

The chapters that follow present detailed information that elaborates upon this outline of nutrient criteria development. For some water quality managers, components of certain criteria development steps may already be completed for existing stream monitoring programs (e.g., sampling design for specific stream systems). Thus, some steps can be excluded as the manager advances further through the process. However, should new or revised monitoring programs be envisioned, review by the water quality manager of each of the steps outlined in this guidance is recommended.

Although this document is meant to provide the best available scientific procedures and approaches for collecting and analyzing nutrient-related data, including examination of nutrient and algal relationships, a comprehensive understanding of nutrient and algal dynamics within all types of stream systems is beyond the current state of scientific knowledge. The National Nutrient Program represents a new effort and approach to criteria development that, in conjunction with efforts made by State and Tribal water quality managers, will ultimately result in a heightened understanding of nutrient/algal relationships. As the proposed process is put into use to set criteria, program success will be gauged over time through evaluation of management and monitoring efforts. A more comprehensive knowledge base pertaining to nutrient and algal relationships will be expanded as new information is gained and obstacles overcome, justifying potential refinements to the criteria development process described here.

1.6 IDENTIFY NEEDS AND GOALS

The overarching goal of developing nutrient criteria is to ensure the quality of our national waters. Ensuring water quality may include restoration of impaired systems, conservation of high quality waters, and protection of systems at high risk for future impairment. The goals of a State or Tribal water quality program will be defined differently based on the needs of each State or Tribe, but should, at a minimum, protect established designated uses for the waterbodies within State or Tribal lands. Once goals and objectives are defined, they should be revisited regularly to evaluate progress and assess the need for refinements or revisions.

The first task of a water quality manager is to set a water quality goal, such as "no nuisance algal blooms such that swimming is restricted during summer months." After such a goal is established, managers must develop a timeline, budget, and plan of action for accomplishing this goal. Needs of the program, such as funding, acquiring relevant data, and assigning employee responsibilities must be addressed. Well-defined needs and goals will help in assessing the success of the criteria development process and will identify attainable water quality goals.

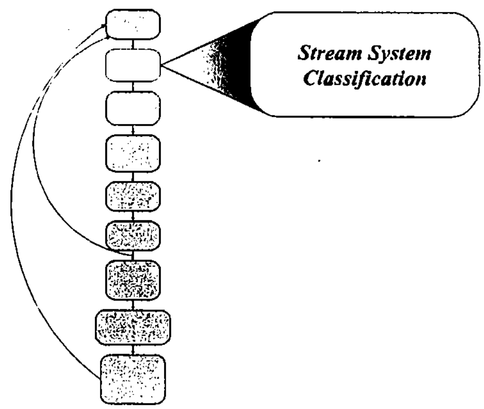
1.7 DOCUMENT STRUCTURE

This manual comprises nine chapters that formulate the steps recommended for nutrient criteria development. The first step of the process, identifying goals and objectives, is unique to each water quality manager and should be revisited regularly to evaluate progress and assess the need for goal and/or objective refinements or revisions. The next step entails stream classification based on physical and nutrient gradient factors (Chapter 2). Sampling variables, including primary and appropriate secondary variables (Chapter 3), should be selected for monitoring efforts. Once these variables are determined, sampling designs for new monitoring programs can be developed (Chapter 4). Chapter 5 discusses potential data sources that can be used by water quality managers to develop criteria and addresses the usefulness of databases in compiling, storing, and analyzing data. A variety of data analysis methods and techniques used to derive criteria are presented in Chapters 6 and 7, respectively. These two chapters are meant to provide water quality managers with a range of options that may be useful for deriving criteria. Nutrient management programs (including nutrient control strategies for point and nonpoint sources) and points of contact or references that may be useful to water quality managers are provided in Chapter 8. Chapter 9 concludes the criteria development process with a brief discussion of continued monitoring and reassessment of goals and the established criteria.

It should be noted that completion of *each* previously described step may *not* be required by all water quality managers. Many State or Tribal water quality agencies already have established stream classes, monitoring programs, and/or databases for their programs and therefore can bypass those steps. This manual is meant to be comprehensive in the sense that all of the criteria development steps are described; however, the process can be adapted to suit existing water quality programs.

Appendix A of the manual contains five case studies: (1) Tennessee ecoregion streams (southeastern U.S.), (2) Clark Fork River (western forested mountains), (3) upper Midwest river basins (prairie-agricultural river systems), (4) Bow River (northern Rockies), and (5) desert streams (arid western U.S.). These case studies are meant to characterize some of the variability observed within North American stream systems and region-specific issues that should be considered when developing nutrient criteria. Appendices B and C provide water quality managers with information and additional references for laboratory/field methods and statistical tests/modeling tools, respectively. Appendix D defines frequently used acronyms and technical terms found throughout the document.

Chapter 2. Stream System Classification



2.1 INTRODUCTION

This chapter discusses classification of streams for water quality assessment and nutrient criteria development. The purpose of classification is to identify groups of rivers or streams that have comparable characteristics (i.e., similar biological, ecological, physical, and/or chemical features) so that data may be compared or extrapolated within stream types. This chapter focuses on providing water quality managers with a menu of tools that can be used to classify the stream system of interest, resulting in different aggregations of physical parameters that correlate with water quality variables.

Classifying rivers and streams reduces the variability of stream-related measures (e.g., physical, biological, or water quality variables) within identified classes and maximizes inter-class variability. Classification schemes based on non-anthropogenic factors such as parent geology, hydrology, and other physical and chemical attributes help identify variables that affect nutrient/algal interactions. Classification can also include factors that are useful when creating nutrient control strategies such as land use characteristics, bedrock geology, and identification of specific point and nonpoint nutrient sources. Grouping streams with similar properties will aid in setting criteria for specific regions and stream system types, and can provide information used in developing management and restoration strategies.

A two-phased approach to system classification is prescribed here. Initially, stream classification is based primarily (though not exclusively) on physical parameters associated with regional and site-specific characteristics, including climate, geology, substrate features, slope, canopy cover, retention time of water, discharge and flow continuity, system size, and channel morphology. The second phase involves further classifying stream systems by nutrient gradient (based upon measured nutrient concentrations and algal biomass). Trophic state classification, in contrast, focuses primarily on chemical and biological parameters including concentrations of nutrients, algal biomass as chlorophyll *a*, and turbidity, and may also include land use and other human disturbance parameters. The additional sub-classification of streams by nutrient condition, in conjunction with an understanding of dose-response relationships between algae and nutrients, helps define the goals for establishing nutrient criteria.

The physical and nutrient characterization discussed above can often be complemented by designated use classifications. These are socially-based classifications developed in accordance with EPA policy and based on the predominant human uses that a State or Tribe has concluded are appropriate for a particular stream or river. Water quality standards, predicated on criteria, are applied to these designated use classifications and are enforceable to protect specified uses. Uses are designated in accordance with relative water quality condition and trophic state. For more information on designated use classifications and their relationship to water quality criteria and standards, see the USEPA Water Quality Standards Handbook (USEPA 1994).

Stream classification requires consideration of stream types at different spatial scales. Drainage basins can be delineated and classified at multiple spatial scales ranging from the size of the Mississippi River basin to the few square meters draining into a headwater stream. The general approach is to establish divisions at the largest spatial scale (river basins of the continent), and then to continue stratification at smaller scales to the point at which variability of algal-nutrient relationships is limited within specific stream classes.

The highest level of classification at the national level is based on geographic considerations. The Nation has been divided into 14 nutrient ecoregions (Omernik 2000) based on landscape-level geographic features including climate, topography, regional geology and soils, biogeography, and broad land use patterns (Figure 4). The process of identifying geographic divisions (i.e., regionalization) is part of a hierarchical classification procedure that aggregates similar stream systems together to prevent grouping of unlike streams. The process of subdividing the 14 national ecoregions should be undertaken by the State(s) or Tribe(s) within each of those ecoregions. Classification of State/Tribal lands invariably involves the professional judgement of regional experts. Experts familiar with the range of conditions in a region can help define a workable system that clearly separates different ecosystem types, yet does not consider each system a special case.

The usefulness of classification is determined by its practicality within the region, State, or Tribal lands in which it will be applied; local conditions determine the appropriate classes. In this Chapter, a regionalization system derived at the national level is presented. This system provides the framework from which State and Tribal water resource management agencies can work to establish appropriate subdivisions. In addition, different classification schemes are presented to provide resource managers with information to use in choosing a stream classification system. It is the intent of this document to provide adequate flexibility to States and Tribes in identifying State and Tribal-specific subregions.

The following sections describe specific examples of first-phase physical classification based on variation in natural characteristics and secondly, nutrient gradient classification schemes for identifying similarities within stream system types. Each classification method is presented and the rationale for its use is provided.

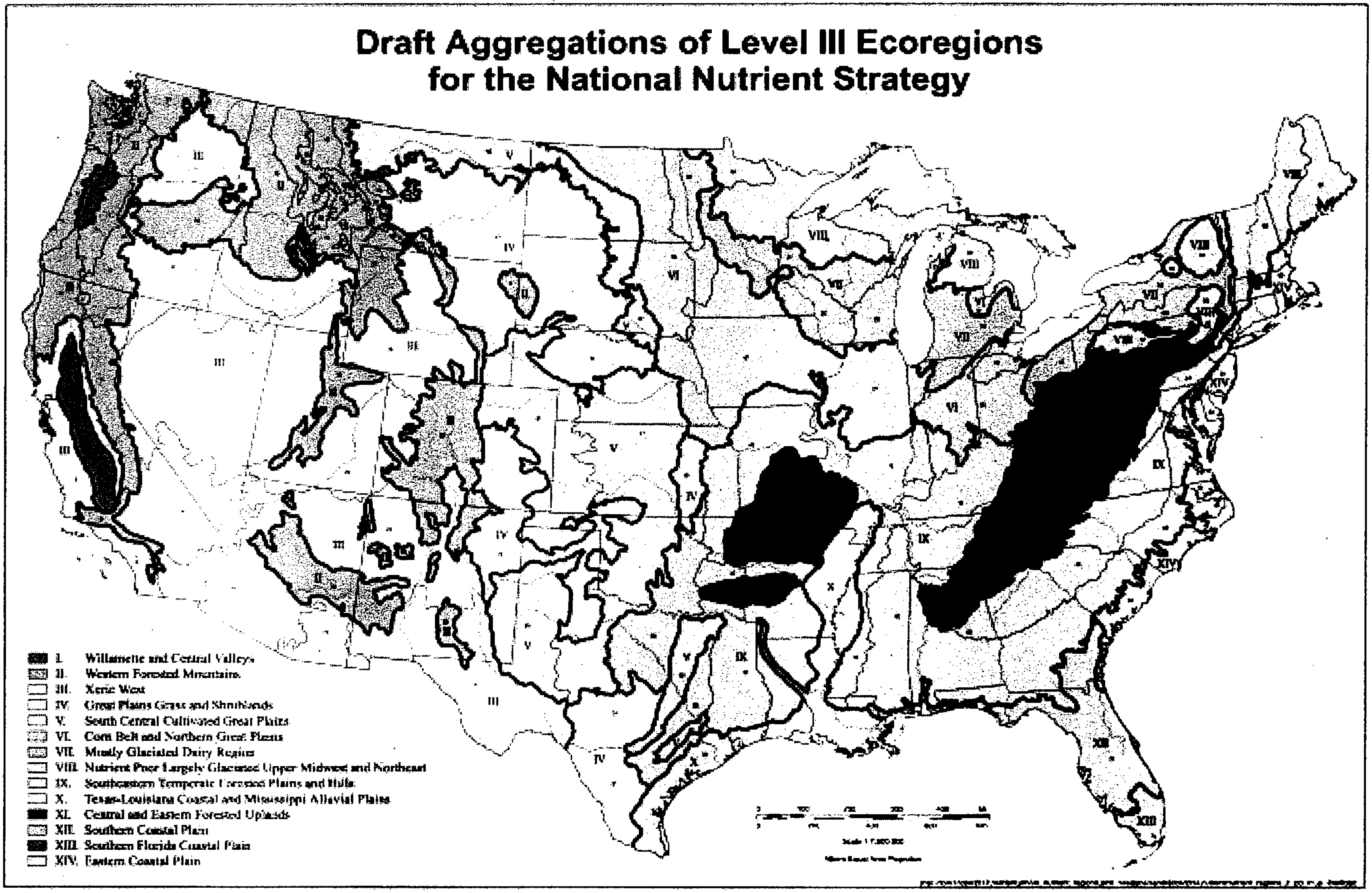


Figure 4. Fourteen nutrient ecoregions as delineated by Omernik (2000). Ecoregions were based on geology, land use, ecosystem type, and nutrient conditions.

2.2 CLASSIFICATION SCHEMES BASED ON PHYSICAL FACTORS

The classification systems described in the following sections (including ecoregional, fluvial geomorphological, and stream order classification schemes) are based on physical stream and watershed characteristics. Stream systems are characterized by the continual downstream movement of water, dissolved substances, and suspended particles. These components are derived primarily from the land area draining into a given channel or the drainage basin (watershed). The climate, geology, and vegetational cover of the watershed are reflected in the hydrological, biological, and chemical characteristics of the stream. Therefore, factors such as general land use, climate, geology and general hydrological properties must be considered regardless of the method of classification used. As described above, the initial classification should be based on physical characteristics of parent geology, elevation, slope, hydrology and channel morphology. Hydrologic disturbance frequency and magnitude are also important when classifying stream systems.

In addition to classification of stream systems, factors contributing to trophic state and macrophyte and algal growth should be considered. Table 1 presents several factors that affect periphyton and plankton biomass levels in stream systems. Macrophyte-dominated systems could occur under conditions similar to those favorable for high periphyton biomass (Table 1), if the velocity is low and the substrate includes organic sediment. Macrophytes are generally unlikely to develop in systems where the stream bottom is composed primarily of gravel or other large substrata (Wong and Clark 1979). The following section specifically addresses the potential effects of hydrology and channel morphology, flow, and parent geology on algal and macrophyte growth within stream systems.

River and stream types (and reaches within these waterbodies) are too diverse to set one criterion for all stream/river types. However, it is not necessarily feasible or recommended to develop site-specific criteria for every stream reach within the U.S. Morphological and fluvial characteristics of a stream influence many facets of its behavior. Streams with similar morphologies may have similar nutrient capacities or similar responses to nutrient loadings. Rivers and streams are very diverse within ecoregions. Reaches within one stream can have a distinct morphology. The geomorphology of a river or stream – its shape, depth, channel materials – affects the way that waterbody receives, processes, and distributes nutrients. Nutrient cycling processes that occur upstream affect communities and processes downstream by altering the form and concentration of nutrients and organic matter in transport (nutrient spiraling); these effects can be further intensified by patch dynamics (Mulholland et al. 1995). The spatial scales which most influence upstream-downstream linkages are the geomorphology-controlled patterns observed at the landscape scale and the nutrient-cycling-controlled patterns observed at the stream reach scale (Mulholland et al. 1995). Therefore, to set appropriate criteria for rivers and streams in an ecoregion, streams must be classified by their morphological characteristics at both the landscape and stream reach scale, with an emphasis on those characteristics most likely to affect nutrient cycling.

ECOREGIONAL CLASSIFICATION

Ecoregions are based on geology, soils, geomorphology, dominant land uses, and natural vegetation (Omernik 1987; Hughes and Larsen 1988) and have been shown to account for variability of water quality and aquatic biota in several areas of the United States (e.g., Heiskary et al. 1987; Barbour et al. 1996). On a national basis, individual streams and rivers are affected by varying degrees of development, and user perceptions of acceptable water quality can differ even over small distances.

Table 1. Geological, physical, and biological habitat factors that affect periphyton and phytoplankton biomass levels in rivers and streams given adequate to high nutrient supply and non-toxic conditions. Note that only one factor is sufficient to limit either phytoplankton or periphyton biomass.

Phytoplankton-Dominated Systems	**Periphyton-Dominated Systems**
High Phytoplankton Biomass · low current velocity(< 10 cm/s)/long detention time (>10 days) and · low turbidity/color and · open canopy and · greater stream depth and · greater depth to width ratio	High Periphyton Biomass · high current velocity (>10 cm/s) and · low turbidity/color and · open canopy and · shallow stream depth and · minimal scouring and · limited macroinvertebrate grazing and · gravel or larger substrata and · smaller depth to width ratio
Low Phytoplankton Biomass · high current velocity (>10 cm/s)/short detention time (<10 days) and/or · high turbidity/color and/or · closed canopy and/or · shallow stream depth	Low Periphyton Biomass · low current velocity (< 10 cm/s) and/or · high turbidity/color and/or · closed canopy and/or · greater stream depth and/or · high scouring and/or · high macroinvertebrate grazing and/or · sand or smaller substrata

Ecoregions are generally defined as relatively homogeneous areas with respect to ecological systems and the interrelationships among organisms and their environment (Omernik 1995). Ecoregions can occur at various scales; broad-scale ecoregions may include the glaciated corn belt of the central and upper Midwest or the arid to semi-arid basin and desert regions of the southwest. At more refined scales, regions within the broader regions can be identified.

Ecoregions serve as a framework for evaluating and managing natural resources. The ecoregional classification system developed by Omernik (1987) is based on multiple geographic characteristics (e.g., soils, climate, vegetation, geology, land use) that are believed to cause or reflect the differences in the mosaic of ecosystems. Omernik's original compilation of national ecoregions was based on a fairly coarse (1:7,500,000) scale that has subsequently been refined for portions of the southeast, mid-Atlantic, and northwest regions, among others (Omernik 1995). The process of defining subregions within an ecoregion requires collaboration with State/Tribal scientists and resource managers. Once appropriate subregions are delineated, reference sites can be identified (see Section 4.2). Similar to the process described for ecoregion refinement, reference site selection involves interactions with scientists and water quality managers that understand local conditions. Field verification techniques, methods for selecting reference sites for small and/or disjunct subregions can be found in Omernik (1995).

FLUVIAL GEOMORPHOLOGY

Fluvial geomorphology mechanistically describes river and slope processes on specific types of landforms, i.e., the explanation of river and slope processes through the application of physical and chemical principles. The morphology of the present-day channel is governed by the laws of physics through observable stream channel features and related fluvial processes. Stream pattern morphology is directly influenced by eight major variables including channel width, depth, velocity, discharge, channel slope, roughness of channel materials, sediment load and sediment size (Leopold et al. 1964). A change in one variable causes a series of channel adjustments which lead to changes in the other variables, resulting in channel pattern alterations. Many stream classification systems, have a fluvial geomorphologic component.

ROSGEN

The stream classification method devised by David Rosgen is a comprehensive guide to river and stream classification (see Rosgen 1994 or 1996). The Rosgen classification system is currently utilized by several States. This system integrates fluvial geomorphology with other stream characteristics. Specifically, Rosgen combines several methods of stream classification into one complete, multi-tiered approach. Rosgen's method has four levels of detail: broad morphological (geomorphic) characterization, morphological description (stream types), stream "state" or condition, and verification. Level I classification, geomorphic characterization, takes into account channel slope (longitudinal profile), shape (plan view morphology, cross-sectional geometry), and patterns. Level I streams are divided into seven major categories and labeled A-G. The Level II morphological delineative criteria include landform/soils, entrenchment ratio, width/depth ratio, sinuosity, channel slope, and channel materials. The 42 subcategories of Level II streams are labeled with a letter and a number, A1-G6 (see Rosgen 1994, 1996). Level III designations are primarily used in specific studies or in restoration projects to assess the quality and/or progress of a specific reach. Level IV classifications may be used to verify results of specific analyses used to develop empirical relationships (such as a roughness coefficient) (Rosgen 1996).

Rivers and streams are complicated systems. A classification scheme is an extreme simplification of the geomorphic and fluvial processes. However, the Rosgen system of classification is a useful frame of reference to :

1. Predict a river's behavior from its appearance;
2. Develop specific hydraulic and sediment relations for a given morphological channel type and state;
3. Provide a mechanism to extrapolate site-specific data collected on a given stream reach to those of similar character; and
4. Provide a consistent and reproducible frame of reference of communication for those working with river systems in a variety of professional disciplines (Rosgen 1994).

Classification of streams and rivers allows comparisons and extrapolation of data from different streams or rivers in an ecoregion. Comparing similar streams may help to predict the behavior of one stream based data and observations from another. *Applied River Morphology* (Rosgen 1996) contains in-depth descriptions of each Level II stream type (A1-G6) and includes photographs and illustrations. Rosgen discusses theoretical characterizations and variables and provides field methods for delineating stream types. The Rosgen classification system may be more detailed than needed for many States and Tribes. For more information on the Rosgen classification system, see Rosgen (1996).

STREAM ORDER

Identifying stream orders in a given delineated watershed can provide a classification system for monitoring streams. A variety of methods have been proposed for ordering drainage networks for stream classification and monitoring. The Horton-Strahler method (Horton 1945; Strahler 1952) is most widely used in the US. Each headwater stream is designated as a first order stream. Two first order streams combine to produce a second order stream, two second order streams combine to produce a third order stream and so on (Figure 5). Only when two streams of the same order are combined does the stream order increase. Numerous lower order streams may enter a main stream without changing the stream order. As a result, utilizing this method for classification may lead to problems of disparity in hydrological and ecological conditions among same order streams even within the same region. Resource managers using stream order as a classification system should ensure that topographic maps used to identify watershed boundaries all utilize the same scale. The inclusion or exclusion of perennial headwater streams should be decided before ordering drainage networks of interest.

Stream order (Strahler 1952) is used to classify streams in the EPA Environmental Monitoring and Assessment Program (EMAP). Sample sites were selected using a randomized sampling design with a systematic spatial component. The survey in the mid-Atlantic region was restricted to wadeable streams defined as 1^{st}, 2^{nd}, or 3^{rd} order as delineated using USGS 1:100,000 scale USGS hydrologic maps that were incorporated into EPA's River Reach File (Version 3). Sample probabilities were set so that approximately equal numbers of 1^{st}, 2^{nd}, and 3^{rd} order stream sites would appear in the sample population. Data were collected at 368 different sites representing 182,000 km of wadeable streams in the mid-Atlantic region (Herlihy et al. 1998).

PHYSICAL FACTORS USED TO CLASSIFY STREAMS AND ANALYZE TROPHIC STATE

The following sections focus on physical characteristics of streams that can be used to sub-classify stream systems. Physical characteristics that can be used for stream classification include system hydrology and morphology, flow conditions, and underlying geology.

Hydrology and Morphology

Hydrologic and channel morphological characteristics are often important determinants of algal biomass. Unidirectional flow of water sets up longitudinal patterns in physical and chemical factors that may also affect macrophyte growth when light and substrate conditions are adequate. Channel morphology or shape of a river or stream channel at any given location is a result of the flow, the quantity and character of the sediment moving through the channel, and the composition of the streambed and banks of the channel including riparian vegetation characteristics (Leopold et al. 1964). Frequent disturbance from floods (monthly or more frequently) and associated movement of bed materials can scour algae from the surface rapidly and often enough to prevent attainment of high biomass (Peterson 1996). In areas with less stable substrata, such as sandy bottomed streams, only slight increases in flow may lead to bed movement and scouring. Scouring by movement of rocks has been directly linked to reduction in algal biomass and subsequent recovery from floods (Power and Stewart 1987). Larger, more stable rocks can have higher periphyton biomass (Dodds 1991; Cattaneo et al. 1997). Thus, in cases where

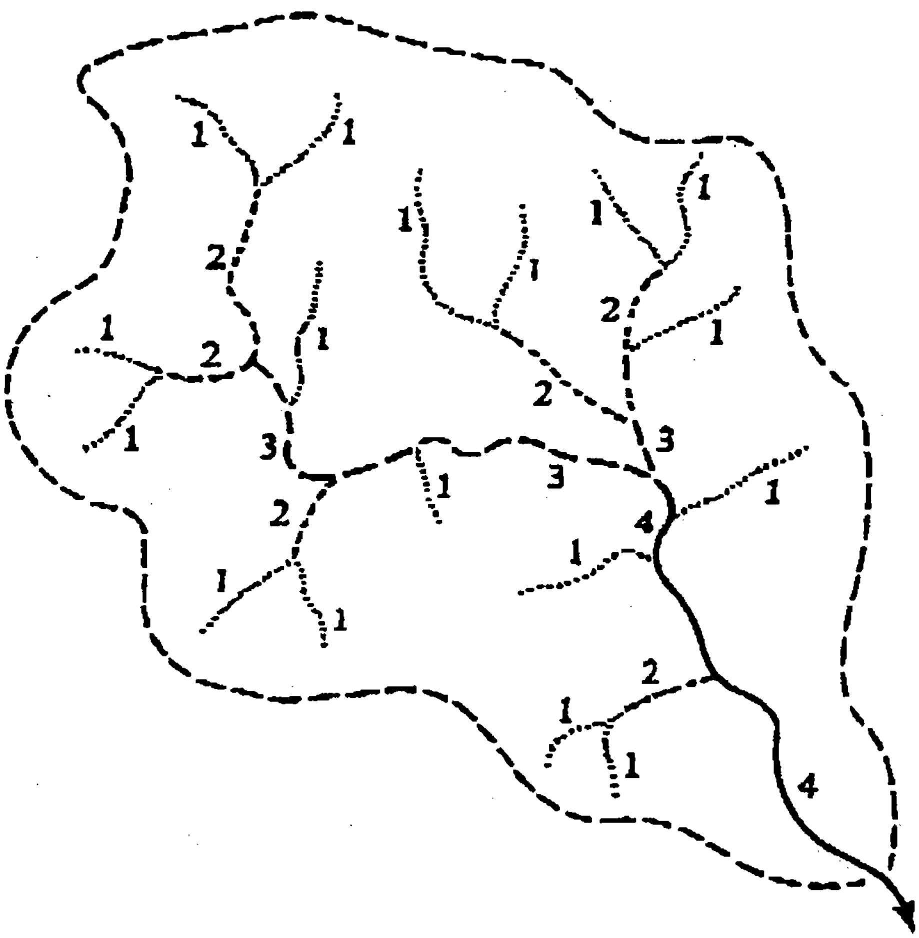


Figure 5. Stream ordering of a watershed basin network using the Strahler method. (Adapted from Strahler [1964]).

there is frequent movement of substrata, high nutrients may not necessarily translate into excessive algal biomass (Biggs et al. 1998a,b).

Consideration of both geology and hydrologic disturbance can provide important insights into factors influencing algal biomass. Research done in New Zealand identified geology, land use patterns, and stream conductivity (as a surrogate for total nutrients) as important determinants of algal biomass because these factors affected nutrient inputs and flood disturbance (Biggs 1995). The effects of disturbance by floods can be complex and complicated by biological factors; very stable stream beds may be associated with an active grazing community and have less biomass than more unstable systems. This notwithstanding, flow regime, channel morphology and bed composition (such as sand versus large boulders) appear to be major controlling factors and should be considered when managing eutrophication in a particular watershed.

Flow Conditions

Low and stable flow conditions should be considered in addition to frequency and timing of floods when physically classifying stream systems. Flood frequency and scouring may be greater in steep-gradient (steep slope) and/or channelized streams and in watersheds subject to intense precipitation events or rapid snow melt. Periods of drying can also reduce algal biomass to low levels (Dodds et al. 1996). A stream may flood frequently during certain seasons, but also remain stable for several months at a time. The effects of eutrophication may be evident during stable low flows. Also, stable flow periods are generally associated with low flow conditions, resulting in the highest nutrient concentration from point source loading. Hence, low-flow periods often present ideal conditions for achieving maximum algal biomass. For these reasons, nutrient control plans may require strategies that vary seasonally (e.g., criteria for a specific system may differ with season or index period).

Underlying Geology

Streams draining watersheds with phosphorus-rich rocks (such as from sedimentary or volcanic origin) may be naturally enriched and the control of algal biomass by nutrient reduction in such systems may be difficult. Bedrock composition has been related to algal biomass in some systems (e.g., Biggs 1995). In addition, nutrient content, and hence algal biomass, often naturally increases as elevation decreases, especially in mountainous areas (Welch et al. 1998). Some naturally phosphorus-rich areas include watersheds draining some volcanic soils, and other areas have high weathering of nitrate from bedrock (Halloway et al. 1998). Review of geologic maps and consultation with a local Natural Resources Conservation Service (NRCS) agent or soil scientist may reveal such problems.

2.3 CLASSIFICATION SCHEMES BASED ON NUTRIENT GRADIENTS

Nutrient loading is the factor most likely to be controlled by humans, but the ability to control algal biomass within the stream itself may be influenced by additional factors. Factors that may control algal biomass in streams include bedrock type and elevation (because they determine the natural or background nutrient supply), physical disturbance (flooding and drying), light, sediment load, and grazing. Many of these factors will be accounted for in the physical classification of stream systems. However, characterization of nutrient gradients in stream systems will be influenced by land use practices as well as point source discharges (Carpenter et al. 1998). The nutrient ecoregions defined by Omernik (2000) separate the country into large ecoregions with common land use characteristics. These ecoregions should be further subdivided for use at the State, Tribal, or local scale.

Changes in the natural processes that control algal production and biomass in a stream or river as one moves downstream through a watershed are obviously an important consideration. The River Continuum Concept (RCC) (Vannote et al. 1980) provides one general model for predictions of stream size effects on algal-nutrient relations. The RCC predicts, among other things, that benthic algal biomass will increase with stream size to a maximum for intermediate stream orders (i.e., third and fourth order stream reaches) as stream width increases and canopy cover consequently decreases. The RCC also suggests that (1) sestonic (suspended) chlorophyll will become more important in larger, slow-moving rivers and (2) turbidity in deep, high order streams causes light attenuation, which tends to prohibit high benthic algal biomass. The RCC may not hold for unforested watersheds (e.g., Dodds et al. 1996) or those with excessive human impacts such as impoundments or severe sediment input from logging. For example, Rosenfield and Roff (1991) observed that stream primary productivity in Ontario streams was largely independent of stream size. However, the RCC is valuable for identifying variables that change with stream size and affect algal-nutrient relations.

Classification by Nutrient Ecoregions

The draft nutrient aggregations map of level III ecoregions for the conterminous United States (Figure 4; Omernik 2000) defines broad areas that have general similarities in the quantity and types of ecosystems as well as natural and anthropogenic characteristics of nutrients. As such, ecoregions are intended to provide a spatial framework for the National Nutrient Criteria Program. In general, the variability in nutrient concentrations in streams, lakes, and soils should be less in those ecoregions having higher hierarchical levels, i.e., nutrient concentrations found in level III ecoregions (84 ecoregions delineated for the mainland U.S.) (Omernik 1987), than those of waterbodies located in draft aggregations of Level III ecoregions.

Classification by Trophic State

The primary response variable of interest for stream trophic state characterization is algal biomass. Algal biomass is usually concentrated in the benthos of fast-flowing, gravel/cobble bed streams (i.e., periphyton dominated) and measured as benthic chl *a* per unit area of stream substrate. In slow-moving, sediment-depositing rivers (i.e., plankton dominated), algal biomass is suspended in the water column and measured as sestonic chl *a* per unit water volume. Trophic classifications for lakes and reservoirs may be appropriately applied to seston in slow-moving rivers as these classifications are based primarily on chl *a* per unit volume (e.g., OECD 1982). However, lake classification schemes have limited value for fast-flowing streams dominated by benthic periphyton because the limited areal planktonic chlorophyll data available for lakes reveal little differentiation between oligotrophic and eutrophic systems (Dodds et al. 1998).

Nitrogen and phosphorus are important variables for classification of trophic state because they are the nutrients most likely to limit aquatic primary producers and are expressed per unit volume in both fast-flowing streams and slow-flowing rivers. Concentrations of total nutrients and suspended algal biomass are well-correlated in lakes and reservoirs (Dillon and Rigler 1974; Jones and Bachmann 1976; Carlson 1977). Developing predictive relationships between nutrient and algal levels in fast-flowing streams may be difficult considering that most available nutrients are in the water column and most chl *a* is in the benthos. Therefore, trophic state classification for periphyton-dominated stream systems is more appropriately based on benthic or areal algal biomass (e.g., mg/m^2 chl *a*) than on concentrations of N and P.

As stated above, classification of trophic state in stream systems is most appropriately based on algal biomass and secondarily on nutrients. When trophic state classification is based upon nutrients, total water column concentrations (TP and TN) are more appropriate than dissolved inorganic nitrogen (DIN) or soluble reactive phosphorus (SRP). Inorganic nutrient pools are depleted and recycled rapidly. Most monitoring programs will not be able to closely track soluble nutrients in a stream system and should therefore focus on total water column concentration rather than soluble nutrient species.

Additional factors also confound the interpretation of dissolved nutrient data. Algae are able to directly utilize inorganic nutrient pools (DIN and SRP) and deplete these pools if algal biomass is high enough relative to stream size and nutrient load. Thus, moderately low levels of DIN and SRP do not necessarily result in low algal biomass. This seeming contradiction is because the supply rate of inorganic nutrients may still be high even if a large biomass of algae has removed a significant portion of the DIN or SRP from the water column. Algal growth rate (including diatoms and filamentous greens) can be saturated at low dissolved inorganic nutrient concentrations (Bothwell 1985, 1989; Watson et al. 1990; Walton et al. 1995). Total phosphorus and TN may better reflect stream trophic status compared to inorganic P and N because algal drift increases with benthic algal biomass. Thus, as soluble nutrient depletion increases with benthic algal biomass, that depletion can be partially compensated for by increases in particulate fractions of TP and TN resulting from benthic algal drift and suspension in the water column.

A trophic classification scheme for streams and rivers, based on chlorophyll *a* and nutrients, was recently developed by Dodds et al. (1998). The approach used by Dodds et al. was based upon establishing statistical distributions of trophic state-related variables. The data were viewed in two ways: 1) three trophic state categories were constructed based on the lower, middle, and upper thirds of the distributions and were assigned to oligotrophic, mesotrophic and eutrophic categories respectively; and 2) the actual distributions (Table 2) were used to determine the proportion of streams in each trophic category. It should be stressed that this approach proposes

Table 2. Suggested boundaries for trophic classification of streams from cumulative frequency distributions. The boundary between oligotrophic and mesotrophic systems represents the lowest third of the distribution and the boundary between mesotrophic and eutrophic marks the top third of the distribution.

Variable (units)	**Oligotrophic-mesotrophic boundary**	**Mesotrophic-eutrophic boundary**	**Sample size (N)**
mean benthic chlorophyll (mg m^{-2})$^{+}$	20	70	286
maximum benthic chlorophyll (mg m^{-2})$^{+}$	60	200	176
sestonic chlorophyll (μg L^{-1})$^{++}$	10	30	292
TN (μg L^{-1})$^{+,+++}$	700	1500	1070
TP (μg L^{-1})$^{+,++,+++}$	25	75	1366

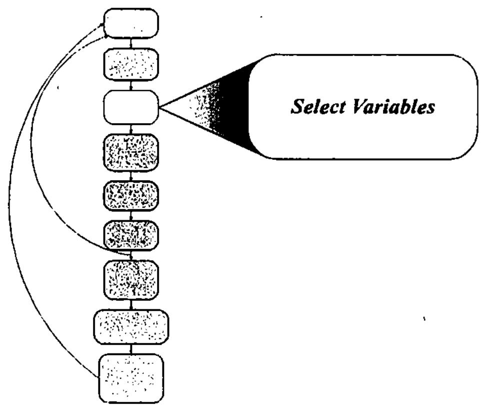
$^{+}$Data from Dodds et al. (1998); $^{++}$data from Van Nieuwenhuyse and Jones (1996); $^{+++}$data from Omernik (1977).

trophic state categories based on the current distribution of algal biomass and nutrient concentrations which may be greatly changed from pre-human settlement levels. These distributions were determined using data for benthic and sestonic chlorophyll and water column TN and TP from a wide variety of previously published studies. The data were gathered from temperate stream sites located in North America and New Zealand. The data for TN and TP used in this analysis were not taken from the same sources as the data for benthic and sestonic chlorophyll *a*. Hence, the distributions should only be used to link nutrient concentrations and algal biomass in a very general sense.

Management Applications

Classifying streams by trophic state can assist water quality managers in setting criteria and identifying those systems most at risk for impairment by nutrient enrichment. For example, an understanding of stream trophic state and ambient nutrient concentrations allows the manager to determine if the system of interest is eutrophic due to nutrient inputs that are natural or cultural. Comparisons with streams in the same local area that have similar physical characteristics will help clarify this issue prior to making management decisions. Management options may be limited if the condition of the stream is caused by high background levels of nutrient enrichment. However, if nutrient sources are largely cultural, establishing nutrient control strategies may realistically result in improvements in stream trophic state and therefore be useful in managing the stream system.

Chapter 3. Select Variables



3.1 INTRODUCTION

Candidate variables, in the context of this document, are measurable water quality variables that can be used to evaluate or predict the condition or degree of eutrophication in a water body. Data that are most useful in determining river and stream trophic status are water column nutrient concentrations and algal biomass. Benthic and/or planktonic biomass can reach nuisance levels in many stream systems. Measurement of these variables provides a means to evaluate the current degree of nutrient enrichment, and can form the basis for establishing regional and waterbody-specific nutrient criteria. Numerous variables can potentially be used as part of nutrient surveys or eutrophication assessments including measures of water column nutrient concentrations (e.g., TP, SRP, orthophosphate, TN, total Kjeldahl nitrogen [TKN], NO_3^-, ammonia [NH_3]); dissolved organic carbon (DOC); water column and algal/macrophyte tissue N:P nutrient ratios; and algal biomass surrogates (e.g., chl *a*, ash-free dry mass [AFDM], turbidity, percent of benthic algal coverage, species composition).

Criteria development at the EPA Regional and National level will begin with nutrient data gleaned from EPA's STORET (STOrage and RETrieval) database. Primary nutrient parameters to be considered include water column concentrations of TN, TP, algal biomass as chl *a*, and turbidity or transparency. These four variables are considered a starting point for criteria development and their efficacy in controlling nutrient enrichment will be re-evaluated over time. Inorganic nutrient species (PO_4 and NO_3) are usually more biologically available, and may need to be considered in instances where small scale effects from specific sources are an important issue (e.g., point source impacts from outfall pipes, and non-point source impacts during rain events immediately following inorganic fertilizer application). STORET data on the primary parameters are the foundation of the dataset used at National and Regional levels for developing nutrient criteria. Supplemental data from other Federal agencies, State/Tribal agencies, and university studies will also be included as available. Sources of available data, the parameters included in the primary datasets, and the minimum data requirements for criteria development are discussed in Chapter 5.

Interpretation of parameter values and their cause-and-effect relationships depends on whether the data are from stream segments that are slow-moving with a depository substratum and plankton-dominated, or that are fast-moving with an eroding (gravel/cobble) substratum and periphyton-dominated. Criteria for streams with intermediate characteristics, i.e., in which the bottom is not generally visible in slow-moving segments and is not likely to have algal biomass problems, may need to be developed primarily for fast moving stream segments. Hence, significance of each individual or group of variables is discussed for each extreme stream/river type; the reader, of course, realizes that flowing waters can be found along all points on the trophic continuum and parameter values can vary even within a stream reach. This chapter lists and describes (1) primary response variables that will be used by EPA to set default criteria and (2) secondary response variables (including sensitive variables, i.e., those likely to be most sensitive to enrichment as influenced by increased primary producer biomass and metabolic activity) that can be used to predict the enrichment status of stream systems.

3.2 PRIMARY VARIABLES

The primary variables considered for nutrient criteria development are water column concentrations of TN, TP, benthic and planktonic algal biomass as chl *a*, and turbidity or transparency. These variables will be used to set criteria ranges for each EPA ecoregion at the National level (see section 1.5). The primary causal variables, TN and TP, are closely related to the response variables, algal biomass as chl *a* and turbidity or transparency, although the relationships between these variables are not as tightly coupled in rivers and streams as they are in lakes. Concentrations of nutrients and algal biomass and measures of turbidity/transparency are more highly variable in rivers and streams because of fluctuating flow conditions. Therefore, knowledge of the flow conditions in the waterbody of concern will be used to help define the nutrient condition of that waterbody, and will be used in criteria development. Criteria will not be established for flow as a variable. Stream sampling should be conducted during periods of peak algal biomass or periods when problems related to algae may be greatest (e.g., low-flow or following rain events with high nonpoint source nutrient inputs). Subsequent sections of the chapter discuss other potential variables that may be useful in developing nutrient criteria. Methods for measuring and analyzing many of the variables discussed in this Chapter can be found in Appendix B.

NUTRIENTS

Nitrogen and phosphorus are the primary macro-nutrients that enrich streams and rivers and cause nuisance levels of algae. Conditions that allow periphyton/plankton biomass to accumulate (i.e., adequate light, optimum current velocity [periphyton], sufficient water detention time [plankton], as well as low loss to grazing) will not result in high biomass without sufficient nutrient supply. Nutrients, especially P, are frequently the key stimulus to increased and high algal biomass.

Phosphorus is the key nutrient controlling productivity and causing excess algal biomass in many freshwaters worldwide. However, nitrogen can become important in waters receiving agricultural runoff and/or wastewater with a low N/P ratio and in waters with naturally phosphorus-rich bedrock (Welch 1992). Nitrogen may have more importance as a limiting element of biomass in streams than in lakes (Grimm and Fisher 1986; Hill and Knight 1988; Lohman et al. 1991; Chessman et al. 1992; Biggs 1995; Smith et al. 1999). Lohman et al. (1991) reported low NO_3-N causing N limitation at sixteen sites in ten Ozark Mountain streams and cited sources for N 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 that N was more often limiting than P in Australian streams.

Analyses of data from 200 rivers suggests that TN is more closely correlated to mean benthic algal biomass than TP, and DIN is more closely correlated to biomass than SRP (Dodds et al. unpublished).

The directly available forms of N and P are mainly inorganic (NO_3^-, NH_4^+ and PO_4^{3-}), although many algae are able to use organic forms (Darley 1982). Total N and TP include these soluble fractions, as well as the particulate and dissolved organic fractions. Particulate and dissolved organic fractions are not immediately available and portions may be relatively refractory. Because soluble inorganic fractions are directly available, soluble inorganic N, P, or both may be low during active growth periods when demand is high and, therefore, may not be good predictors of biomass (Welch et al. 1988). Total N and TP are often good predictors of algal biomass in lakes and reservoirs, to a large extent because much of the particulate fraction is live algal biomass. That is not the case in fast-flowing, gravel/cobble bed streams where the total nutrient concentration includes detritus but not the living periphytic algae where biomass measurements are taken. In fast-flowing systems, water column nutrients flow past the living periphyton biomass before they can be completely assimilated. Therefore, the relationship between benthic chlorophyll and water column nutrients is weaker in fast-flowing versus standing water systems (Dodds et al. 1998).

ALGAL BIOMASS AS CHLOROPHYLL *a*

Algae are either the direct or indirect cause of most problems related to excessive nutrient enrichment; e.g., algae are directly responsible for excessive, unsightly periphyton mats or surface plankton scums, and may cause high turbidity, and algae are indirectly responsible for diurnal changes in DO and pH. Chl *a* is a photosynthetic pigment and sensitive indicator of algal biomass. It can be considered the most important biological response variable for nutrient-related problems. The following discussion of chl *a* as a primary variable includes information for both benthic and planktonic chl *a*. Benthic chl *a* can be difficult to measure reliably due to its patchy distribution and occurrence on non-uniform stream bottoms. Periphyton is often analyzed for AFDM, which includes non-algal organisms. Additional factors that can be used to determine which type of chlorophyll (benthic or planktonic) is most important in the system of interest can be found in Table 1, Section 2.2.

Unenriched, light-limited, or scour-dominated stream systems typically have benthic chl *a* values much less than 50 mg/m^2. Biggs (1995) reported the following range of chl *a* values from monthly observations over a one year period in 16 New Zealand streams: 1) unenriched streams in forested catchment (0.5-3 mg/m^2), 2) moderately enriched streams in catchments with moderate agricultural use (3-60 mg/m^2), and 3) enriched streams in catchments highly developed for agriculture and/or underlain with nutrient-rich bedrock (25-260 mg/m^2). Lohman et al. (1992) reported a range of 42 to 678 mg/m^2 chl *a* from over two years of spring to fall biweekly observations at 22 sites on 12 Missouri Ozark Mountain streams, with higher levels occurring at more enriched sites. Unenriched sites exhibited mean biomass values that did not exceed 75 mg/m^2. However, highly and moderately enriched sites exceeded a nuisance level mean biomass (150 mg/m^2) within 3 or 4 weeks, respectively, following flood-scour events. The highest maximum value observed at ten sites in late summer 1987 in the Clark Fork River, Montana, was approximately 600 mg/m^2 (Watson and Gestring 1996). Furthermore, values for benthic chl *a* as high as 1200 mg/m^2 have been observed in gravel/cobble bottom bed streams (Welch et al. 1992).

Planktonic chl *a* in deep, slow-moving rivers will have an upper limit determined by light attenuation, which increases with the suspended chl *a* concentration. Maximum chl *a* can be low (<10 μg/L) even if slow-moving systems are nutrient enriched because most flowing systems disperse phytoplankton before high algal biomass develops. However, under low flow conditions (accompanied by low mixing and shallow depth), large planktonic algal blooms often develop in slow-moving, nutrient enriched rivers. The theoretical maximum attainable before light limits photosynthesis in lakes (assuming light is attenuated by algae only) is about 250 mg/m^2. This theoretical maximum is equivalent to 25 mg/m^3 (μg/L) in a 10-m depth water column or 125 μg/L in a 2 m deep lake. Van Nieuwenhuyse and Jones (1996) compiled summer mean suspended chl *a* values for rivers, and found no values greater than 180 μg/L. Mixing and light attenuation from non-algal particulate matter, which are typical in deep, slow-moving rivers, may further limit light availability for photosynthesis.

A conceptual distribution of algal biomass in the euphotic zone over a range of water detention times was suggested by Rickert et al. (1977) (see Welch 1992). For example, the lower Duwamish River, Washington estuary typically contained around 2 μg/L chl *a* during summer, even though it was heavily enriched with secondary treated sewage effluent. However, when the water detention time increased and mixing decreased as a combined result of minimum range tidal conditions and low river flow in August, chl *a* reached a maximum of 70 μg/L (Welch 1992).

Algal biomass data in fast-flowing, gravel/cobble bed streams and deep, slow-moving, turbid rivers must be interpreted in light of the physical constraints that determine the potential for nutrient utilization (see Chapter 2). Relatively low biomass can be observed in highly enriched waters, if physical (light, temperature, current) or grazing constraints are severe. Relatively high algal biomass can occur with low enrichment if physical constraints approach the optima for algal growth. However, chl *a* concentrations near the maximum values cited above will not occur without nutrient enrichment.

TOTAL SUSPENDED SOLIDS, TRANSPARENCY, AND TURBIDITY

Total suspended solids (particulate matter suspended in the water column) attenuate light and reduce transparency, whether the source is algae, algal detritus or inorganic sediment. Streams may also have high concentrations of light-absorbing dissolved compounds (e.g., blackwater streams). The concentration of total suspended solids can be determined directly or as an effect on light transmission or scattering. Quantitative relationships have been developed for individual and/or groups of waters to predict transparency from particulate matter and/or chl *a* (Reckhow and Chapra 1983; Welch 1992). However, relationships of chl *a* and transparency (as an effect of nutrients) are not prevalent in fast-moving streams systems; most likely because of interference from time- and flow-variable inorganics and large diameter suspended solids. Total suspended solids may increase due to algae and detritus sloughed from large algal mats, but caution should be exercised in interpreting these data. During high flow, the concentration of suspended solids (and water clarity) will likely be more strongly influenced by inputs of inorganic sediment or channel erosion in streams, especially in urbanized and agricultural watersheds.

Turbidity, as NTUs (Nephelometric Turbidity Units), measures suspended matter in the water column whether of organic (i.e., chl *a*) or inorganic origin. Turbidity correlated with rain-event sampling may help identify non-point source loadings. Although turbidity is not commonly used as an index of eutrophication in either lakes or streams, it nonetheless should increase in streams with increasing algal biomass due to nutrient enrichment.

Periphyton are directly affected by suspended solids (as turbidity) due to light attenuation. Quinn et al. (1992) found that waters with turbidity measurements that range between 7-23 NTUs have reduced abundance and diversity of benthic invertebrates. They attributed the reduction in benthic invertebrates to turbidity, largely because of its adverse effect on periphyton production as an invertebrate food source (Quinn et al. 1992). In Illinois, the turbidity of agricultural streams (NTU 10-19) had more effect on periphyton accrual than did nutrient enrichment (Munn et al. 1989). Total suspended solids ranging from about 22 to 30 mg/L increased the loss rate of periphyton (mixture of filamentous blue-green and diatoms) tenfold, although increased velocity with and without solids caused more loss (Horner et al. 1990).

The vertical water column in relatively clear-water, gravel/cobble bed streams/rivers is usually insufficient to determine Secchi disk depth. However, the white Secchi disk routinely used in lakes and reservoirs to determine transparency is appropriate for slow-moving streams and rivers (Welch 1992). Transparency, as influenced by low concentrations of particulate matter in shallow, fast-flowing streams systems, can also be determined with a black disk (Davies-Colley 1988). The path length for transparency is measured horizontally in such shallow streams, as opposed to vertically in lakes, reservoirs and deep rivers/estuaries. As periphyton biomass increases, particulate matter sloughed and/or eroded from the substratum also increases, reducing transparency.

FLOW AND VELOCITY

The rate of discharge or flow in a stream system can be separated into two primary components, baseflow and storm or direct runoff. Baseflow comprises the regular groundwater inputs to a stream. This water typically reaches the stream through longer flow paths than direct runoff and sustains streamflow during rainless periods. Direct runoff is hillslope or overland flow runoff that reaches a stream channel during or shortly after a precipitation event. Both components of flow are reflected in a hydrograph (a graph of the rate of discharge plotted against time) of the stream segment. Runoff processes (including stream discharge and groundwater recharge), seasonal variation of flow, and methods to calculate average stream velocity, the annual probability hydrograph and flow duration curves are discussed at length in Dunne and Leopold (1978).

The flow of a river or stream affects the concentration and distribution of nutrients. Generally, point source concentrations are higher during low flow conditions due to reduced water volumes; in contrast, nutrients from non-point sources may be more highly concentrated during high flow conditions due to increased flow paths through the upper soil horizons and overland flow. There is also a rough correlation of total dissolved solids concentration with climate and hydrology. Streams in arid regions tend to have high concentrations of total dissolved solids (though the total annual solute transport is low because of low runoff), whereas in humid regions, concentrations tend to be lower with higher total annual solute transport (Dunne and Leopold 1978). However, the complexity of the interactions of nutrient concentration and flow make it important to examine both point sources and non-point sources of nutrients and wet weather (high flow) and dry weather (low flow) stream conditions to verify nutrient sources and concentrations in multiple flow conditions (Dunne and Leopold 1978).

Brandywine Creek, Pennsylvania, provides an example of how stream flow can affect nutrient concentrations in a stream system (Dunne and Leopold 1978). The Brandywine Creek watershed drains portions of the Piedmont plateau and Atlantic coastal plain into the Delaware River. The watershed land use is a mix of urban, agricultural and suburban uses, and includes both point and non-point pollution sources. Brandywine Creek was sampled during periods of storm runoff and dry-weather flow for P and stream discharge. Point discharges of P were diluted as stream discharge increased following storm events. As storm runoff occurred, concentrations of P increased dramatically at sampling sites not dominated by point discharges. At sites not dominated by point discharges, runoff from forested and cultivated hillslopes washed large amounts of P into the Brandywine Creek in both solid organic form and sorbed to soil particles.

Hydrologic variability is an important consideration in the development of nutrient and algal criteria for all streams; nonetheless, there is often a higher degree of variability for specific types of regional stream systems. In particular, the spatial and temporal heterogeneity found in arid regions, the stark contrast between wet and dry, can be dramatic (see Desert Streams Case Study, Appendix A). When viewing desert catchments from above, the observer is often presented with a dry landscape of high relief bisected by the string of glistening beads that is the spatially intermittent stream. The dry arroyos or quiet, disconnected pools and short reaches of wetted stream that characterize desert streams during dry periods are in complete contrast to the raging torrents that they can become at flood stage. This hydrologic variability and the unique chemical and biological characteristics of arid lands aquatic ecosystems make the use of broad generalizations to explain nutrient regimes difficult.

In arid landscapes, stream ecosystems are dynamically linked with the surrounding upland ecosystem. In addition, surface discharge regimes may vary from completely dry, to flows as much as three to five orders of magnitude greater than mean annual flow, all within a period of hours or days. In comparison to streams in more mesic regions, the coefficient of variation of annual flow is 467% greater in arid lands streams (Davies et al. 1994). The aquatic ecosystems structured by these chaotic flow regimes (Thoms and Sheldon 1996) may require different techniques for nutrient criteria development than those used in more homogeneous environments.

Drying disturbance, or more specifically the contraction and fragmentation of a stream ecosystem, is a critical component of the hydrologic regime of desert streams. Drying occurs as a spatially or temporally intermittent stream recedes after a wet period. In streams where the dry period and extent may be greater than the wet, drying is likely to be an important determinant of biological pattern and process (Stanley et al. 1997; Stanley and Boulton 1995).

In order to properly characterize the nutrient regime of a stream ecosystem, the flow of water, surface and subsurface, flood or base flow, wet or dry must be considered at ecologically significant temporal and spatial scales. It is also important that the manager address this hydrologic regime at the scale of the question to be answered. If a stream is dry for 75% of the average year, or for 75% of its length, is it correct to characterize it from surface water data alone? If 50% of the entire annual load of a limiting nutrient passes through a stream ecosystem in three discrete storm events, what is the effect of that nutrient on the stream ecosystem itself? What is the effect to downstream ecosystems? Due to the spatial and temporal variability of flow patterns, the characterization of desert stream nutrient dynamics is an intricate undertaking. However, stream complexities will only be understood through appropriate assessment and evaluation.

3.3 SECONDARY RESPONSE VARIABLES

The following sections describe additional variables that may be useful in criteria development. These variables comprise chemical, physical, and biological parameters, some of which exhibit heightened response to nutrient enrichment.

SENSITIVE RESPONSE VARIABLES

The variables discussed below that are apt to be most sensitive to nutrient enrichment, via increased algal productivity and biomass are: 1) DO and pH, 2) benthic community metabolism, and 3) autotrophic index. These variables should vary directly with algal productivity and detect relatively small changes in nutrient condition. While other variables such as total suspended solids, macroinvertebrate indices, dissolved organic matter, and secondary production may be directly affected by algal productivity and biomass, they may also be strongly dependent on other natural factors and/or sources/types of pollutants.

Dissolved Oxygen and pH

Periphyton algal biomass above nuisance levels often produces large diurnal fluctuations in DO and pH. Photosynthesis/respiration by dense periphyton mats commonly causes water quality violations (Anderson et al. 1994; Watson et al. 1990; Wong and Clark 1976). These water quality impairments occur in stream systems as a result of nutrient-produced excessive algal biomass in fast-flowing, gravel/cobble bed streams as well as sluggish stream systems. Excessive macrophyte biomass can produce similar swings in DO and pH (Wong and Clark 1979; Wong et al. 1979).

The extent of diurnal swings in DO and pH will depend on several factors, such as turbulence (which affects reaeration), light, temperature, buffering capacity, and the amount and health of algal and/or macrophyte biomass. Sluggish streams and rivers may show a greater range in DO and pH per unit biomass compared to faster streams due to less turbulence and associated atmospheric exchange of CO_2 and O_2 (Odum 1956; Welch 1992). Light limitation may also be a common feature of algae in enriched streams, and therefore, light is likely an important control on diurnal DO and pH swings (Jasper and Bothwell 1986; Boston and Hill 1991; Hill 1996). Higher temperatures tend to enhance algal growth in many streams and may increase photosynthesis and respiration in many systems resulting in greater variation in diurnal DO and pH values. Streams with low buffering capacity will show greater diurnal swings in pH. Furthermore, biomass-specific metabolic rate (especially respiration–see photosynthesis/respiration discussion) tends to be greater in fast-flowing waters because periphytic growth is stimulated by velocity. The influence of the above factors on DO concentration and pH value reduce the specificity and potentially reduce the reliability of these variables to indicate response from nutrient enrichment. Therefore, direct measures of algal biomass, such as chl *a*, are preferred response variables.

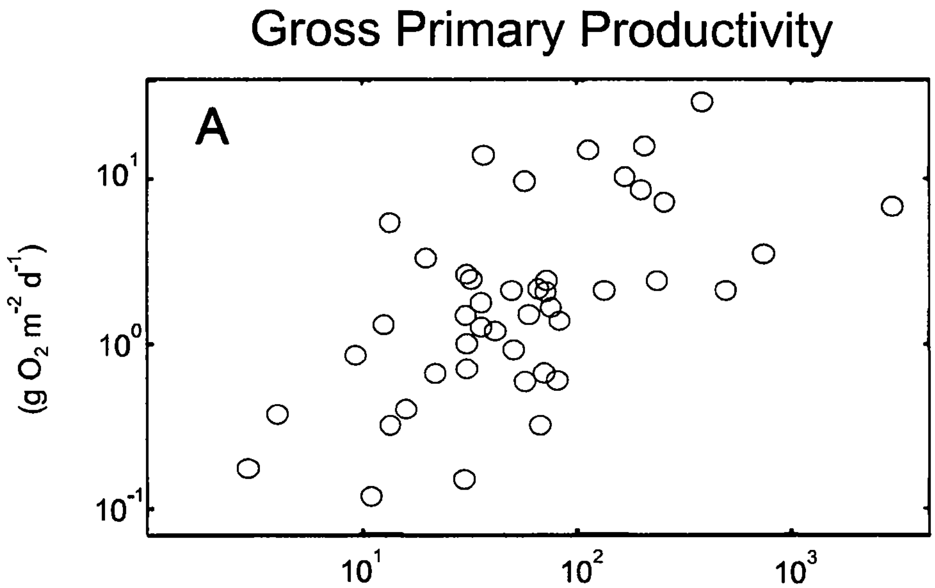
Aquatic animals are affected most by maximum pH and minimum DO, rather than by the daily means for these variables (Welch 1992). Hence, monitoring for water quality should include pre-dawn hours to observe the diurnal minimum DO and afternoon hours for maximum pH. Routine grab samples in monitoring programs usually do not include such strict protocols. It may be possible to estimate minimum DO from equilibrated average and maximum DO (Slack 1971) which occurs during mid-day to afternoon, along with maximum pH.

Metabolism

Photosynthetic rate, or primary productivity, is often considered a more sensitive variable of response to nutrients than algal biomass. Biomass is a net result of gains (productivity) minus losses (algae lost due to death, scour, etc.) (see discussion in Stevenson 1996). Productivity is essentially growth, and therefore is a more direct measure of nutrient effects. Productivity can be determined for whole stream reaches by monitoring diurnal DO concentrations (see methods section, Appendix B) or alternatively, productivity and respiration may be measured using light/dark chambers. Whole-stream metabolism measurements are integrative over all components of the stream system and eliminate artifacts of enclosure that commonly confound results in chamber experiments. Marzolf et al. (1994, 1998) detail the methods for measuring whole-stream metabolism. Productivity and respiration in light/dark chambers may vary on an hourly and daily basis with temperature, light, and nutrients; short-term measurements must be corrected for those factors (Welch et al. 1992). The necessity of normalizing measurements and the greater analytical difficulty of productivity, has made algal biomass the preferred variable to indicate nutrient effects on periphyton and phytoplankton as evidenced by the generally established trophic state criteria for lakes and reservoirs (Welch 1992), and proposed for streams/rivers (Dodds et al. 1998). The rate at which maximum biomass is attained is dependent mostly on nutrient availability, minus losses to grazing and scouring, or washout in the case of phytoplankton. While integrated daily productivity is usually directly related to biomass as chl *a* (Boston and Hill 1991), there can be considerable variability in the relationship due to the variables discussed above, as shown by the ratio of productivity to biomass as chl *a* (Figure 6). The ratio of productivity to biomass as chl *a* is an index of growth rate. If there is no variability in productivity:biomass, the relationship will be constant and will not vary on a day-to-day basis.

Gross photosynthesis/respiration ratios (P/R ratios) can be useful indicators of trophic characteristics. P/R ratios have long been recognized to indicate the relative autotrophic (P/R >1) or heterotrophic (P/R <1) character of streams and rivers. Measurement of P/R and interpretation of results is dependent on the scale at which the measurements are made, and the point in the annual cycle when the measurements are taken. For example, low-order streams that flow through forested watersheds tend to be heterotrophic with photosynthesis limited by light due to shading; mid-order streams and rivers flowing through areas with minimal riparian vegetation, or largely unshaded due to width, are usually autotrophic (unless organic waste inputs are significant); high order rivers tend to return to a heterotrophic character due to light limitation brought on by increased depth and turbidity (Vannote et al. 1980; Bott et al. 1985). Furthermore, the P/R ratio for a short-term measurement (24-72 hours) in the spring may indicate an autotrophic stream, while on an annual basis the stream is heterotrophic (Hall and Moll 1975; Wetzel 1975; Wetzel and Ward 1992).

There are problems with interpreting P/R ratios, however. Photosynthesis/respiration ratios can vary seasonally and could actually reflect a temporary heterotrophic condition during a period of low periphyton biomass, due to scouring or low light, while otherwise it would be autotrophic. Decreased velocity can also decrease stream/river P/R, because mat thickness of periphytic diatoms can increase while the depth of active photosynthesis remains relatively constant (Biggs and Hickey 1994). Thus, photosynthesis is limited by light attenuation in the mat, but respiration is stimulated by movement of organic materials to heterotrophic organisms in the mat.



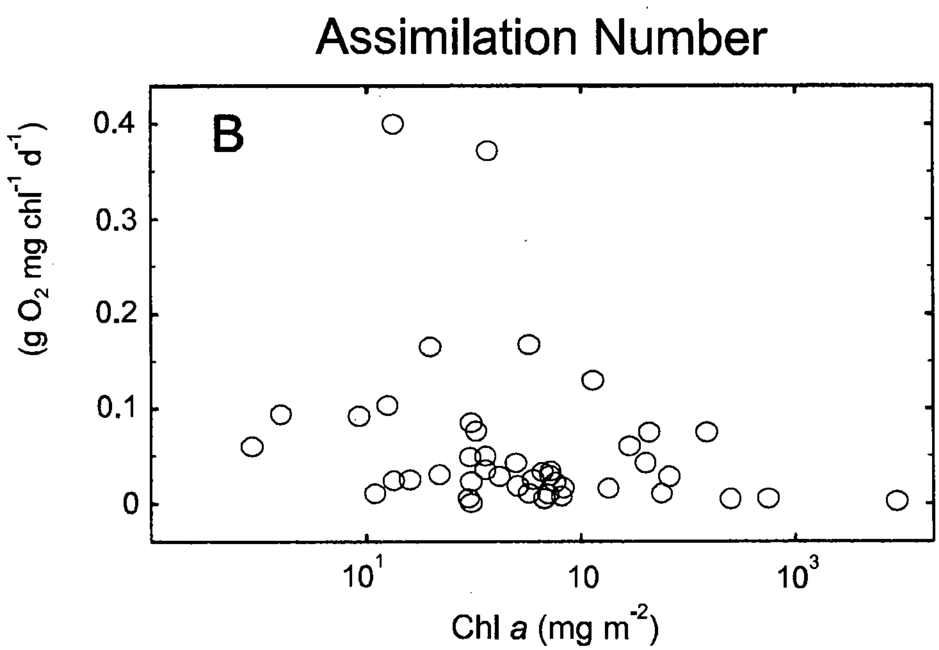


Figure 6. Integrated daily productivity related to biomass as chlorophyll *a* (data compiled by Dodds from published literature; many of the data from Bott et al. 1985).

Autotrophic Index

The ratio of AFDM to chl *a* is termed the autotrophic index for periphyton and is used to distinguish the relative response of inorganic (N and P) and organic (BOD) enrichment. Periphyton growing in surface water that is relatively free of organic matter contain approximately one to two percent chl *a* by weight. Surface water that is high in particulate organic matter may support large populations of bacteria, fungi and other non-chlorophyll bearing microorganisms, and have a larger ratio of AFDM to chl *a*. Increased ratios indicate that heterotrophs utlizing organic substances comprise a larger percentage of AFDM than autotrophic periphyton that rely largely on inorganic nutrients to increase biomass (Weber 1973). Ratios of AFDM/chl *a* can vary over three orders of magnitude, with values >400 indicating organically polluted conditions (Collins and Weber 1978). Ratios of AFDM/chl *a* around 250 are more typical for streams enriched with inorganic nutrients that are likely to have existing or potential eutrophication problems (Watson and Gestring 1996; Biggs 1996). The autotrophic index should be used with caution, because non-living organic detrital material may artificially inflate the ratio.

Interpretation of Sensitive Response Variables

High algal productivity can cause supersaturated DO and high pH during the day, P/R ratios >1, and unusually low autotrophic indices. Unfortunately, broad predictive relationships do not exist between nutrient concentration and algal/macrophyte biomass, DO, or pH. However, relationships could be developed for individual streams and rivers. Nevertheless, without inclusion of other factors that affect DO and pH (such as exchange with the atmosphere for specific stream systems), a biomass limit to prevent low DO (e.g., <5 mg/L) cannot be determined from any existing relationship, such as the chl *a* - TP relationships discussed earlier (Lohman et al. 1992; Dodds et al. 1997). As concentrations of nutrients and algae increase, diel fluctuations in DO and pH also increase (see Dissolved Oxygen and pH discussion above). However, established relationships observed in lakes and reservoirs, such as TP loading and hypolimnetic DO deficit (Welch 1992), do not exist for streams and rivers.

OTHER SECONDARY RESPONSE VARIABLES

Additional chemical, physical, and biological attributes may be useful when evaluating nutrient and algal relationships. Descriptions for several potential useful variables are provided below.

Chemical Waterbody Characteristics

Conductivity

Specific conductance (typically measured as conductivity) has also been used as an indicator of nutrient enrichment (Biggs and Price 1987; Biggs 1996). Conductance reflects the concentrations of macro-ions, so nutrients dissolved from bedrock are assumed to increase proportionately with increases in total ions. Conductance at low flow was found to increase proportionately with urbanization in 23 western Washington streams and was hypothesized to be a loose surrogate for soluble nutrient supply during summer when residual soluble nutrient concentration was low due to algal demand (May et al. 1997). However, conductance may be a poor indicator of nutrient availability in calcareous regions or those with high concentrations of dissolved salts that are not typically limiting nutrients.

Dissolved Organic Carbon

DOC is an important energy source that drives the heterotrophic community and can alter a river's response to algal growth problems. DOC can originate as allochthonous inputs naturally from the watershed through decomposition of terrestrial primary production, or from cultural waste production. The heterotrophic community will dominate the periphyton in gravel/cobble bed streams and rivers that have high inputs of labile DOC.

Inflow and in-stream DOC should be related to the autotrophic index, as discussed previously. Streams and rivers enriched with DOC will have high autotrophic indices, and may be more prone to low oxygen events that can be exacerbated by excessive periphyton biomass. High rates of autochthonous DOC production` is usually a result of inorganic nutrient enrichment. Such eutrophication-caused DOC production can be an important source of decomposition by-products (e.g., tri-halomethane precursors and other sources of taste and odor problems) which is a concern for drinking water supplies.

Physical Waterbody Characteristics

Temperature

Algal metabolic rate, at a given biomass and growth phase (relative cell health), is controlled by temperature (DeNicola 1996), water movement, nutrients and light. In general, the response to enrichment will be faster at higher than lower temperature; e.g., twice as fast at 20°C as at 10°C (McIntire and Phinney 1965; Welch 1992). However, the maximum biomass will depend on nutrient availability; temperature will determine only the rate at which the maximum is reached (Welch 1992).

Temperature, as it interacts with light and nutrients, will determine which taxa dominate the algal biomass. The various algal taxa have individual thermal optima. In general blue-greens have higher optima than greens which have higher optima than diatoms (Rodhe 1948; Cairns 1956; Hutchinson 1967). For example, the nuisance filamentous green, *Cladophora*, apparently has an optimum around 18°C and its growth stops at 25°C (Storr and Sweeney 1971). As a result of differing thermal optima, seasonal succession of taxa is often observed, with diatom dominance during spring low temperature and greens and blue-greens dominating in summer. However, nutrients often override temperature effects, with diatoms dominating the periphyton throughout the spring-summer period at low nutrient concentrations and greens (and/or blue-greens) dominating for the whole period at high nutrient concentrations (Welch 1992).

Biological Attributes

Algal Biomass as Ash-Free Dry Mass

Algal biomass or standing crop is often expressed as AFDM. However, the weight of particulate detritus in fresh water frequently exceeds that of the algae. No reasonable method currently exists to separate algae from detrital material in the water. Therefore, chl *a* is usually the primary biomass indicator because it is specific to algae, while AFDM can include other living or non-living organic matter (Darley 1982; Wetzel 1975).

Algal Biomass - % Cover of Bottom by Nuisance Algae

Extent of periphyton coverage of a stream bed can be an important indicator of algal biomass problems. As enrichment increases, the fraction of periphyton biomass composed of filamentous greens increases, as does the percent of stream bed covered with algae (Welch et al. 1988; Lohman et al. 1992; Biggs 1996). However, there may be an uncoupling between percent cover and total biomass depending on the thickness of the algal mat, e.g. , a system might have 100% algal cover, but if the algal growth was very thin (e.g., "sheets" of *Oscillatoria* filaments), the total biomass could be far less than a system with 50% cover of *Cladophora*. Nevertheless, estimates of percent cover are often a useful indicator of the intensity of algal proliferation in gravel/cobble-bed streams, and as an index of aesthetic appeal. The occurrence of floating blue-green algae scums in slow-moving rivers, lakes, and reservoirs is likewise an aesthetic nuisance, but there has been no attempt to quantify scum intensity/surface-cover similarly to periphyton in fast-flowing streams, largely due to the variable, diurnal nature of floating blue-green scums.

Pigment Ratios

Two pigment ratios are commonly used in periphyton assessments. One is the chl *a*:AFDM ratio, which is a modified version of the autotrophic index (Weber 1973; Stevenson 1996; Stevenson and Bahls 1999) and indicates the relative importance of autotrophy versus heterotrophy in streams. Values of the autotrophic index increase when algae (chl *a*) become a greater proportion of benthic biomass. The second is the chl *a*:phaeophytin ratio, which is an indicator of periphyton health. Phaeophytin is a degradation product of chlorophyll. Relatively low values of phaeophytin, thus relatively high values of the chl *a*:phaeophytin index, indicate periphyton is actively growing.

Chemical Composition of Algae (N:P Stoichiometry)

Phosphorus and N concentrations in periphyton increase with nutrient concentrations and trophic status of streams (Humphrey and Stevenson 1992; Biggs 1995). Periphyton can be analyzed for P and N content, as well as chl *a* or AFDM. Then P and N concentrations in periphyton can be expressed as a fraction of algal biomass as indicated by chl *a* or AFDM (μg P/μg chl *a* or μg P/mg AFDM). This metric can be another valuable complement to assessments of P and N availability, especially when P and N concentrations are variable in the stream.

Nutrient ratios in periphyton may provide a line of evidence to indicate whether N or P is limiting algal growth. The range of ambient or cellular N:P ratios has been used as to define the transition between N and P limitation for benthic algae (Schanz and Juon 1983). If ambient N:P ratios are greater than 20:1, then P can be assumed to be in limiting supply. If the ambient N:P ratio is less than 10:1, then N can be assumed to be in limiting supply. The distinction of the limiting nutrient when ambient N:P ratios are between 10 and 20 to 1 is not precise. Nutrient enrichment studies have supported these transition ratios in broad terms (e.g., Grimm and Fisher 1986a; Peterson et al. 1993). However, the accuracy of ambient nutrient ratio analysis decreases when greater amounts of detritus occur in periphyton samples. In streams, N:P ratios of periphyton can be different than N:P ratios in the water column (Humphrey and Stevenson 1992). Periphyton N:P ratios may better indicate relative nutrient availability to the periphyton than ratios based on water column nutrient concentrations. In addition, ambient ratios may not reflect the cellular ratio relevant to physiological growth processes when nutrients are abundant. Cellular nutrient ratios are a more direct measurement of nutrient limitation (Borchardt 1996). Even so, nutrient ratios only suggest limitation–bioassays are required to establish cause and effect relationships.

Phosphatase Activity

Alkaline phosphatase is an enzyme excreted by algae in response to P limitation. Alkaline phosphatase hydrolyzes phosphate ester bonds, releasing PO_4 from organic P compounds (Steinman and Mulholland 1996). Concentration of alkaline phosphatase in the water column can be used to evaluate P limitation. Alkaline phosphatase activity (APA), monitored over time in a waterbody, can be used to assess the influence of P loads on the growth limitation of algae (Smith and Kalff 1981). Artificial stream channel experiments by Klotz (1992) support the hypothesis that stream N:P ratio is the important factor in determining periphyton APA. In this study, APA varied seasonally, and shading of the stream channel resulted in lower APA. Results from studies of cultured algae appear to indicate that phosphatase levels above 0.003 mmol (micromoles) mg chl a^{-1} h^{-1} indicate moderate P deficiency, and phosphatase levels above 0.005 mmol mg chl a^{-1} h^{-1} indicate severe P deficiency (Steinman and Mulholland 1996).

Algal Species Composition

Assessment of algal species composition can indicate that nutrient related problems exist or that conditions are right for such problems to develop (Kelly and Whitton 1995; Pan et al. 1996). Since algae are often the problem associated with nutrient contamination, assessments of algal species composition can show whether nuisance algae are present or whether biotic integrity of this target community has changed. Assessment of algal species composition is more time consuming than simpler measurements of water chemistry or chl *a* measurement, however algal species composition may provide more reliable indicators of trophic status in streams and rivers than one-time sampling and assessment of water chemistry and benthic algal biomass (Stevenson, unpublished data). Assessment of algal species composition is an element of periphyton programs in all States that monitor periphyton. One of the reasons for relying on species composition is periphyton biomass is so variable spatially and temporally, and challenging to measure accurately. In addition, species composition is highly informative, especially when linked to the ecology of a species in relation to the environment, i.e., the autecological information about the species (Stevenson and Bahls 1999).

Many attributes of algal species composition can be used as metrics or indicators of nutrient conditions, trophic status, and biotic integrity (Stevenson and Bahls 1999). Indicators of nutrient status based on algal taxa fall in three categories: diversity, deviations in species composition from reference conditions, and weighted-average autecological indices. Diversity is comprised of two components: 1) the variety of species (species richness), and 2) the relative abundance of species (eveness). Shannon diversity (a measure of diversity which combines the components of diversity [Pielou 1975]) usually decreases with increasing trophic status because evenness decreases. Weighted-average autecological indices based on pollution tolerance, or more specifically, nutrient requirements can be used to infer nutrient status or trophic conditions in a habitat (Steinberg and Scheifele 1988; Schiefele and Schreiner 1991; Van Dam et al. 1994; Kelly and Whitton 1995; Pan et al. 1996). Dissimilarity in species composition between test and reference sites can be used to determine whether water quality is similar in test and reference sites. A more complete review of metrics and how algae can be used in environmental assessment of rivers and streams can be found in McCormick and Cairns (1994), Stevenson and Pan (1999) or Stevenson and Bahls (1999).

Grazers and Secondary Production

Dense populations of algae-consuming grazers may lead to negligible algal biomass in spite of high levels of nutrients (Steinman 1996). The existence of a "trophic cascade" (control of algal biomass by community composition of grazers and their predators) has been demonstrated for some streams (e.g., Power 1990). Grazer biomass was related more strongly with P concentration in 12 Quebec streams than was periphytic algal biomass, which was considered controlled by grazing in spite of TP concentrations ranging from 5 to 60 µg/L (Bourassa and Cattaneo 1998). The potential for manipulations of foodwebs to control eutrophication certainly warrants more investigation, but there is not currently enough information on trophic cascades in streams to allow for use of foodweb dynamics as a management option. Managers still should realize the potential control of algal biomass by grazers, but also be aware that populations of grazers may fluctuate seasonally or unpredictably, and fail to control biomass at times. Consideration of grazer populations may at least explain why some stream systems with high nutrients have low algal biomass.

Phytoplankton losses in slow-moving rivers due to filter-feeding grazers can also be significant. Bivalve communities can filter large volumes of water on a daily basis (as much as 10-100% of the water column, depending on population density) (Strayer et al. 1999). The amount of particulate matter grazed from this filtration may exceed losses to pelagic filter-feeders or downstream advection. Significant losses of pelagic phytoplankton have been observed in large rivers. Strayer et al. (1999) describe a zebra mussel invasion of the Hudson River ecosystem that drastically reduced phytoplankton (and zooplankton) biomass by 80-90%, as well as a 50% reduction in phytoplankton biomass in a reach of the Potomac River following colonization by the bivalve *Corbicula fluminea*. Ecosystem response to severe biomass reduction by filter-feeding grazers is often characterized by an increase in dissolved nutrients like SRP, reduced turbidity, and proliferation of macrophytes. Inherent qualities of the waterbody (e.g., mixing, sediment stability, and light attenuation) are a factor in determining whether phytoplankton biomass is permanently reduced, regardless of increases in nutrient concentration, or temporarily reduced and then replenished with a shift in dominant phytoplankton species (Caraco et al. 1997).

Production and biomass of consumers is expected to be greater in streams/rivers enriched with N and P. At some point, however, productivity and biomass will cease to increase at all or the rate of increase per unit nutrient will be greatly reduced. One feature of highly enriched lakes and reservoirs is the switch to grazer-resistant filamentous/colonial blue-green algae, which reduces the efficiency of nutrient utilization and energy conversion to higher trophic levels (Welch 1992). Although not well documented, the same phenomenon may be expected in enriched streams and rivers resulting in increased biomass and percent coverage of filamentous green algae. On the other hand, low-level enrichment of oligotrophic streams and rivers may result in pronounced increases in benthic invertebrates and fishes in addition to increased algal biomass. For example, continuous enrichment of the P-limited Keogh River and Grilse Creek on Vancouver Island, British Columbia, led to substantial increases in secondary producers, but did not produce nuisance biomass levels of periphyton (Perrin et al. 1987; Slaney and Ward 1993). Enrichment of the Keogh River and Grilse Creek with 5-10 and 5 μg/L SRP, respectively, produced maximum periphyton biomass (chl *a*) levels of 100-150 and 50-100 mg/m^2. Consequently, benthic invertebrate biomass increased from 2-7 fold and fish size 1.4-2 fold. Phosphorus fertilization (10 μg/L) of a tundra river led to increased fish and algae production, but negligible increases in invertebrate production (Peterson et al. 1993). In some cases, enrichment of oligotrophic waters may result in increased grazer biomass with little or no change in periphyton biomass (Biggs and Lowe 1994).

Even if nuisance levels of periphyton are produced, secondary production will probably be higher than in unenriched waters in spite of reduced efficiency of conversion. Enrichment of Berry Creek, Oregon, with sucrose (1-4 mg/L) produced large, nuisance mats of filamentous bacteria, but benthic invertebrate biomass increased 4.5 fold and fish (cutthroat trout) increased 6.3 fold with enrichment (Warren et al. 1964). Although adverse effects of periphytic mats and water quality were apparently not evaluated, fish growth obviously prospered from the large biomass of chironomids that consumed the filamentous bacteria.

Secondary production can clearly respond to enrichment and the response may be more efficient and beneficial in oligotrophic than eutrophic streams systems. A transition region in enrichment from beneficial to detrimental effects has not been defined to the extent that it has for lakes and reservoirs (Welch 1992), but probably exists for different physical types of streams and rivers. Two recent studies have provided independent estimates of target streamwater nutrient concentrations that should be maintained in order to assure acceptable water quality needed for fish growth (Smith et al. 1999). McGarrigle (1993) concluded that maintaining a mean annual SRP concentration <47 mg m^{-3} was necessary to prevent the nuisance growth of attached algae and to preserve water quality suitable for salmonid fishes in Irish rivers. Similarly, Miltner and Rankin (1998) observed deleterious effects of eutrophication on fish communities in low order Ohio streams when total inorganic nitrogen (TIN) and SRP concentrations exceeded 610 mg m^{-3} and 60 mg m^{-3}, respectively.

Invertebrate and fish biomass are considered very useful variables, albeit more demanding to measure than other indices discussed above. Measuring such variables could prove useful because: 1) both may respond to enrichment, 2) fish are of direct economic and recreational importance, and 3) case studies are needed to develop guidelines for regions of enrichment that represent a transition between beneficial and detrimental effects of enrichment.

Macrophytes

Macrophyte is a general term of no taxonomic significance that is applied to many species of aquatic vegetation. Aquatic plants (macrophytes) can be classified into four groups: emergent, floating-leaved, submersed, and freely floating and are large enough to be observed by the naked eye. Aquatic macrophytes represent a taxonomically diverse group of aquatic plants and include flowering vascular plants, mosses, ferns, and macroalgae (USEPA 1973; Wetzel 1975). Macrophytes are found in most waterbodies and play an important role in the aquatic community providing food for other aquatic organisms, processing nutrients or toxic elements in the water column, and aiding in the stabilization of river/stream sediments (Davis 1985).

The four categories of macrophytes are defined by their connection or anchor to the waterbody substrate: free-floating, emergent (rooted but breaking the water surface), floating leaf anchored, and immersed floating mat anchored (USEPA 1973). The type of growth form plays an important role in the effects of eutrophication on macroscopic plant communties in rivers and streams. For example, the large surface area provided by the thin narrow leaves of *Potamogeton pectinatus* (sago pondweed) allow this species to persist in flowing water with high turbidity (Hynes 1969; Goldman and Horne 1983). Emergent macrophytes grow on the banks of rivers and streams in depths of water less than a meter and are typically rooted in the sediment, have their basal portions submersed in water and have their upper structural biomass growing in the air. Most emergent macrophytes are perennials (living for more than one year). Common emergent macrophytes include plants such as reeds (*Phragmites* spp.), bulrushes (*Scirpus* spp.), cattails (*Typha* spp.), and wild rice (*Zizania* spp.). Floating-leaved macrophytes are rooted to the river bottom with leaves that float on the surface of the water such as waterlilies (*Nymphaea* spp.) and spatterdock (*Nuphar* spp). Submersed macrophytes are a diverse group that grow completely under the water and include mosses (*Fontinalis* spp.), muskgrasses (*Chara* spp.), stoneworts (*Nitella* spp.) and numerous native vascular plants such as various pondweeds (*Potamogeton* spp.), tape-grass (*Vallisneria* spp.), and exotic species including hydrilla and Eurasian watermilfoil. Free-floating macrophytes typically float on or just under the water surface with their roots suspended in the water column. These unattached macrophytes range in size from small duckweeds (*Lemna* spp.) and water fern (*Salvinia* spp.) to larger surface floating plants such as water hyacinth (*Eichhornia crassipes*). Free-floating species are entirely dependent on the water for their nutrient supply. The distribution and abundance of free-floating macrophytes in streams is affected by current velocity and wind. Thus, they are most frequently found in backwaters and embayments (Goldman and Horne 1983).

The most important environmental factors affecting the abundance and distribution of aquatic macrophytes in rivers are light availability (Spence 1975; Chambers and Kalff 1985; Canfield et al. 1985), nutrients and water chemistry (Hutchinson 1975; Beal 1977; Kadono 1982; Hoyer et al. 1996), substratum characteristics (sediment texture, nutrient content) (Pearsall 1920; Barko et al. 1986; Nichols 1992), and current velocity. Aquatic plants require light for growth, thus light availability is often considered the single most crucial environmental factor regulating the maximum depth of plant growth (Pearsall 1920; Spence 1975; Chambers and Kalff 1985). Light availability is directly linked to water clarity; as water depth increases or water clarity decreases, both the amount and spectral quality of light for photosynthesis decreases (Canfield et al. 1985; Chambers and Kalff 1985). Light availability in rivers is controlled by riparian canopy cover and water clarity, which can be due to both organic and inorganic suspended particles (Vannote et al. 1980). Thus, shaded, turbid, and deep rivers will have fewer aquatic macrophytes.

There are few reports of nutrient-related growth limitation for aquatic plants; nutrients supplied from sediments combined with those in solution are usually adequate to meet nutritional demands of rooted aquatic plants, even in oligotrophic systems (Barko et al. 1986). There are exceptions, however. Barko et al. (1991) showed that interstitial ammonia limited the growth of hydrilla in the Potomac estuary. Nutrient enrichment of nutrient poor waters will increase plant production if no other factors constrain growth. However, the effects of enrichment for macrophytes are confounded by competition with planktonic and epiphytic algae that may reduce underwater light penetration of submerged macrophytes and negate any direct effects of nutrient enrichment (Chambers et al. 1999). Bottom sediments act as the primary nutrient source for macrophytes, and for the most part, water column nutrients must be incorporated into the sediments before they become available for uptake by macrophytes (Chambers et al. 1999).

The physical aspects of sediment texture and as an anchoring point for aquatic plants are also important to the success of macrophytes in stream systems. Some bottom types (e.g., rocks or cobble) are so hard that plant roots cannot penetrate them and fast flowing gravel/cobble bottom stream systems rarely contain enough sediment to support rooted macrophytes. Other sediments are too soft or unstable to anchor rooted macrophytes well enough to endure changes in velocity. In addition, extremely coarse-textured sediment (sand) can be nutritionally poor and therefore require accumulation of organic matter from plant growth or erosion to provide suitable substrate for macrophyte growth (Goldman and Horne 1983).

Macrophytes affect the water quality and human uses of water, other resident organisms, and nutrient cycling. In turn, the above factors influence the growth and abundance of the macrophyte community. To obtain the desired biological integrity of an aquatic community, macrophytes should be present and healthy. However, excess natural or cultural enrichment may yield an overabundance or nuisance growth of macrophytes (USEPA 1973). Macrophytes can inhibit phytoplankton growth by competing for nutrients and sunlight, and by limiting light penetration and therefore photosynthetic processes below the surface (Wetzel 1975). Macrophytes affect the DO and carbon dioxide (CO_2) concentrations, alkalinity, pH, and nutrient supply of a water body through primary production and respiration. Overgrowth of macrophytes in rivers and streams may decrease sediment transport by lowering the flow velocity.

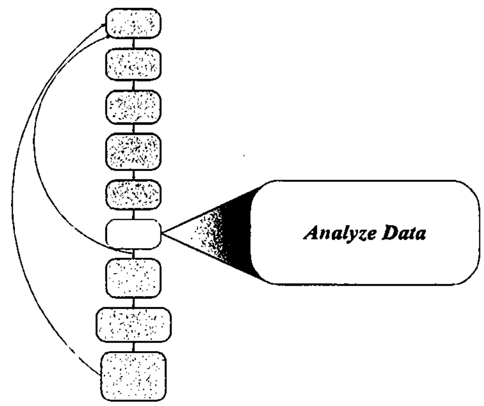
Current velocity, sediment type, and light availability to a large extent determine the plant types that occur in rivers (Hynes 1969; Goldman and Horne 1983; Chambers et al. 1999).

Macrophytes can be an important index of biological health in a waterbody. Their abundance or shortage may be an indicator of excess or deficient nutrient supply. By monitoring macrophytes over a long period of time (along with other parameters), relationships may be developed between macrophyte productivity and nutrients, nutrient cycling, eutrophication, sediment, and other biota (USEPA 1973). Depending on natural nutrient conditions or waterbody trophic state, N or P may be the limiting nutrient in algal/macrophyte biomass accumulation (USEPA 1973; Smart 1990). Phosphorus in particular, but also N and other nutrients, may be taken up by submerged macrophytes from sediment, uncoupling macrophyte growth from water column nutrient concentrations (Welch 1992). Hence, water column measurements of total N and P (or soluble N and P) are usually not indicative of macrophyte growth potential. However, macrophyte growth has been shown to be responsive to sediment pore-water ammonia content. As noted in the Bow River case study (see Appendix A), macrophytes declined in the Bow River following N removal from point source wastewater plants. This decline was hypothesized to have resulted from reductions in sediment N.

Macroinvertebrate Multi-Metric Indices

Indices employing macroinvertebrates as indicators of nutrient pollution have great potential because they are the most reliable and frequently used organisms to indicate the quality of water. Macroinvertebrates are 1) highly sensitive to changes in water quality and disturbance, 2) relatively immobile, long-lived and easy to sample, and 3) an important food supply for fish and therefore economically important. While the productivity and biomass of macroinvertebrates, as secondary producers, readily respond to enrichment as noted above, the individual taxa also respond. Some macroinvertebrates are particularly sensitive to nutrient enrichment, but local metrics of macroinvertebrates must be developed to reliably use macroinvertebrates as indicators of nutrient enrichment. The peer-reviewed stream ecology literature describing nutrient and macroinvertebrate interactions is extensive. Wallace and Webster (1996) provide a review of the literature. Specific methods for sampling macroinvertebrates and developing metrics for different stressors are described in Barbour et al. (1999). Further discussion of macroinvertebrate multi-metric index development can be found in Resh and Rosenberg (1984) and Resh et al. (1996). This type of metric development could be used to derive macroinvertebrate indices of nutrient enrichment in wadeable streams and rivers. In addition, Norton et al. (2000) describes procedures to use biological assessments, including multi-metric indices, for identifying nutrient stress on both macroinvertebrates and fish.

Chapter 6. Analyze Data



6.1 INTRODUCTION

Data analysis is critical to nutrient criteria development. Proper analysis and interpretation of data determines the scientific defensibility and effectiveness of the criteria. Therefore, it is important to re-evaluate short and long-term goals for stream systems within the ecoregion of concern. These goals should be addressed when analyzing and interpreting nutrient and algal data. Specific objectives to be accomplished through use of nutrient criteria should be identified and revisited regularly to ensure that goals are being met. The purpose of this chapter is to explore methods for analyzing data that can be used to develop nutrient criteria. Included are techniques that link relationships between nutrient loading and algal biomass, statistical analyses to evaluate compiled data, and a discussion of computer simulation models.

The difficulty associated with understanding predictive relationships between nutrient loading and algal biomass is perhaps the biggest challenge to establishing meaningful nutrient criteria. Several relatively simple methods of making this link for a variety of stream systems are discussed in this chapter. This chapter also presents more in-depth methods to use when simpler techniques prove inadequate.

Macrophytes depend primarily on sediments for nutrient uptake, and are relatively unaffected by nutrient water column concentrations. However, attempts to relate macrophyte growth or biomass with sediment nutrient content have been largely unsuccessful (Chambers et al. 1999). Links between macrophytes and nutrient enrichment are more indirect than with algae, and are therefore not considered here. A review of macrophytes and the current state of the science can be found in Chambers et al. (1999).

Statistical analyses are used to interpret monitoring data for criteria development. Statistical methods are data-driven, and range from very simple descriptive statistics to more complex statistical analyses. The type of statistical analysis required for criteria development will be determined by the source, quality, and quantity of data being analyzed. Concerns to be aware of during statistical analyses are discussed in this chapter. Specific statistical tests that may be useful in criteria development are described in Appendix C.

Models are abstractions designed to represent something real. In this sense, models can be anything from a representation of the human form in plaster, or a statistical equation expressing assumed relationships between parameters of interest. This chapter discusses modeling as mathematical abstractions for the purposes of analyzing data to derive nutrient criteria. Mathematical models can be categorized as process-based or empirical, and are used for different purposes. This guidance focuses on empirical models that serve to illuminate the relationship between the behavior of the system and measurements of one or many attributes of the system. Empirical models identify patterns but do not explain them. In contrast, process-based models are explanatory, and are built of equations that contain directly definable, observable parameters. The rules used for process-based models invoke levels of organization other than the components being modeled (Wiegert 1993).

Empirical models can be simple, statistical models or more complex simulation models. A linear regression of chlorophyll and P (phosphorus) data from a population of streams is a simple empirical model, in that it elucidates the relationship between chlorophyll and P in a single equation represented by a line. A more complex empirical model is the computer simulation model CE-QUAL-RIV1, which is comprised of a set of equations that predicts a constituent concentration over time. Prediction by both linear regression and computer simulation are based on empirical observations of a stream or population of streams. The linear regression described above is an example of a static model; static models do not represent changes over time. Dynamic models, such as CE-QUAL-RIV1, represent changes in system constituents over time (Wiegert 1993).

6.2 LINKING NUTRIENT AVAILABILITY TO ALGAL RESPONSE

When evaluating the relationships among nutrients and algal response within stream systems, it is important to first understand which nutrient is limiting. Once the limiting nutrient is defined, critical nutrient concentrations can be specified and nutrient and algal biomass relationships can be examined to identify potential criteria to avoid nuisance algal levels. This section will discuss defining the limiting nutrient, establishing predictive nutrient-algal relationships, analysis methods for establishing nutrient-algal relationships, analysis of algal species composition for system response to nutrients, characterizing biotic integrity and response to nutrients, developing a multimetric index of trophic status, assessing nutrient-algal relationships using experimental procedures, and a few other issues to keep in mind while analyzing data.

DEFINING THE LIMITING NUTRIENT

Defining the limiting nutrient is the first step in identifying nutrient-algal relationships. Nuisance levels of algal biomass are common in areas with strong nutrient enrichment, ample light, and stable flow regime. Experimental data have demonstrated that given optimum light, non-scouring flow, and modest to low grazing, enrichment of an oligotrophic stream will usually increase algal biomass and even secondary production (Perrin et al. 1987; Slaney and Ward 1993; Smith et al. 1999). Identification of the limiting nutrient is the first step in controlling nutrient enrichment and algal growth (Smith 1998; Smith et al. 1999). Criteria will be set for both TN and TP, but it is often more cost-effective to reduce the loading of one nutrient (N or P) to achieve reduction of nuisance algal growths.

Nitrogen frequently limits algal growth in streams and some have argued that this might be more common in streams than it is in lakes (Grimm and Fisher 1986; Hill and Knight 1988; Lohman et al. 1991; Chessman et al. 1992; Biggs 1995; Smith et al. 1999). However, there is evidence that P still often limits stream algae (Dodds et al. 1998; Welch et al. 1998; Smith et al. 1999). If nonpoint sources of nutrients predominate (assuming relatively high background levels of N), then N control may be a more important issue than control of P. However, if N limits growth in a stream due to point source discharges such as wastewater with low N:P, then the logical, cost-effective measure to control nuisance biomass is to reduce P input, because N:P should then increase and cause P limitation (see Section 3.3 Secondary Response Variables). If N and P are co-limiting, increasing the concentration of one nutrient will result in the other nutrient becoming limiting (e.g., an increase in N concentrations will result in P becoming limiting). The most prudent approach to controlling nutrient enrichment, regardless of the limiting nutrient, is to set criteria for maxima of N and P, and try to limit inputs of both.

Nitrogen usually becomes more limiting as enrichment increases because (1) wastewater N:P ratios are low, (2) N is increasingly lost through denitrification; (3) P is more easily sorbed to sediment particles than N and, thus, tends to be deposited in the sediment (in a waterbody with enough residence time to allow sedimentation) more effectively than does N (Welch 1992); and (4) P is released from high P-yielding bedrock. However, N lost through anaerobic denitrification may be limited by streamflow aeration, although denitrification rates may still be relatively high if the subsurface (hyporheic and parafluvial) components of the stream ecosystem are considered (see Holmes et al. 1996). Furthermore, P dissolved from bedrock or soil, whether complexed or not, is apt to remain in the water until it reaches a waterbody with enough residence time to allow sedimentation, therefore loss of nutrients via sedimentation is not usually important in most streams.

Although N may be a relatively more important controlling factor for growth in streams than lakes, there is evidence that P can limit stream algae. For instance, ratios of soluble N:P averaged 90:1 (by weight) in seven western Washington streams draining both forested and urbanized watersheds (Welch et al. 1998). Soluble N:TP ratios averaged 13:1 in three other western Washington streams (Welch et al. in press). Even more convincing evidence for a greater prevalence for P limitation in streams comes from the large data set discussed later in this chapter (Dodds et al. 1998). These data show that: 1) TN:TP ratios are nearly all >10:1, and 2) TN:TP ratios declined as enrichment increased from 24:1 (10% of data; TN = 316 and TP = 13 µg/L) to 20:1 (50% of data; TN = 1000 and TP = 50 µg/L) to 12.6:1 (90% of data; TN = 2512 and TP = 100 µg/L). The second point indicates that TN:TP in streams behaves similarly to that in lakes as enrichment increases, i.e., as enrichment increases, the ratio of water column TN:TP declines. An important cause for this may be the high concentration of P in wastewater (N:P = 3:1; Welch 1992) and in the runoff from applied animal manure (N:P $\leq$ 3:1; Daniel et al. 1997). As an in-stream example, DIN to SRP ratios in seven New Zealand streams receiving wastewater averaged 57:1 upstream and 13:1 downstream from effluent inputs (Welch et al. 1992).

Many experimental procedures are used to determine which nutrient (N, P, or carbon) limits algal growth. Algal growth potential (AGP) bioassays are very useful for determining the limiting nutrient and revealing the presence of chemical inhibitors (USEPA 1971). Yet, results from such assays usually agree with what would have been predicted from N:P ratios in the water or, especially N:P in biomass. While limiting nutrient-potential biomass relationships from AGP bottle tests are useful in projecting maximum potential biomass in standing or slow-moving water bodies, they are not as useful in fast-flowing, and/or gravel or cobble bed environments. Also, the AGP bioassay utilizes a single species which may not be representative of the natural species assemblage response.

Limitation may be detected by other means, such as alkaline-phosphatase activity, to determine if N is actually limiting in spite of a high N:P ratio. Alkaline phosphatase is an enzyme excreted by some algal species in response to P limitation. This enzyme hydrolyzes phosphate ester bonds, releasing orthophosphate (PO_4) from organic phophorus compounds (Steinman and Mulholland 1996). Therefore, the concentration of alkaline phosphatase in the water column can be used to assess the degree of P limitation. Alkaline phosphatase activity, monitored over time in a waterbody, can be used to assess the influence of P loads on the growth limitation of algae (Smith and Kalff 1981).

Periphyton biomass accrual experiments using nutrient-diffusing substrata (Pringle and Triska 1996) are useful for determining the limiting nutrient for a mixed species assemblage in running water and include the important factors of velocity-enhanced, nutrient uptake as well as constraints imposed by mat thickness that are nonexistent with bottle tests (Grimm and Fisher 1986b; Lohman et al. 1991; Pringle and Triska 1996). However, the existing ambient nutrient concentrations produced from the nutrient diffusing substrata and available for algal uptake are largely unknown with such tests.

Another experimental technique to determine ambient nutrient-maximum periphyton biomass potential in running water is with constructed channels, either with controlled light and temperature in the laboratory (Horner et al. 1983) or with natural light and temperature outdoors, along side natural streams (Stockner and Shortreed 1976; Bothwell 1985, 1989; Pringle and Triska 1996). Pringle and Triska (1996) describe methodologies for both nutrient diffusing substrata and in-stream channels.

Correlations between algal biomass and TN and TP (Dodds et al. 1997) indicate that N explains more of the variance than does P, although P may frequently be the limiting nutrient in stream systems. However, these results may be biased by the stream data used in correlation analyses. That is, the systems where nuisance algal biomass has been measured may be primarily N limited, although this may not be a reflection of a tendency for N limitation in all stream systems generally. In addition, sediment-bound particulate P may remain suspended in streams, confounding the relationship between P and algal biomass. Finally, the nutrient that limits growth in the short term may not always be the most cost-effective nutrient to control. Therefore, careful evaluation of nutrient limitation should be undertaken prior to criteria development and restoration efforts.

ESTABLISHING PREDICTIVE NUTRIENT-ALGAL RELATIONSHIPS

Once the limiting nutrient has been identified, the data need to be analyzed to characterize nutrient-algal relationships and patterns that clarify those relationships. Data analyses can provide mathematical approximations of the relationships that will allow prediction of algal biomass as a function of nutrient concentration. Predictive relationships between nutrients and periphyton (or phytoplankton) biomass are required to identify the critical or threshold concentrations that produce a nuisance algal biomass.

Relationships between TP and/or TN and periphytic biomass in streams have relatively low r^2 values on the order of 0.4-0.6 (Lohman et al. 1992; Dodds et al. 1997). Therefore, the following considerations need to be taken into account when establishing predictive nutrient-algal relationships. Critical and highly variable factors other than nutrients – shading, type of attachment surfaces, scour, water level fluctuations that result in dessication, grazing intensity – have major effects on algal biomass levels and may provide an explanation for the weakness of the predictive relationships in streams. In addition, TP in the stream water column contains more sediment-and detrital-bound P than observed in lakes, and sediment-bound P is not necessarily available for algal uptake. The high detritus level in streams is indicated by TP versus chl *a* per volume (i.e., seston) relationships in streams where chl *a*/TP ratios ranged from 0.08 to 0.22 (Van Nieuwenhuyse and Jones 1996). These ratios suggest that the high detritus levels in streams are indicative of high proportions of water-column P bound to sediment or heterotrophic components of detrital material. Finally, inorganic nutrient species (PO_4 and NO_3) are frequently more available for uptake, and may need to be considered in instances where small scale effects from specific point and nonpoint sources are an important issue.

There are few existing relationships that predict algal biomass as a function of TN and TP. Dodds et al. (1997) compiled and analyzed the largest and broadest dataset (approximately 200 sites) in the literature that predicts relationships for benthic algal biomass. The best general approach for predicting mean suspended chlorophyll was developed using data from 292 temperate streams (Van Nieuwenhuyse and Jones 1996). The equations derived from these analyses are presented for use with periphyton-dominated and plankton-dominated systems, respectively.

The equations suggested by Dodds et al. (1997) are recommended to predict benthic algal biomass if more local, ecoregion-specific relationships are unavailable:

$$\log(\textbf{\textit{mean chl a}}) = 1.091 + \log(\text{TP}) * 0.2786 \quad (r^2 = 0.089)$$

$$\log(\textbf{\textit{mean chl a}}) = 0.01173 + \log(\text{TN}) * 0.5949 \quad (r^2 = 0.35)$$

$$\log(\textbf{\textit{maximum chl a}}) = 1.4995 + \log(\text{TP}) * 0.28651 \quad (r^2 = 0.071)$$

$$\log(\textbf{\textit{maximum chl a}}) = 0.47022 + \log(\text{TN}) * 0.60252 \quad (r^2 = 0.28)$$

where seasonal mean and maximum benthic chlorophyll are in mg/m^2 and TN and TP are in µg/L. The above equations are fairly simple and, although they have low r^2 values, are best suited for use with data having high TN and TP concentrations. Note that the graphical illustration of the relationships from which these equations were derived, shows a broad distribution of the data (Figure 7). This distribution suggests that periphytic algae tend to respond in a similar fashion to nutrients, regardless of location.

A second set of equations, also derived by Dodds et al. (1997), combines TN and TP measures resulting in higher r^2 values, but may be inaccurate in some high nutrient situations.

$$\log(\textbf{\textit{mean chl}}) = -3.233 + 2.826(\log \text{TN}) - 0.431(\log \text{TN})^2 + 0.255(\log \text{TP}) \quad (r^2 = 0.43)$$

$$\log(\textbf{\textit{max chl}}) = -2.702 + 2.786(\log \text{TN}) - 0.433(\log \text{TN})^2 + 0.306(\log \text{TP}) \quad (r^2 = 0.35).$$

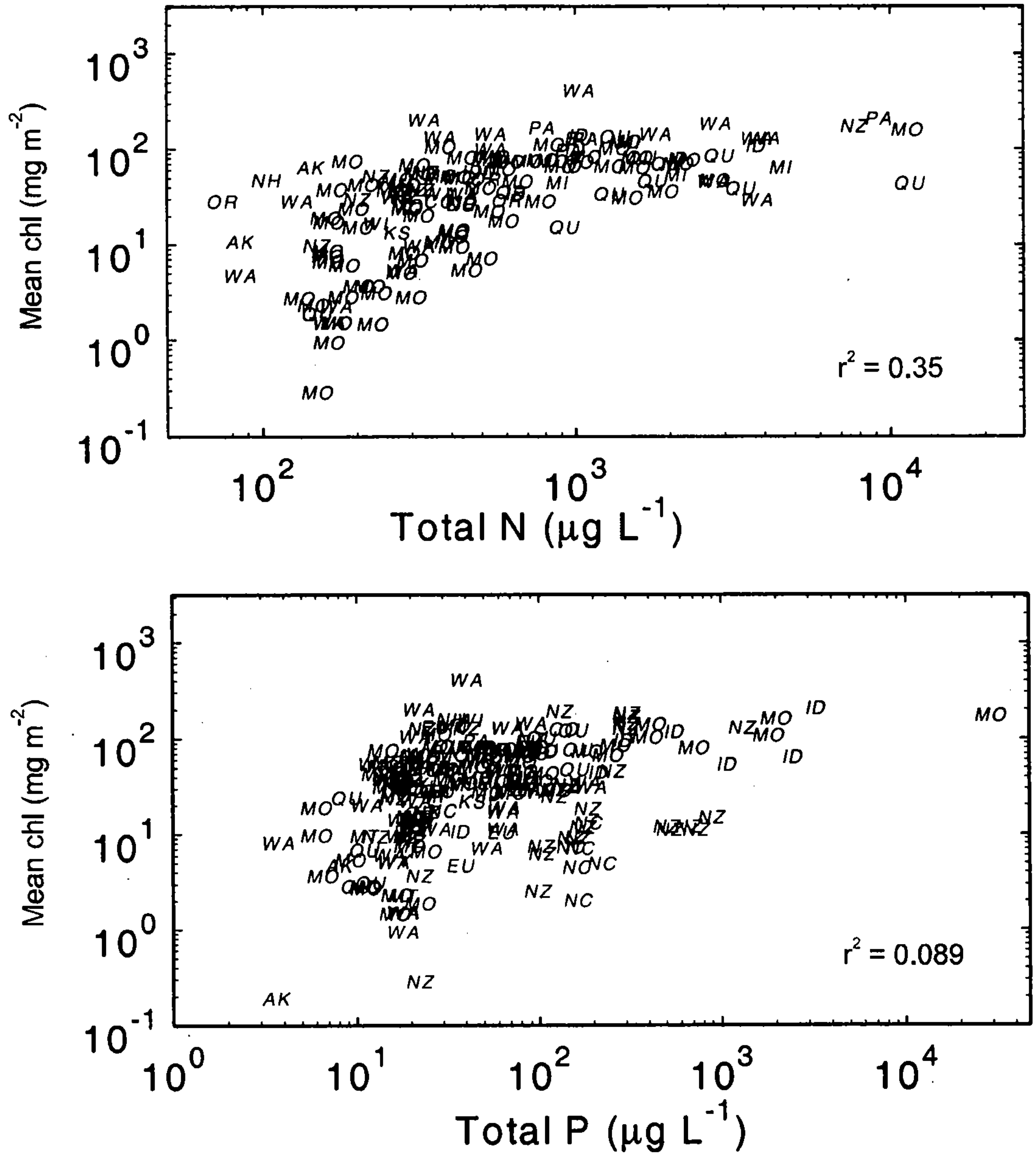


Figure 7. Relationships of log-transformed mean chlorophyll *a* as a function of TN and TP.

Data points are represented by abbreviations identifying the State or country of origin: AK- Alaska, ID-Idaho, MI- Michigan, MO-Montana, NH-New Hampshire, NC-North Carolina, OR-Oregon, PA-Pennsylvania, WA-Washington, QU-Quebec, EU-Europe, NZ-New Zealand.

It should be kept in mind that there is considerable variance in these relationships, and if extensive data for a single system are available, tighter predictive relationships may be constructed. More local, ecoregion-specific data sets should produce tighter relationships.

The equation suggested by Van Nieuwenhuyse and Jones (1996) is recommended to predict mean suspended chlorophyll if more local, ecoregion-specific relationships are unavailable:

$$\log \text{Chl} = -1.65 + 1.99(\log \text{TP}) - 0.28(\log \text{TP})^2 \quad (r^2 = 0.67)$$

Where chl = summer mean chlorophyll and TP are expressed in mg/m^3.

Yields of algal biomass from given nutrient concentrations derived from regression models differ from the yield observed in controlled channel experiments. This discrepancy creates a problem when attempting to predict nutrient-periphyton chl *a* relationships in streams. For example, to produce a mean chl *a* of 100 mg/m^2 would require approximately 100-200 µg/L TP according to regression models of Lohman et al. (1992) and Dodds et al. (1997). Brezonik et al. 1999 used the equation from Van Nieuwenhuyse and Jones (1996) that includes the catchment size (basin area) to predict likely improvements in concentrations of growing season mean chl *a* that would occur with corresponding reductions in growing season mean TP.

$$\log \text{Chl} = -1.92 + 1.96(\log \text{TP}) - 0.30(\log \text{TP})^2 + 0.12(\log A_c) \quad (r^2 = 0.73, n = 292)$$

Where A_c = stream catchment area.

They predicted that a reduction of streamwater TP from 125 to 100 µg/L would result in a chl *a* reduction of 18%, and a TP reduction from 50 to 25 µg/L would result in a chlorophyll *a* reduction of 52%. However, in-channel experiments have produced 600 to 1000 mg/m^2 chl *a* in a mixed algal assemblage using in-channel SRP and TP concentrations of 10-15 and 20-50 µg/L , respectively, a yield of ~10-50 chl *a*/TP (Horner et al. 1983, 1990; Walton et al. 1995; unpublished data). This seeming discrepancy may result from the nutrient demand by heterotrophic organisms in the detritus of natural streams. Residence time was short (16 minutes or less) in the above cited channel experiments, nutrient input was controlled to low levels, and velocity was usually constant with little sloughing during the growth period (Horner et al. 1990). Such characteristics would generate little detritus and low ambient TP and, hence, higher in-channel chl *a*/TP ratios than in natural streams sampled throughout the year.

The discrepancy in algal biomass yield between regression models and channel experiments may partly justify the use of regression models generated from large field data sets in recommending nutrient criteria. Channel data are not significantly confounded by the sloughed biomass that produces detrital material in natural streams and is unavailable for uptake and algal biomass increase. Although the correlation between chl *a* and nutrients in natural streams may be weakened (from the cause-effect standpoint) due to interference with detritus, the relations may nonetheless be useful for extrapolation and management because nutrient criteria must be applied where high detritus levels do exist.

Soluble nutrient concentrations determine periphytic growth rate and biomass; uptake is clearly saturated at very low (<10 µg/L SRP) concentrations (Bothwell 1985, 1989; Walton et al. 1995) and is independent of TP concentrations. However, soluble nutrients are usually lowest when biomass is highest, due to depletion by algal uptake, similar to the situation in lakes. Therefore, estimates of inflow nutrient concentrations, in-stream concentrations during non-growth periods or at least annual mean concentrations are required to use soluble nutrients to set critical levels and relate soluble nutrients to algal biomass. These data/relationships are not currently available, but should be pursued in order to develop more direct, stronger nutrient-biomass relationships for streams.

Analysis Methods for Establishing Nutrient-Algal Relationships

The following analysis methods are suggested to develop predictive nutrient-biomass relationships in stream systems. These methods were primarily developed for gravel/cobble bed streams, but should function for other stream types with modifications. Intermittent and effluent-dominated streams will benefit from supplemental analysis methods specific to those stream types as the seasonal sampling discussed here may not be possible (see Appendix A). Samples for soluble and/or total N and P should be collected for at least one, preferably two or more years at sites with high as well as low summer biomass. Ideally, samples for periphyton biomass should be collected weekly or biweekly during summer low flow, beginning immediately after spring runoff or any subsequent high water, scouring event. Monthly data collection may be sufficient to define algal-nutrient relationships if supporting long-term trend data is available. Data can be analyzed using one or all of the following methods to establish predictive nutrient-biomass relationships in stream systems.

1. Relate the total concentration of a limiting nutrient (e.g., TN, TP) with the mean and maximum algal biomass as chl *a*; both data sets should be collected at the same time during summer (or season of maximum algal biomass). Such data were used by Dodds et al. (1997) to develop the relationship between nutrients and algal biomass discussed in the previous section. Relate the low/non-growing period mean concentration of limiting nutrient to summer maximum biomass as chl *a*.

2. It may also be possible to relate the pre-maximum growth period (usually spring, immediately following runoff) mean soluble limiting nutrient concentration to maximum algal biomass. Inorganic soluble N (ammonium and nitrate) should be used as the limiting nutrient if the N:P (soluble) is <10 (by weight) and SRP should be used if N:P >10. The threshold of 10 is chosen to simplify the assessment protocol, although N and P are known to be co-limiting over a rather wide range in N:P ratio (7-15) (Smith 1982; Welch 1992). Data should be stratified into discrete ranges of N:P ratios, if this approach does not produce sound relationships, in a manner similar to the methods used by Prairie et al. (1989).

   This analysis selects data that would most closely represent an "inflow concentration" of dissolved inorganic limiting nutrient because it utilizes the available form of the designated limiting nutrient during a period when algal nutrient uptake is minimal. The pre-growth period nutrient concentration should be analogous to the inflow limiting nutrient concentration (including groundwater) entering a continuous algal culture system, whether planktonic or periphytic, that yields a maximum steady-state biomass. Analysis of N and P loading could be used for this assessment in stream systems, though it has not been tested. However, because rivers, streams, lakes, and estuaries form a linked system in the context of a watershed, load analysis becomes crucial at watershed scales. Relationships can be sought for TP and TN using this method and in method 3 below, which may be more appropriate for criteria throughout an ecoregion, although less specific for given streams.

3. Relate annual mean soluble nutrient concentration to the 75th percentile mean algal biomass. This approach does not provide sound continuous culture rationale like inflow concentration-maximum biomass relationships, but annual mean values for nutrients were used in the cellular N and flood frequency versus chl *a* relationship discovered by Biggs (1995), as well as soluble N and P concentrations versus maximum chl *a* for different accrual times (Biggs 2000). In instances where nutrient data are inadequate to provide distinct and reliable values used in method 2 above, an annual mean approach may offer a reasonable approximation of nutrient availability.

4. Another possibility for developing strong predictive relationships is the use of cellular concentrations of limiting nutrient (same ratio criterion used in 2 above) determined during the summer growth period, related to maximum algal biomass. This approach estimates the available nutrient directly from physiologically relevant data, as opposed to using the pre-growth soluble fractions in water to infer what is available for uptake. The validity of this approach is supported by a multiple relationship among cellular N, chl *a*, and flood frequency, in which cellular N content varied over a range of four-fold (Biggs 1995). A sound relationship between cellular nutrient content and periphytic algal biomass would, however, still require a link to the respective limiting nutrient concentration in water for management purposes. That could be accomplished by developing a relationship between cellular nutrient and ambient nutrient concentrations (either soluble or total) using constant flow laboratory channel experiments.

As further evidence for the potential of this approach, Wong and Clark (1976) described a direct relationship (r^2=0.80) between cellular P and ambient TP in six rubble-bed streams in Ontario, such that;

$$TP_w = 0.05\ P_t - 0.02$$

where P_t is tissue content, and TP_w is ambient water column TP. They determined further that photosynthetic rate of *Cladophora* at optimum light availability, decreased below 1.6 mg P/g dry weight, which was equivalent to 60 mg/L TP in the water. Nevertheless, this had no predictive value for maximum biomass. Development of a relation between cellular limiting nutrient and biomass, instead of productivity, would be necessary to back- calculate to ambient nutrient content, either soluble nutrient as in methods two or three above, or total nutrient as from method one and Wong and Clark (1976).

ANALYSIS OF ALGAL SPECIES COMPOSITION TO CLASSIFY STREAM RESPONSE TO NUTRIENTS

Differences in algal species composition among streams can identify important regional environmental gradients that may affect algal-nutrient relationships. Algal species composition should be used in data analysis to validate stream classification and enable development of indicators of nutrient conditions and the likelihood of nuisance algal blooms. Different classes of streams may require different nutrient criteria, depending upon algal responses to nutrients in different stream classes. For example, algal-nutrient problems may be related to proliferation of filamentous green algae *Cladophora* or *Spirogyra*, benthic or planktonic diatoms, dinoflagellates, or blue-green algae. Each of these problems may occur at different nutrient concentrations, but will probably only occur in certain classes of streams during specific seasonally-optimum conditions (Biggs et al. 1998b).

Cluster analysis is used to identify groups of streams with similar algal assemblages. TWINSPAN (Two Way INdicator SPecies Analysis; Hill 1979) and UPGMA (Unweighted Pair Group Method using Arithmetic averages; Sneath and Sokal 1973) represent two examples of cluster analysis that are commonly used and differ in how results are generated. TWINSPAN employs a divisive approach in which all algal assemblages are initially grouped in one cluster and then that cluster is divided into two groups based on the greatest dissimilarities between the groups. Subsequently, each cluster is divided into two more clusters so that one cluster becomes two, two becomes four, four becomes eight, and eight becomes 16, etc. In contrast, UPGMA is an aggregational technique that begins with all algal assemblages separated into single assemblage clusters and builds clusters by aggregation of the most similar clusters. So N clusters becomes N-1 clusters, and N-1 clusters becomes N-2 clusters, and so on. At each step, one algal assemblage is grouped with another assemblage or group of assemblages. Results of both techniques can be used together by identifying groups of assemblages (and associated streams) that cluster the same in both analyses. These groups can be designated as core clusters. Assemblages that are not grouped in the same clusters in both analyses can be associated with core clusters based on some simple evaluation, such as percent similarity to assemblages in the core cluster.

Cluster analysis of algal assemblages can be used as one step in classifying streams based on their response to nutrients (e.g., Pan et al. in press). Habitat classification is based on assemblages in reference conditions, because human impacts may constrain species membership in assemblages and mask diversity among stream classes and impacts that nutrients have on that diversity. In addition, algal assemblages in different classes of streams may respond differently to nutrient addition (Biggs et al. 1998b). The number of stream classes that should be used depends on many factors, but the number should be limited based on practicality, utility in explaining algal responses to nutrient enrichment, and utility in explaining algal responses to remediation. In addition, statistical significance of clusters, based on discriminate analysis for example, can also form the basis for determining the number of stream classes. Algal assemblage clusters can be related to the physical classification (described in Chapter 2), to predict responses of similar stream classes to further enrichment or remediation (Biggs et al. 1998b).

The form of species composition data used in classification of stream algal assemblage, and other analyses as well, has a substantial effect on resolution of patterns that are related to the phenomena with which we are concerned. Algal species composition data based on species densities (cells/cm^2), relative abundance (% of assemblage), and presence/absence differ successively in sensitivity to diurnal and daily changes in environmental conditions. Both theoretically and in practice, species composition data based on species densities are more sensitive to small-scale spatial and temporal variability than are data based on species relative abundances and presence/absence data (Stevenson unpublished data). Most stream classification analyses should be done with relative abundances because they integrate over space and time and most results in the literature are presented in this form.

Ordination helps to visualize differences in species assemblages among classes of streams. When species composition is combined with environmental data or algal autecological characteristics, the important environmental factors affecting species composition in a region can be deduced. These environmental factors may be important for constraining algal response to nutrient concentration and may therefore be important for identifying confounding factors in the relationship between algal assemblages and nutrient conditions. Caution should be exercised in using ordination to develop attributes of algal assemblages for use in establishing nutrient criteria. Ordination scores for taxa and classifications will change as new data are added and ordinations are recalculated. Therefore, ordinations should not be recalculated after a standard classification system or assessment system has been established. Species scores based on the original ordination should be used in subsequent classifications and assessments (Barbour et al. 1999).

CHARACTERIZING NUTRIENT STATUS WITH ALGAL SPECIES COMPOSITION

Theory and empirical evidence indicate that algal species composition may be a more precise indicator of nutrient status and the potential for nuisance algal problems than one-time sampling and assessment of nutrient concentrations and algal biomass. Shifts in algal species composition may be more sensitive to changes in nutrient concentrations and may therefore help define nutrient criteria. Many monitoring programs utilize multiple lines of evidence to increase the certainty of assessments. Algal species composition, as well as growth form and mat chemistry, can provide evidence of nutrient condition and a greater certainty of assessing nutrient conditions. This topic has been the subject of many recent reviews (McCormick and Cairns 1994; Kelly et al. 1995; Whitton and Kelly 1995; Lowe and Pan 1996; Stevenson 1998; McCormick and Stevenson 1998; Wehr and Descy 1998; Kelly et al. 1998; Ibelings et al. 1998; Stevenson and Pan 1999; Stevenson and Bahls 1999; Stoermer and Smol 1999; Stevenson in press).

Species composition and autecological characteristics of algae are commonly used to evaluate environmental conditions, ranging from organic (sewage) contamination to pH and nutrient conditions (Kolkwitz and Marsson 1908; Zelinka and Marvan 1961; Renberg and Hellberg 1982; Charles and Smol 1988; Whitmore 1989; Kelly and Whitton 1995; Pan et al. 1996). With diurnal and weekly variability in environmental concentrations within streams due to metabolic and weather-related factors or periodic releases of pollution from point sources, it is assumed that the biological assemblages that develop over longer periods of time are adapted to the average conditions in those habitats and tolerant to the environmental maxima and minima. Thus, if environmental tolerances and sensitivities of organisms are known, the physical, chemical, and potentially biological conditions for a habitat can be inferred if environmental effects differed among species.

Autecological characteristics, the environmental preferences for specific taxa, are frequently documented in the literature, particularly for diatoms (see van Dam et al. [1994] or Stevenson and Bahls [1999] for a literature list). Autecological characteristics have been compiled and summarized in several publications (Lowe 1974; Beaver 1981; Van Dam et al. 1994). Accuracy of the autecological characterizations in these compilations is limited to multi-category classification systems. For example, a categorical characterization of nutrient sensitivity might vary with the integers from 1-5, where 1 would be assigned to species least sensitive to low nutrients and 5 would indicate taxa most sensitive to low nutrients (van Dam et al. 1994). Thus, high abundance in a habitat of taxa classified as 5 would indicate highly eutrophic conditions. In contrast, more accurate characterizations of algal taxa have been achieved recently by using weighted averages of species relative abundances and a quantitative assessment of the environmental conditions in which they are observed (e.g., ter Braak and van Dam 1989; Birks 1988). The result is an accurate assessment of the specific environmental conditions in which a species will have its highest relative abundance (environmental optima). The weighted average approach assumes that species have optima along environmental gradients if each gradient (nutrients, pH, salinity, organic contamination) includes a broad range of conditions that includes most of a species range. These weighted average descriptions of species autecologies have been developed for optimal total phosphorus concentrations in streams (Pan et al. 1996).

A trophic status indicator (TSI) can be calculated by summing the products of species relative (proportional) abundances (p_i, ranging from 0-1) and their autecological characterization for trophic status (Θ_i) for all *i* species:

$$TSI = \sum_{i=1,s} p_i \Theta_i$$

If all *i* species do not have autecological characteristics, normalize the index by adjusting description of the community to only those taxa that have autecological characteristics:

$$TSI = \sum_{i=1,s} p_i \Theta_i / \sum_{i=1,s} p_i$$

Weighted average indices can be calculated easily with a spreadsheet. The weighted average formula can be used with categorical or weighted average autecological characterizations; see Kelly and Whitton (1995) and Pan et al. (1996) respectively. When indices are used with the highly accurate environmental optima determined by weighted average regression, they actually infer the phosphorus concentration or nitrogen concentration in the stream (Pan et al. 1996). Comparisons of precision of inferring TP concentrations with weighted average indicators and one-time measurement of TP concentration in a stream show that diatom indices are more precise (Stevenson and Smol in press).

Kelly and Whitton (1995) make several adjustments to sample processing and index calculation that make processing easier while maintaining index performance and distinguishing between organic and inorganic nutrients. They make sample processing easier by only counting 200 diatoms and a single set of diatom taxa that are easy to identify and that are good indicators of nutrient condition (Kelly 1996). Weights of species can be added to this formula to decrease the importance of taxa that have a broad tolerance to trophic status (see formula in Kelly and Whitton 1995), but they may not improve precision of the indices (Pan et al. 1996). Finally, autecological information is also available for assessing organic (sewage) contamination in waters. This information can be used with a TSI to distinguish enrichment effects due to inorganic and organic pollution Kelly and Whitton (1995).

Most autecological characteristics of diatom taxa have been described from European populations. Further testing will be important to determine how well autecological characterizations of taxa found in Europe compare to those in North America. However, these autecological indices should be useful for general classification of relative trophic status in streams when reference conditions and relations between changes in species composition and nutrient concentrations have not been established. The relative benefits of more accurately defining autecological characteristics with weighted averages versus coarse scale categories have not been thoroughly evaluated. Investigations have shown that inferences of environmental conditions based on indices using weighted average autecologies are more precise than those using categorical autecologies (ter Braak and van Dam 1989; Agbeti 1992). Tradeoffs may exist between greater precision for indices that are calculated with weighted average autecologies when they are used in conditions similar to those where the autecologies were developed versus less error associated with categorical autecologies when indices are used across broad diverse regions. Details and references to development of algal indices of environmental conditions can be found in recent reviews (Birks 1998; Stoermer and Smol 1999, Stevenson and Pan 1999; Stevenson and Smol in press).

DEVELOPING MULTIMETRIC INDICES TO COMPLEMENT NUTRIENT CRITERIA

Multimetric indices are valuable for summarizing and communicating results of environmental assessments and may be developed as an alternative to numeric criteria. Furthermore, preservation of the biotic integrity of algal assemblages, as well as fish and macroinvertebrate assemblages, may be an objective for establishing nutrient criteria. Multimetric indices for macroinvertebrates and fish are common (e.g., Kerans and Karr 1994; Barbour et al. 1999), and multimetric indices with benthic algae have recently been developed and tested on a relatively limited basis (Kentucky Division of Water 1993; Hill et al. 2000). However, fish and macroinvertebrates do not directly respond to nutrients, and therefore may not be as sensitive to changes in nutrient concentrations as algal assemblages. It is recommended that relations between biotic integrity of algal assemblages and nutrients be defined and then related to biotic integrity of macroinvertebrate and fish assemblages in a stepwise, mechanistic fashion. This section provides an overview for developing a multimetric index that will indicate algal problems that are associated with trophic status in streams.

The first step in developing a multimetric index of trophic status is to select a set of ecological attributes that respond to human changes in nutrient concentrations or loading in streams. Attributes that respond to an increase in human disturbance are referred to as metrics. Six to ten metrics should be selected for the index based on their sensitivity to human activities that increase nutrient availability (loading and concentrations), their precision, and their transferability among regions and habitat types. Selected metrics should also respond to the breadth of biological responses to nutrient conditions (see discussion of metric properties in McCormick and Cairns 1994; Stevenson and Smol in press).

Many structural and functional attributes of algal assemblages can be used to characterize the biotic integrity of algae (McCormick and Cairns 1994; Stevenson 1996; Kelly et al. 1998; Stevenson and Pan 1999). Biomass, species composition, species diversity, chemical composition, productivity, respiration, and nutrient turnover rates (spiraling distance) are examples of these attributes. All of these attributes are important and respond with different lag times to spatial and temporal variability in environmental conditions. Most monitoring programs measure structural attributes because structural characteristics vary less than functional characteristics on diurnal and daily time scales. For example, state monitoring programs (e.g., KY, MT) rely on changes in species composition, rather than biomass and chemical composition, to assess ecological conditions in streams because species composition is hypothesized to vary less. However, the relationship between all algal attributes, if characterized for an appropriate time and space, can be related to nutrient concentrations to determine the effect of nutrients on algal assemblages in streams.

Many algal metrics can be used to characterize the valued ecological attributes that we want to protect in a habitat or the nuisance problems that may develop as a result of nutrient enrichment. These are "response" or "condition" metrics (Paulsen et al. 1991; Barbour et al. 1999) and they should be distinguished from "stressor" or "causal" indicators, such as nutrient concentrations (water chemistry or periphyton chemistry) and biological indicators of nutrient concentrations. While both "response" and "stressor" metrics could be used in a single multimetric index, we recommend that separate multimetric indices be used for "response" and "stressor" assessment. Distinguishing between "response" and "stressor" indices can be accomplished utilizing a risk assessment approach with separate hazard and exposure assessments that are linked with response-stressor relationships (USEPA 1996; Stevenson 1998; Barbour et al. 1999; Stevenson and Smol in press). A multimetric index that specifically characterizes "responses" can be used to clarify goals of management (maintainance or restoration of valued ecosystem attributes) and to measure whether goals have been attained with nutrient management strategies.

Measurements of nutrient concentrations and algal indicators of nutrients could be combined to develop a multimetric "stressor" index specifically for nutrient conditions. Metrics of nutrient concentrations such as water and mat chemistry (μg P/mg AFDM, μg N/mg AFDM) are described in Appendix C and are relatively straight forward. Biological indicators of nutrient concentrations are described in the above section, Characterizing Nutrient Status with Algal Species Composition. The following paragraphs discuss algal metrics that characterize valued ecological attributes and nuisances.

Algal metrics can be distinguished with respect to types of designated use that is being impaired. Algal biomass can be measured as percent cover by filamentous algae, turbidity, mg chl a/m^2, g $AFDM/m^2$. Determining when biomass becomes a nuisance will require relating biomass to designated uses, such as support of aquatic life (biotic integrity), or potability. Effects of nutrients on algal biomass and effects of algae on the biotic integrity of macroinvertebrates and fish should be characterized to aid in developing nutrient criteria that will protect designated uses related to aquatic life (e.g., Miltner and Rankin 1998). Potability can be impaired by algae that cause taste and odor problems and whose growth may be stimulated by nutrients. Thus, relationships should be developed between nutrients and taste and odor producing algae or nutrients and the frequency of taste and odor complaints to develop management plans and criteria to support potability as a designated use. Relative abundance or biomass of taste and odor algae (Palmer 1962) may be good indicators of the potential for potability problems. Percent toxic algae could provide indicators of potential for toxic algal blooms in streams at low flow in which wildlife and livestock could be endangered, although little is known about the effects of toxic algae in streams.

Biotic integrity of algal assemblages may be indicated by many quantitative attributes of algal assemblages (Stevenson 1996; Stevenson and Pan 1999). Attributes of species composition can be characterized at different levels of resolution, e.g., actual biomass ($biovolume/cm^2$), relative biovolume relative abundances, cell density, or presence/absence at each taxonomic level. Relative biovolume is usually used to characterize changes in functional groups (as defined by physiognomy and taxonomic division) of algae in assemblages because cell sizes vary so much among functional groups (e.g., filamentous cyanobacteria, colonial cyanobacteria, diatoms, and large cells of filamentous green algae). Relative abundances are usually used to characterize changes in species composition of specific groups of taxa, such as diatoms. Many environmental programs only evaluate diatom assemblages for species level indicators (e.g., Kentucky Division of Water 1993; Pan et al. 1996; Kelly et al. 1998).

Even though many taxonomic attributes of algal assemblages would be expected to change in response to increasing nutrient concentrations, analyses should be focused to some extent on variables that have intrinsic value. Thus, changes in relative biovolume from non-nuisance algae (e.g., diatoms) to filamentous green algae with nutrient addition may be an indicator of loss in biotic integrity, because habitat structure and food availability for invertebrates (e.g., Holomuzki et al. 2000). Loss of species may be an issue: such as some macroalgae that are relatively sensitive to nutrient enrichment and overgrowth by diatoms (e.g., filamentous red algae or some nitrogen-fixing, blue-green algae such as *Nostoc*).

Another approach for characterizing biotic integrity of algal assemblages as a function of trophic status in streams is to calculate the deviation in species composition or algal growth forms at assessed sites from composition in the reference condition. Multivariate similarity or dissimilarity indices need to be calculated for multivariate attributes such as taxonomic composition (Stevenson 1984; Raschke 1993) as defined by relative abundance of different algal growth forms or species, or species presence/absence. One standard form of these indices is percent community similarity (PS_c, Whittaker 1952):

$$PS_c = \sum_{i=1,s} \min(a_i, b_i)$$

Here a_i is the percentage of the i^{th} species in sample a, and b_i is the percentage of same i^{th} species in sample b. A second common community similarity measurement is based on a distance measurement (which is actually a dissimilarity measurement, rather than similarity measurement, because the index increases with greater dissimilarity, Stevenson 1984; Pielou 1984). Euclidean distance (ED) is a standard distance dissimilarity index, where:

$$ED = \sqrt{\left(\sum_{i=1}(a_i - b_i)^2\right)}$$

log-transformation of species relative abundances in these calculations can increase precision of metrics by reducing variability in the most abundant taxa. Theoretically and empirically, we expect to find that multivariate attributes based on taxonomic composition more precisely and sensitively respond to nutrient conditions than do univariate attributes of algal assemblages (see discussions in Stevenson and Smol accepted). High precision and sensitivity argues for including assessments of algal species composition and its response to nutrient conditions in the process of developing nutrient criteria. The response of algal species composition to increases in nutrient concentrations can be used as another line of evidence to develop a rationale for specific nutrient criteria in specific classes of streams.

To develop the multimetric index, metrics must be selected and their values normalized to a standard range such that they all increase with trophic status. Criteria for selecting metrics can be found in McCormick and Cairns (1994) or many other references. Basically, sensitive and precise metrics should be selected for the multimetric index and selected metrics should represent a broad range of impacts and perhaps, designated uses. Values can be normalized to a standard range using many techniques. For example, if 10 metrics are used and the maximum value of the multimetric index is defined as 100, all ten metrics should be normalized to the range of 10 so that the sum of all metrics would range between 0 and 100. The multimetric index is calculated as the sum of all metrics measured in a stream. A high value of this multimetric index of trophic status would indicate high impacts of nutrients in a stream and should be a robust (certain and transferable) and moderately sensitive indicator of nutrient impacts in a stream. A 1-3-5 scaling technique is commonly used with aquatic invertebrates (Barbour et al. 1999; Karr and Chu 1999) and could be used with a multimetric index of trophic status as well.

Arguments have been made for limiting membership of metrics in a mulitmetric index to only biological metrics and only biological metrics from one assemblage of organisms (Karr and Chu 1999). We generally concur with that recommendation. More detailed descriptions of this multimetric index development can be found in Karr and Chu (1999), Barbour et al. (1999), and Hill et al. (2000)

ASSESSING NUTRIENT-ALGAL RELATIONSHIPS USING EXPERIMENTAL PROCEDURES

Management of nutrients to ensure high stream quality is greatly strengthened by examining relationships between the limiting nutrient and maximum algal biomass (i.e., potential) that will occur if/when other factors are optimum. Relationships between ambient nutrient content and existing biomass may not adequately predict maximum biomass potential for any single stream because other factors, such as light, high-flow scouring, and grazing often limit biomass accrual in natural streams. Experimental procedures are valuable for determining the maximum biomass potential of a system. However, physical constraints imposed in experimental setups are often unrealistic. Thus, the value of extrapolating results from laboratory experiments to natural conditions is often uncertain. There are many more experimental results reported to determine which nutrient (N, P, or carbon) limits algal growth, than to determine nutrient-biomass relationships. Experimental procedures to determine the limiting nutrient/s for algal growth are discussed earlier in this section (see Defining the Limiting Nutrient).

As indicated previously, biomass levels up to 1000 mg/m^2 chl *a* were accrued on stones of in-stream channels receiving as little as 10 mg/L SRP (Walton et al. 1995). Although *Cladophora* has not been grown in channels, other filamentous green algae (FGA) (*Mougeotia*, *Stigeoclonium*, *Ulothrix*) have dominated in such experiments. In contrast, bottle tests with unattached *Cladophora* have shown that growth/biomass is not saturated at such low SRP concentrations (Pitcairn and Hawkes 1973), indicating results from flowing-water channel experiments more closely represent natural systems. Nevertheless, Bothwell (1989) did show added accrual of diatom films from about 250 mg/m^2 chl *a* at an SRP of 5 µg/L, increasing to 350 mg/m^2 at about 50 µg/L.

There may be problems with achieving a species assemblage in channel experiments that is representative of the natural stream(s) in question. In fact, accurate prediction or even chatacterization of ambient assemblages in dynamic systems may be challenging. *Cladophora* has been difficult, if not impossible, to establish in such systems, and other FGA have not established on Styrofoam substrata (used by Bothwell 1985), even when abundant in the source stream. Diatoms are usually first to establish, with more time required for FGA to colonize due to their more complex reproduction requirements. Natural stones seem to be the most effective substratum for colonizing either diatoms or FGA in these systems, but resulting dominant taxa in channels may not replicate exactly as in natural streams, even though channels are inoculated from stream rocks. Moreover, diatoms may, in fact, dominate the biomass in channels even though FGA establishes and appears most abundant to the eye. Correctly predicting community composition in future stages of succession is very difficult, even in simple systems. Given the complexity inherent in dynamic ecosystems, only excessively broad predictions may be possible. Data gathered from channel experiments may be little better at characterizing process than a grab sample is at characterizing water chemistry. Only simple extrapolations can be made employing data gathered from simple systems.

Caution is recommended in applying nutrient-biomass relationships developed in channel experiments to natural streams, primarily for two reasons: (1) TP and TN content required to produce a maximum biomass will probably be higher in natural streams than in channels, as previously discussed, because more detrital TP and TN will accumulate in enriched natural streams than in short-detention time channels. Hence, the yield (i.e., slope of regression line) of chl *a*/TP or TN in channels will be greater. (2) The more or less continual input of soluble nutrients from groundwater to the natural stream is usually unknown, so inflow soluble nutrient-maximum biomass relations from short-detention time channels may not be applicable to natural streams where in-stream soluble nutrients are low as a result of algal uptake during long travel times, yet may have a relatively high inflow concentration of soluble nutrients.

OTHER ISSUES TO KEEP IN MIND

Changes in certain physical factors including: (1) riparian vegetation; (2) total suspended solids (TSS); (3) reduced flow following scouring-flood conditions; (4) greatly reduced summer flow due to prolonged drought (somewhat common); or (5) reduced grazing may cause nuisance algal growths in stream systems. Identifying the controlling physical constraint(s), should be rather straightforward. If the stream is shaded, available light at the streambed should be measured to determine the extent to which photosynthesis is inhibited (Jasper and Bothwell 1986; Boston and Hill 1991). Shading can substantially reduce production (Welch et al. 1992), even though photosynthesis of periphyton is usually saturated at relatively low intensities (<25% full sunlight; Boston and Hill 1991). Turbidity can inhibit periphyton at relatively low levels (>10 NTU) (Quinn et al. 1992).

Biggs (1996) argued that flood disturbance is "perhaps the fundamental factor" determining the physical suitability for algal accrual in unshaded streams. Floods act as a "reset" mechanism, initiating a new cycle of accrual, succession, and loss due to grazing. Post-flood (scour) accrual rates are related to enrichment level (Lohman et al. 1992). The role of scouring high flow should be readily discernible from flow records and the seasonal pattern of periphyton accrual (Biggs 1996).

Flow can also regulate biomass. For example, *Cladophora* was observed to reach high biomass followed by senescence and detachment from substrata in enriched, unregulated northern California rivers, which experienced winter flooding and scour (Power 1992). In regulated rivers, where the flood, scour, and re-growth phenomenon did not occur, low biomass levels of *Cladophora* were maintained through grazing.

6.3 STATISTICAL ANALYSES

Statistical analyses are used to identify variability in data and to elucidate relationships among sampling parameters. Several statistical approaches for analyzing data are mentioned here. We advocate simple descriptive statistics for initial data analyses, i.e., calculating the mean, median, mode, ranges and standard deviation for each parameter in the system of interest. The National Nutrients Database discussed in Chapter 5 will calculate simple descriptive statistics for queried data. Creating a histogram or frequency distribution of the data for the class of streams of concern can identify the nutrient condition continuum for that class of streams. Specific recommendations for setting criteria using frequency distributions are discussed in Chapter 7, although the basis for the analysis is discussed here. Methods of statistical analyses are included in Appendix C to provide relevant references for the investigator if additional analyses are needed to understand and interpret data for criteria derivation.

FREQUENCY DISTRIBUTION

Frequency distributions can be used to aid in the setting of criteria. Frequency distributions do not require prior knowledge of individual stream condition prior to setting criteria. Criteria are based on and, in a sense, developed relative to the population of stream systems in the Region, State, or Tribe.

Data plotted on a scale of mean nutrient concentration versus frequency of occurrence in a specific stream class produces a frequency distribution of mean nutrient concentration. Plots of frequency distributions of mean TP, mean TN, mean chl *a*, and turbidity for the index period (discussed in Chapter 4) should be examined to determine the normalcy of the data in the distribution and to locate patterns for the class of streams being investigated. A sample size of thirty streams within a stream class is recommended for developing nutrient criteria. Smaller sample sizes will require more reference streams, more complete knowledge of the stream systems being investigated, more in-depth statistical analyses, and/or modeling to complete criteria derivation. Sample sizes smaller than thirty may be highly affected by extreme values in the dataset. Data that are not normally distributed are often transformed into a distribution more approximating the normal distribution by taking the logarithm of each value. Analysis of outliers may assist in explaining variability in small data sets. Additional analysis can be conducted to identify the statistical significance of population differences.

CORRELATION AND REGRESSION ANALYSES

The relationship between two variables may be of use in analyzing data for criteria derivation. Correlation and regression analyses allow the relationship to be defined in statistical terms. A correlation coefficient, usually identified as r, can be calculated to quantitatively express the relationship between two variables. The appropriate correlation coefficient is dependent on the scale of measurement in which each variable is expressed (whether the distribution of data is continuous or discrete) and, whether there is a linear or non-linear relationship. Results of correlation analyses may be represented by indicating the correlation coefficient, and represented graphically as a scatter diagram which plots all of the collected data, not just a measure of central tendency. The statistical significance of a calculated correlation coefficient can be determined with the t test. The t test is used to determine if there is a true relationship between two variables. Therefore, the null hypothesis states that there is no correlation between the data variables measured within the population. A critical α value is chosen as a criterion for determining whether to reject the null hypothesis. If the null hypothesis is rejected, the alternate hypothesis states that the correlation at the calculated r value between the two variables is significant.

Regression analyses provides a means of defining a mathematical relationship between two variables that permits prediction of one variable if the value of the other variable is known. In contrast to correlation analyses, there should be a true independent variable (a variable under the control of the experimenter) in regression analyses. Regression analyses establishes a relationship between two variables that allows prediction of the dependent variable (predicted variable) for a given value of an independent variable (predictor variable). However, scientists (other than statisticians) apply regression analyses to field data when a relationship is known to exist, even when there is no true independent variable (e.g., cell counts of algae and chlorophyll concentration; nutrient concentrations and chlorophyll concentration) (Ott 1988, 1995; Campbell 1989; Atlas and Bartha 1993).

TESTS OF SIGNIFICANCE

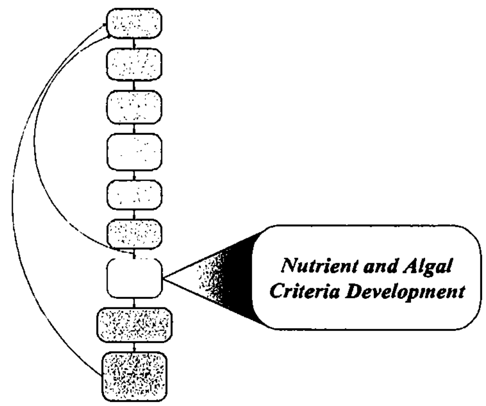
Various statistical tests are used to assess the hypotheses being tested. Statistical tests of significance differ in their applicability to the dataset of interest, and the power of the test (the ability of the test to detect a false null hypothesis). A parametric test of significance assumes a normal distibution of the population. Non-parametric analyses are valid for any type of distribution (normal, log-normal, etc.) and can be used if the data distribution is not normal or unknown. A parametric test has more power than a non-parametric test when its assumptions are satisfied. Two types of errors can be made when testing hypotheses: Type I–where a correct null hypothesis is mistakenly rejected, and Type II–when there is a failure to reject a false null hypothesis. The parametric test is less likely to make a Type II error, when the assumptions are met, than a non-parametric test. Therefore, if given a choice, the parametric test should be used rather than the non-parametric test when the assumptions of the parametric test are fulfilled. Less powerful, non-parametric tests of significance must be used in cases where the data do not fit the assumption of a normal distribution (Ott 1988; Campbell 1989; Atlas and Bartha 1993). Parametric tests include: the student *t* test, analysis of variance, multivariate analysis of variance, and multiple range tests. Non-parametric tests include: chi square, Mann Whitney U test; and the Kruskal - Wallis test (Ott 1988; Campbell 1989; Atlas and Bartha 1993) Detailed descriptions of these and other relevant statistical tests can be found in Appendix C.

6.4 USING MODELS AS MANAGEMENT TOOLS

Computer simulation modeling and probability testing can be used to predict responses to candidate criteria (i.e., numeric nutrient concentrations). Models that have been calibrated and verified can be used to extrapolate to a projected nutrient condition where existing data are either insufficient or unavailable. Data from the same system that is far removed from the present can be used if parameters can be adjusted to the present conditions. The model output can be compared to data from a similar stream system of the same class and in the same ecoregion for validation. Data from a similar system may also be used to extrapolate the nutrient condition when data for the system of interest are unavailable. In both cases, data are complemented by a set of clearly stated assumptions developed from data representing one point in time to estimate conditions in the future. In some instances, surrogate information such as turbidity and chl *a* concentration can be used to estimate nutrient concentrations.

Site-specific simulation models can also be developed for a system of interest, although this is frequently a time-consuming, expensive process. Site-specific computer simulation models should be solicited from the regional academic community, because they are more accurate for predicting specific waterbody concentrations and loadings. This section will not discuss site-specific model development, although several ecological and water quality modeling texts and articles can assist the investigator in developing such a model (see Fry [1993] and McIntire et al. [1996]). Appendix C provides information on several relevant stream water quality models.

Chapter 7. Nutrient and Algal Criteria Development



7.1 INTRODUCTION

This chapter addresses the details of developing scientifically defensible criteria for nutrients and algae. Three approaches are presented that water quality managers can use to derive numeric criteria for streams in their State/Tribal ecoregions. The approaches that are presented include: (1) the use of reference streams, (2) applying predictive relationships to select nutrient concentrations that will result in appropriate levels of algal biomass, and (3) developing criteria from thresholds established in the literature. Considerations are also presented for deriving criteria based on the potential for effects to downstream receiving waters (i.e., the lake, reservoir, or estuary to which the stream drains). The chapter concludes with the process for evaluating proposed criteria including the role of the Regional Technical Assistance Group (RTAG) in reviewing criteria, guidance for interpreting and applying criteria, considerations for sampling for comparison to criteria, potential revision of criteria, and final implementation of criteria into water quality standards.

The most rational approach for deriving criteria is to determine nutrient values in the absence of non-nutrient related factors that influence growth of algal biomass (e.g., light availability, flow). Then, refinements and exceptions to the criteria can be made based on the extent to which non-nutrient related factors are present for specific streams in an ecoregion or subecoregion. Thus, for both periphyton- and plankton-dominated systems, criteria should be set with the goal of reaching an acceptable algal biomass in streams with little or no light limitation, during periods of stable, post-flood/runoff, and moderate numbers of grazing invertebrates. For periphyton-dominated streams, substrata for attachment is assumed to be adequate and stable.

Expert evaluations are important throughout the criteria development process. The data upon which criteria are based and the analyses performed to arrive at criteria must be assessed for veracity and applicability. The EPA RTAGs are responsible for these assessments. The RTAG is composed of State, Tribal, and Regional specialists that will help the Agency and States/Tribes establish nutrient criteria for adoption into State/Tribal water quality standards. The RTAG is tasked with conducting an objective and exhaustive evaluation of regional nutrient information to establish protective nutrient criteria for the ecoregional waterbodies located in their EPA Region.

7.2 METHODS FOR ESTABLISHING NUTRIENT AND ALGAL CRITERIA

The following discussions focus on three methods that can be used in developing nutrient and algal criteria ranges. The first method requires identification of reference reaches for each established stream class based on either best professional judgement (BPJ) or percentile selections of data plotted as frequency distributions. The second method advocates refinement of trophic classification systems, use of models, and/or examination of system biological attributes to assess the relationships among nutrient and algal variables. The two methods described above should be based on data for the selected index period (see Chapter 4). Finally, the third method provides several published nutrient/algal thresholds that may be used (or modified for use) as criteria. A weight of evidence approach that combines one or more of the three approaches described below will produce criteria of greater scientific validity. This section also discusses how to develop criteria for streams that feed into standing receiving waters.

USING REFERENCE REACHES TO ESTABLISH CRITERIA

One approach that may be used in developing criteria is the reference reach approach. Reference reaches are relatively undisturbed stream segments that can serve as examples of the natural biological integrity of a region. There are three ways of using reference reaches to establish criteria.

1. Characterize reference reaches for each stream class within a region using best professional judgement and use these reference conditions to develop criteria.

2. Identify the 75th percentile of the frequency distribution of reference streams for a class of streams and use this percentile to develop the criteria (see Figure 8 and the Tennessee case study, Appendix A).

3. Calculate the 5th to 25th percentile of the frequency distribution of the general population of a class of streams and use the selected percentile to develop the criteria (Figure 8).

Identification of reference streams allows the investigator to arrange the streams within a class in order of nutrient condition (i.e., trophic state) from reference, to at risk, to impaired. Defining the nutrient condition of streams within a stream class allows the manager to identify protective criteria and determine priorities for management action. Criteria developed using reference reach approaches may require comparisons to similar systems in States or Tribes that share the ecoregion so that criteria can be validated, particularly when minimally-disturbed systems are rare.

Best professional judgement-based reference reaches may be identified for each class of streams within a State or Tribal ecoregion and then characterized with respect to algal biomass levels, algal community composition, and associated environmental conditions (including factors that affect algal levels such as nutrients, light, and substrate). The streams classified as reference quality by best professional judgement may be verified by comparing the data from the reference systems to general population data for each stream class. Reference systems should be minimally disturbed and should have primary parameter (i.e., TN, TP, chl *a*, and turbidity) values that reflect this condition. Factors that are affected by algae, such as DO and pH, should also be characterized. At least three minimally impaired reference systems should be identified for each stream class (see Chapter 2). Highest priority should be given to identifying reference streams for stream types considered to be at the greatest risk from impact by nutrients and algae, such as those with open canopy cover, good substrata, etc. [Conditions at the reference reach (e.g., algal biomass, nutrient concentrations) can be used in the development of criteria that are protective of high quality, beneficial uses for similar streams in the ecoregion.]

Alternatively, a reference condition for a stream class may be selected using either of two frequency distribution approaches. In both of the following approaches, an optimal reference condition value is selected from the distribution of an available set of stream data for a given stream class.

In the first frequency distribution approach, a percentile is selected (EPA generally recommends the 75th percentile) from the distribution of primary variables of known reference systems (i.e., highest quality or least impacted streams for that stream class within a region). As discussed in Chapter 3, primary variables are TP, TN, chl *a*, and turbidity or TSS. It is reasonable to select a higher percentile (i.e., 75th percentile) as the reference condition, because reference streams are already acknowledged to be in an approximately ideal state for a particular class of streams (Figure 8).

The second frequency distribution approach involves selecting a percentile of (1) all streams in the class (reference and non-reference) or (2) a random sample distribution of all streams within a particular class. Due to the random selection process, an upper percentile should be selected because the sample distribution is expected to contain some degraded systems. This option is most useful in regions where the number of legitimate "natural" reference water bodies is usually very small, such as highly developed land use areas (e.g., the agricultural lands of the Midwest and the urbanized east or west coasts). The EPA recommendation in this case is usually the 5th to 25th percentile depending upon the number of "natural" reference streams available. If almost all reference streams are impaired to some extent, then the 5th percentile is recommended.

Both the 75th percentile for reference streams and the 5th to 25th percentile from a representative sample distribution are only recommendations. The actual distribution of the observations should be the major determinant of the threshold point chosen. Figure 8 shows both options and illustrates the presumption that these two alternative methods should approach a common reference condition along a continuum of data points. In this illustration, the 75th percentile of the reference stream data distribution produces a TP reference condition of 20 μg/L. The 25th percentile of the random sample distribution produces a value of 25 μg/L. Because there is little distinction in this case, the Agency may select either 20 μg/L, 25 μg/L, or the intermediate 23 μg/L value as illustrated in Figure 8.

Each State or Tribe should similarly calculate its reference condition initially using both approaches to determine which method is most protective. The more conservative approach is recommended for subsequent reference condition calculations. A State or Tribe may choose to draw one single line vertically through the data distribution to set their criterion (the equivalent of the line drawn at the 23μg/L TP concentration shown in Figure 8). The obvious difficulty is choosing where the line is drawn. If drawn to the left of the central tendency point, most streams are in unacceptable condition and significant restoration management should occur. If the line is drawn to the right of the central tendency point, then most streams would be in acceptable condition and far less effort would be needed for

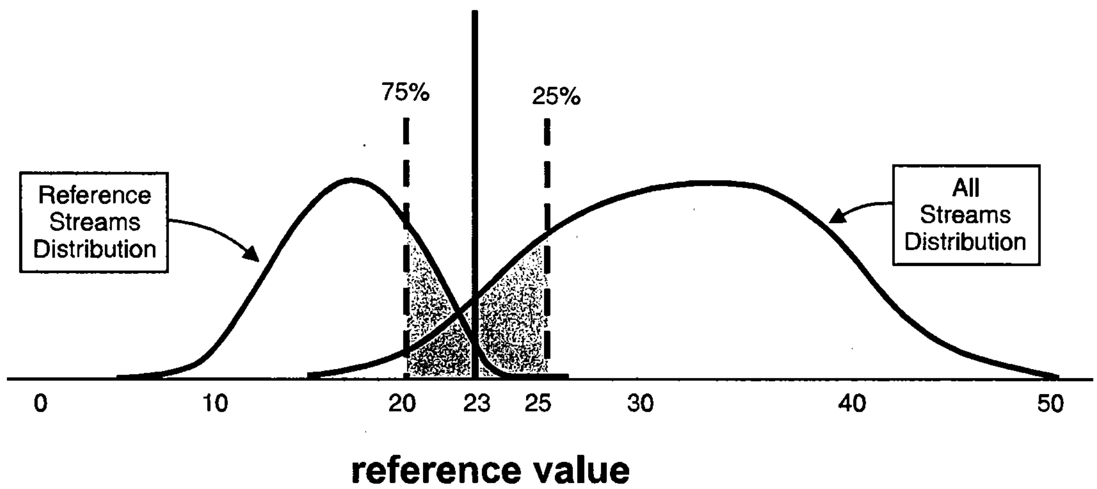


Figure 8. Selecting reference values for total phosphorus concentration (μg/L) using percentiles from reference streams and total stream populations.

restoration. The establishment of a reference condition helps to set the position of the line as objectively as possible.

It is important to understand that any line drawn through the data has certain ramifications; streams in unacceptable condition (on the right) should be dealt with through restoration. The streams to the left of the line are in acceptable condition, and should not be allowed to increase their nutrient concentrations. These streams should be protected according to the State's or Tribe's approved antidegradation policy, and through continued monitoring to assure that no future degradation occurs.

If a State or Tribe desires greater flexibility in setting their criteria, the frequency distribution can be divided into more than two segments (Figure 9). Using this approach, a criterion range is created and a greater number of stream systems fall within the criterion range. This approach divides systems into those that are of reference quality, currently in acceptable condition, or impaired. In this case, emphasis may be shifted from managing stream systems based on a central tendency (as shown above when a single line is drawn through the frequency distribution) to managing systems based on the level of impairment. This approach will also aid in prioritizing systems for protection and restoration. Stream data plotted to the right represent an increasingly degraded condition. Use of this approach requires that subsequent management efforts focus on improving stream conditions so that, over time, stream data plots shift to the left of their initial position.

State or Tribal water quality managers may also consider analyzing stream data based on designated use classifications. Using this approach, frequency distributions for specific designated uses could be examined and criteria proposed based on maintenance of high quality systems that are representative of each designated use.

In summary, frequency distributions can be used to aid in setting criteria. The number of divisions used has significant implications with respect to system management. A single criterion forces the manager to make decisions about the number of streams that will be in unacceptable condition, with considerable ramifications from that decision. If the distribution is divided into three segments, the majority of streams will be in acceptable condition (assuming that these streams are meeting their specified designated uses and do not contribute to downstream degradation of water quality), which will minimize management requirements. The method that is used may depend on the goals of the individual State or Tribe; some may wish to set criteria that encourage all State/Tribal stream systems to be preserved or restored to reference conditions. Other managers may consider additional options, such as developing criteria specific to protect the designated uses established for local streams.

USING PREDICTIVE RELATIONSHIPS TO ESTABLISH CRITERIA

The following section provides several options that can be used to evaluate nutrient and algal relationships in stream systems. These options include use of trophic state classifications, models, and biocriteria.

Trophic State Classification

One challenge associated with setting criteria is defining the relative trophic state of a stream. It is difficult to determine whether a stream is excessively eutrophic if its trophic state is not known relative

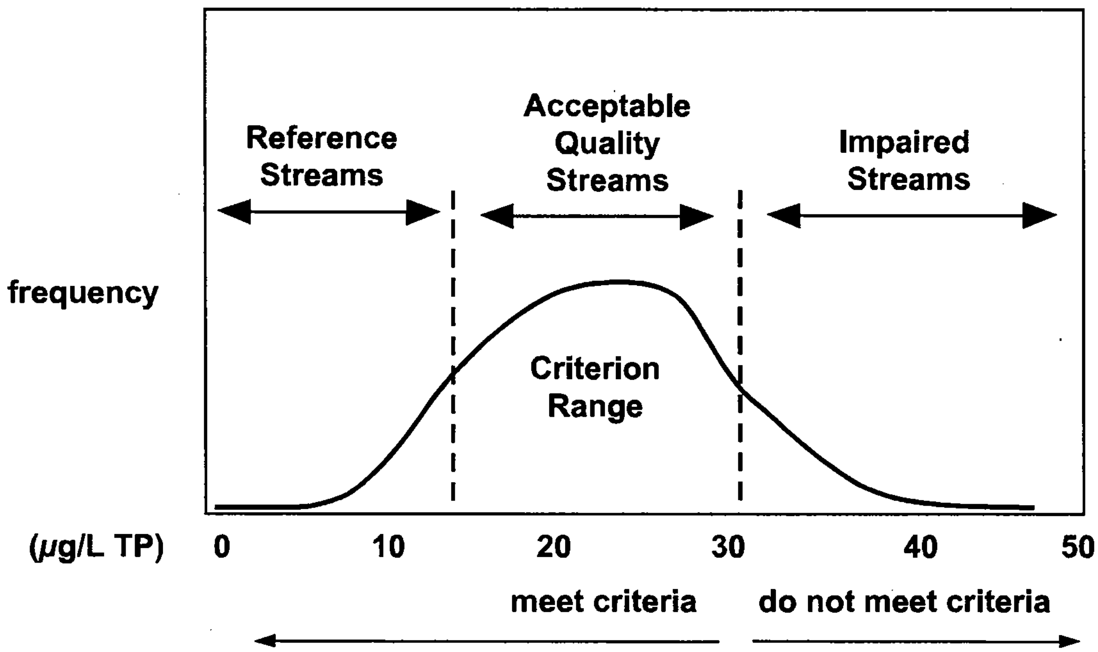


Figure 9. Frequency distribution divided into three segments that represent (from left to right) high-quality reference streams, acceptable quality streams, and impaired streams.

to other streams. There is no generally accepted system for classifying the trophic states of streams (Dodds et al. 1998). The only proposed system divides data plotted as cumulative frequency diagrams into oligotrophic (lower third), mesotrophic (middle third), and eutrophic (upper third) categories (see Chapter 2) (Dodds et al. 1998). This approach is similar to the reference reach method described in the previous section. More data are necessary to determine the applicability of such a classification scheme to streams from different ecoregions.

Models

A few models establish correlations between TN/TP and benthic algal biomass in streams (e.g., Lohman et al. 1992; Dodds et al. 1997; Bourassa and Cattaneo 1998; Chételat et al. 1999; Biggs 2000). Such models estimate algal biomass as a function of water column nutrients (as has often been done for lakes and reservoirs).

A regression model linking TP to river phytoplankton has been published (Van Nieuwenhuyse and Jones 1996). This model can be used to set TP criteria. The TP levels can in turn be used to calculate corresponding TN concentrations using the Redfield ratio (Harris 1986). This model captures additional variance when watershed area is considered (as discussed in Chapter 6).

Finally, it is necessary to relate instream TN and TP concentrations to nonpoint and point sources of nutrients. Models allowing prediction of nutrient loading in streams are needed. A method for determining instream TN and TP concentrations based on loading from point sources has been developed for use in the Clark Fork River (Dodds et al. 1997). Simple correlation techniques using data available from various regions may yield a nutrient and chlorophyll relationship that can be used to predict what management strategies are necessary to bring nutrients from point sources, and consequently algal biomass, to target levels.

Biocriteria

Biocriteria involve the use of biological parameters to establish nutrient impairment in streams. There are two ways to use biocriteria to establish water quality criteria. The first approach involves the protection and restoration of ecosystem services, which is almost exclusively related to biological features and functions in aquatic ecosystems. Although it is recognized that chemical and physical factors play a critical role in the algal-nutrient relationship, it is felt that the effect of nutrients on algae and other components of aquatic ecosystems is critical. This is why ecoregional and waterbody-specific nutrient criteria are recommended and chl *a* and Secchi depth/turbidity, arguably biocriteria, are required. The second approach is based on the concept that attributes of biological assemblages vary less in space and time than most physical and chemical characteristics. Thus, fewer mistakes in assessment may occur if biocriteria are employed in addition to physical and chemical criteria.

Multimetric indices are a special form of biocriteria in which many metrics are used to summarize and communicate in one number the state of a complex ecological system. Multimetric indices for macroinvertebrates and fish are used successfully as biocriteria in many States. A multimetric index of trophic status could be developed to complement N, P, and chl *a* criteria (see Section 6.2, Developing Multimetric Indices to Complement Nutrient Criteria).

The same approaches used to establish nutrient and algal criteria could be employed to establish criteria for other biological attributes, such as a Diatom Index of Trophic State (DITS). Frequency distributions of reference conditions or a random sample of streams would provide a target for management and restoration efforts. Alternatively, dose-response relations (predictive models) between biocriteria and nutrients could be used to set nutrient and biocriteria, based on a desired level of biotic integrity or other valued ecosystem component.

A fourth approach is also possible when characterizing the responses of many biological attributes to nutrients. Some of these factors change linearly with increasing nutrient concentrations, for a number of reasons, and some factors change non-linearly. Non-linear changes in metrics indicate thresholds along environmental gradients where small changes in environmental conditions cause relatively great changes in a biological attribute. These thresholds are valuable for setting nutrient criteria, but changes in these metrics are not necessarily the best indicators of biotic integrity. They can for example, remain relatively constant as human disturbance increases until a stress threshold is reached. Alternatively, during restoration, they may not respond to remediation until a lower threshold is reached. Thus, metrics or indices that change linearly (typically higher-level community attributes such as diversity or a multimetric index) provide better variables for establishing biocriteria because they respond to environmental change along the entire gradient of human disturbance. However, parameters changing non-linearly along environmental gradients are valuable for determining where along the environmental gradient the physical and chemical criteria should be set and, correspondingly, where to establish other biocriteria.

USING PUBLISHED NUTRIENT THRESHOLDS OR RECOMMENDED ALGAL LIMITS

In addition to using the 'reference reach' concept or applying predictive relationships to establish criteria for trophic state variables, other methods to consider include using thresholds and criteria already recommended in the literature. These approaches might be used as limits if identifying reference reaches proves difficult or as temporary measures until reference reaches can be adequately described. The following text describes potential criteria for several nutrient-related variables. Because most of the following threshold concentrations were derived primarily for northern to mid-temperate cobble-bottom streams, caution should be exercised when applying them to streams found in other geographic areas such as southern temperate and subtropical regions. The nutrient/algal relationships described below may not be valid for sandy streams of the southeast and southwest and should be tested on intermittent and effluent-dominated systems. Literature values may be used as criteria if a strong rationale is presented that demonstrates the suitability of the threshold value to the stream of interest (i.e., the system of interest should share characteristics with the systems used to derive the threshold, published values).

Nutrients

Criteria for nutrients in streams have been set or suggested by various agencies and investigators (Table 4). However, in contrast to lake management schemes, there is much less agreement on whether to use total nutrient concentrations, soluble nutrient concentrations, or nutrient concentrations that might produce a given biomass level or an undesirable effect in gravel-bed streams. Although much of the total nutrient concentrations in the water column of streams is not immediately available (due to a high fraction of detritus, as discussed previously), total concentrations probably have more general applicability than soluble fractions. While soluble fractions are more available, they also may be held at low levels during high-biomass periods due to uptake (Dodds et al. 1997). Nevertheless, some investigators have had considerable success relating soluble nutrients to algal biomass if annual mean or seasonal values are used for nutrient concentrations. Using the Bow River as an example, mean TDP during summer was more useful than TP (Table 4).

Table 4. Nutrient (µg/L) and algal biomass criteria limits recommended to prevent nuisance conditions and water quality degradation in streams based either on nutrient-chlorophyll *a* relationships or preventing risks to stream impairment as indicated.

PERIPHYTON Maximum in mg/m²						
TN	**TP**	**DIN**	**SRP**	**Chlorophyll *a***	**Impairment Risk**	**Source**
				100-200	nuisance growth	Welch et al. 1988, 1989
275-650	38-90			100-200	nuisance growth	Dodds et al. 1997
1500	75			200	eutrophy	Dodds et al. 1998
300	20			150	nuisance growth	Clark Fork River Tri-State Council, MT
	20				*Cladophora* nuisance growth	Chetelat et al. 1999
	10-20				*Cladophora* nuisance growth	Stevenson unpubl. data
		430	60		eutrophy	UK Environ. Agency 1988
		100[1]	10[1]	200	nuisance growth	Biggs 2000
		25	3	100	reduced invertebrate diversity	Nordin 1985
			15	100	nuisance growth	Quinn 1991
		1000	10[2]	~100	eutrophy	Sosiak pers. comm.
PLANKTON Mean in µg/L						
TN	**TP**	**DIN**	**SRP**	**Chlorophyll *a***	**Impairment Risk**	**Source**
300[3]	42			8	eutrophy	Van Nieuwenhuyse and Jones 1996
	70			15	chlorophyll action level	OAR 2000
250[3]	35			8	eutrophy	OECD 1992 (for lakes)

[1]30-day biomass accrual time
[2]Total Dissolved P
[3]Based on Redfield ratio of 7.2N:1P (Smith et al. 1997)

Notwithstanding the sparse set of cases, there is an indication of some consistency for total and soluble P criteria (Table 4). In two separate data sets, the tendency for *Cladophora* to begin dominating the periphyton was observed at TP concentrations of 10-20 µg/L (Chetelat et al. 1999; Stevenson pers. comm.). This general range was also selected by the Clark Fork Tri-State Council to limit maximum biomass to levels below 150 mg chl *a*/m^2. Setting a criterion equivalent to 'no filamentous green algae', even if chl *a* levels exceed 150 mg/m^2, would protect aesthetic use and still may not limit fisheries production.

Using a criterion for periphytic or planktonic biomass to initially judge if nutrient concentrations are excessive, may have a practical management and enforcement appeal. Advantages are several: (1) there is general agreement among some investigators and agencies on a biomass level that minimizes risk to recreational and aquatic life uses (see Table 4), (2) problems of algal control that result in poor dose-response relationships of nutrients versus biomass (due to shading by riparian canopies or suspended sediment and grazing) are averted, and (3) TMDLs and resultant controls would be required only for situations in which biomass criteria were exceeded. However, criteria for nutrients (specifically TN and TP) will ultimately be required for all stream classes within an ecoregion.

Algal Biomass

Criteria for levels of periphyton algal biomass that present a nuisance condition in streams and impact aesthetic use have been recommended by several investigators. There is surprising consistency in these values, with a maximum of about 150 mg/m^2 chl *a* being a generally agreed upon criterion (Table 4). As objective support for that criterion, percent coverage by filamentous forms was less than 20 percent, but increased with increased biomass and noticeably affected aesthetic quality (Welch et al. 1988). At this level, there were no apparent effects on DO, pH, or benthic invertebrates, which, as described earlier, occur at higher biomass levels.

Furthermore, a literature review of 19 cases indicated biomass levels greater than 150 mg/m^2 tended to occur with enrichment and when filamentous forms were more prevalent (Horner et al. 1983). As noted earlier, Lohman et al. (1992) observed that biomass rapidly recovered following flood-scour events in 12 Ozark streams when biomass exceeded the 150 mg/m^2 level at moderately to highly enriched sites. Pre-disturbance biomass did not recover as rapidly when initial levels did not exceed approximately 75 mg/m^2 at unenriched sites.

A provisional guideline of a maximum 100 mg/m^2 chl *a* and 40 percent coverage of filamentous forms was proposed for New Zealand streams to "protect contact recreation". There was insufficient evidence for protection of other uses that require specific DO and pH thresholds, which in turn vary due to atmospheric exchange (area:volume ratio) and buffering capacity (Quinn 1991).

While the 150 mg/m^2 level cannot be supported as an absolute threshold above which adverse effects on water quality and benthic habitat readily occur, it nonetheless is a level below which an aesthetic quality use will probably not be appreciably degraded by filamentous mats or any other of the adverse effects attributed to dense mats of filamentous algae (e.g., objectionable taste and odors in water supplies and fish flesh, impediment of water movement, clogging of water intakes, restriction of intra-gravel water flow and DO replenishment, DO/pH flux in the water column, or degradation of benthic habitat) (Welch 1992). Avoidance of these problems in many stream systems may be achieved with a maximum 150 mg/m^2 chl *a* criterion. As an example, control strategies were developed for the Clark Fork River, Montana, using a 100-150 mg/m^2 maximum as a criterion (see Appendix A case studies) (Watson and Gestring 1996; Dodds et al. 1997).

CONSIDERATIONS FOR DOWNSTREAM RECEIVING WATERS

More stringent nutrient criteria may be required for streams that feed into lentic or standing waters. For example, it is proposed that 35 µg/L TP concentration and a mean concentration of 8 µg/L chl *a* constitute the dividing line between eutrophic and mesotrophic lakes (OECD 1982). In contrast, data from Dodds et al. (1997) suggest that seasonal mean chlorophyll *a* values within stream systems of 100 mg/m^2 are likely at concentrations of 221 µg/L TP. Thus, unacceptable levels of chlorophyll may occur in lakes at much lower nutrient concentrations compared to streams (Dodds and Welch 2000).

7.3 EVALUATION OF PROPOSED CRITERIA

During criteria derivation, the RTAG will provide expert assessment of any proposed criteria or criteria ranges and their applicability to all streams within the class of interest. Criteria will need to be verified in many cases by comparing criteria values for a stream class within an ecoregion across State and Tribal boundaries. In addition, prior to recommending any proposed criterion, the RTAG must consider the potential for the proposed criterion to cause degradation of downstream receiving waters. In developing criteria, States/Tribes must consider the designated uses and standards of downstream waters and ensure that their water quality standards provide for the attainment and maintenance of water quality standards in downstream waters. Criteria recommended by the RTAG can be adopted by the State or Tribe as approved by EPA if there is documented evidence that no adverse effects will result downstream. However, if downstream waters are not adequately protected at the concentration level associated with the proposed criteria, then the criteria should be adjusted accordingly. Load estimating models, such as those recommended by EPA (USEPA 1999), can assist in this determination (see Section 4.2, Nutrient Load Attenuation). Water quality managers responsible for downstream receiving waters should also be consulted.

GUIDANCE FOR INTERPRETING AND APPLYING CRITERIA

After evaluating criteria proposed for each stream class, determining streams condition in comparison with nutrient criteria can be made by following the steps:

1. Calculate duration and frequency of criteria violations as well as associated consequences. This can be done using modeling techniques or correlational analysis of existing data.

2. Develop and test hypothesis to determine agreement with criteria. Analyze for alpha and beta (Type I and II) errors (see Appendix C).

3. Reaffirm appropriateness of criteria for protecting designated uses and meeting water quality standards.

The goal is to identify protective criteria and standards. Criteria should be based on ecologically significant changes as well as statistically significant differences in compiled data. Although criteria are developed exclusively on scientifically defensible methods, assignation of designated uses requires consideration of social, political, and economic factors. Thus, it is imperative that some thought be given during the criteria development process of how realistically the criteria can be implemented into standards that are accepted by the local public.

SAMPLING FOR COMPARISON TO CRITERIA

Once criteria have been selected for each indicator variable, a procedural rule to assess stream concurrence with criteria should be established. The four primary criteria variables include two causal variables (TN and TP) and two response variables (chl *a* and Secchi depth or a similar indicator of turbidity). Failure to meet either of the causal criteria should be sufficient to require remediation and typically the biological response, as measured by chl *a* and turbidity, will follow the nutrient trend. Should the causal criteria be met, but some combination of response criteria are not met, then a decisionmaking protocol should be in place to resolve the issue of whether the stream in question meets the proposed nutrient criteria.

Sampling to evaluate agreement with the standards implemented from nutrient and algal criteria will have to be carefully defined to ensure that State or Tribal sampling is compatible with the procedures used to establish the criteria. If State or Tribal observations are averaged over the year, balanced sampling is essential and the average should not exceed the criterion. In addition, no more than ten percent of the observations contributing to that average value should exceed the criterion.

A load estimating model (e.g., BASINS [see Appendix C]) may be applied to a watershed to back-calculate the criteria concentration for an individual stream from its load allocation. This approach to criteria determination may also be applied on a seasonal basis and should help States/Tribes relate their stream reach criteria with their lake or estuarine criteria. It may also be particularly important for criteria developed for streams and rivers that cross State/Tribal boundaries.

Algal Sampling for Comparison to Criteria

Once criteria for algal biomass have been established, certain sampling considerations must be addressed to obtain meaningful samples. This section discusses some of the more relevant considerations, using several questions as the basis for determining stream condition with respect to nutrients and algae.

1. How can algal criteria be applied to samples that come from only certain depths of the stream? Aesthetic criteria should be applied to the wadeable portion of large rivers, as has been done in British Columbia (Nordin 1985; see Table 4). The level necessary to protect aquatic life is likely to be system-specific and is best evaluated by determining how algal biomass affects DO, pH, and aquatic communities.

2. How large an area must exceed an algal criterion (e.g., 150 mg chl *a*/m^2) to be considered unacceptable? The area must be large enough to interfere with aesthetics and recreation or to cause undesirable water quality changes. Obviously, regional and site-specific testing of criteria will be necessary. The related sampling question is: how large an area should be characterized when assessing whether a reach exceeds a quantitative criterion? To ensure that a reasonably representative portion of a reach is sampled, replicate samples should be distributed over a reach at least 100 m long. Before selecting a point for sampling, a walk upstream and downstream a few hundred meters should be conducted to ensure that the preferred sampling point is not atypical of the reach being characterized.

Low altitude aerial photos taken on a sunny day in mid-to-late growing season can be used to determine the longitudinal extent of conditions similar to those at the sampling site. Floating the stream by boat can serve a similar purpose.

3. For how long must algal biomass exceed criteria to be considered unacceptable?
Attached algal biomass does not change as rapidly as water column parameters. Hence, one sample a month(from June to September) may be adequate to assess algal biomass, though weekly or bi-weekly sampling is ideal. If only two samplings can be afforded, the likely period containing the highest biomass levels should be bracketed. However, such a sampling scheme may be regarded as unacceptable if both sample values exceed aesthetic criteria. If algal biomass is high enough to cause excessive DO and pH fluctuations that violate water quality standards or that release toxins at unacceptable levels, then the time frames for those water quality violations should be used to judge the acceptability of algal biomass levels. As an example, some States or Tribes might regard the exceedance of algal biomass criteria once in 10 years (i.e., only during the 10-year low-flow) as acceptable, but more frequent exceedances may be deemed unacceptable.

4. How many replicate samples at a site are needed to obtain acceptable precision of data in order to detect differences between sites and changes over time? This depends on the variability in algal biomass in the particular system. The Kendall test with Sen slope estimate (Hirsch et al. 1982) allows the determination of the number of replicate samples needed to detect a certain percent change in annual means of a variable or a certain percent trend over a period such as 10 years (see Clark Fork River case study, Appendix A).

CRITERIA MODIFICATIONS

There may be specific cases identified by States or Tribes that require modification of established criteria, either due to unique stream system characteristics or specific designated uses approved for a stream or stream reach. Two examples of acceptable criteria modifications are presented below.

Site Specific Criteria
If a State/Tribe has additional information and data which indicate a different value or set of values is more appropriate for specific stream systems than ecoregionally-derived criteria, a scientifically defensible argument should be prepared that a "site specific" criteria modification is required. Once approved by EPA, this value can be incorporated into State or Tribal water quality standards. If no action is taken by the State or Tribe involved, EPA may propose to promulgate criteria based on the regional values and best available supporting science at the time.

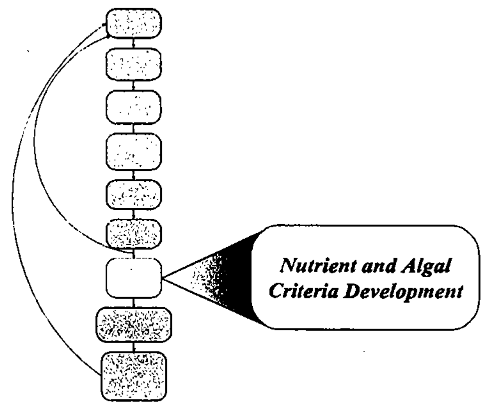
Designated Use Approaches
Once a regional criterion has been established, it is subject to periodic review and calibration. Any State or Tribe in the region may elect to use the criterion as the basis for developing its own criteria to protect designated uses for specific stream classes. This is entirely appropriate as long as the criteria are as protective as the basic EPA criterion for that region. This ecoregional criterion represents EPA's "304(a)" recommendation for protection of an aquatic life use.

The Clean Water Act as amended (Pub. L. 92-500 (1972), 33 U.S.C. 1251, *et seq.*) requires all States to establish designated uses for their waters (Section 303[c]). Designated uses are set by the State. EPA's interpretation of the Clean Water Act requires that wherever attainable, standards should provide for the protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the water (Section 101[a]). Other uses identified in the Act include industrial, agricultural, and public water supply. However, no waters may be designated for use as repositories for pollutants (see 40 CFR 131.10[a]). Each water body must have legally applicable criteria or measures of appropriate water quality that protect and maintain the designated use of that water. It is therefore proper for States and Tribes to set nutrient criteria appropriate to each of their designated uses in so far as they are as protective as the regional nutrient criteria established for those classes of waters.

IMPLEMENTATION OF NUTRIENT CRITERIA INTO WATER QUALITY STANDARDS

Criteria, once developed and adopted into water quality standards by a State or Tribe, are submitted to EPA for review and approval (see 40 CFR 131). EPA reviews the criteria (40 CFR 131.5) for consistency with the requirements of the Clean Water Act and 40 CFR 131.6, which requires that water quality criteria be sufficient to protect the designated use (40 CFR 131.6[c] and 40 CFR 131.11). The procedures for State/Tribal review and revision of water quality standards, EPA review and approval of water quality standards, and EPA promulgation of water quality standards (upon disapproval of State/Tribal water quality standards) are found at 40 CFR 131.20 -22 (see Figure 1, Chapter 1). The Water Quality Standards Handbook (EPA 1994) provides guidance for the implementation of these regulations.

Chapter 7. Nutrient and Algal Criteria Development



7.1 INTRODUCTION

This chapter addresses the details of developing scientifically defensible criteria for nutrients and algae. Three approaches are presented that water quality managers can use to derive numeric criteria for streams in their State/Tribal ecoregions. The approaches that are presented include: (1) the use of reference streams, (2) applying predictive relationships to select nutrient concentrations that will result in appropriate levels of algal biomass, and (3) developing criteria from thresholds established in the literature. Considerations are also presented for deriving criteria based on the potential for effects to downstream receiving waters (i.e., the lake, reservoir, or estuary to which the stream drains). The chapter concludes with the process for evaluating proposed criteria including the role of the Regional Technical Assistance Group (RTAG) in reviewing criteria, guidance for interpreting and applying criteria, considerations for sampling for comparison to criteria, potential revision of criteria, and final implementation of criteria into water quality standards.

The most rational approach for deriving criteria is to determine nutrient values in the absence of non-nutrient related factors that influence growth of algal biomass (e.g., light availability, flow). Then, refinements and exceptions to the criteria can be made based on the extent to which non-nutrient related factors are present for specific streams in an ecoregion or subecoregion. Thus, for both periphyton- and plankton-dominated systems, criteria should be set with the goal of reaching an acceptable algal biomass in streams with little or no light limitation, during periods of stable, post-flood/runoff, and moderate numbers of grazing invertebrates. For periphyton-dominated streams, substrata for attachment is assumed to be adequate and stable.

Expert evaluations are important throughout the criteria development process. The data upon which criteria are based and the analyses performed to arrive at criteria must be assessed for veracity and applicability. The EPA RTAGs are responsible for these assessments. The RTAG is composed of State, Tribal, and Regional specialists that will help the Agency and States/Tribes establish nutrient criteria for adoption into State/Tribal water quality standards. The RTAG is tasked with conducting an objective and exhaustive evaluation of regional nutrient information to establish protective nutrient criteria for the ecoregional waterbodies located in their EPA Region.

7.2 METHODS FOR ESTABLISHING NUTRIENT AND ALGAL CRITERIA

The following discussions focus on three methods that can be used in developing nutrient and algal criteria ranges. The first method requires identification of reference reaches for each established stream class based on either best professional judgement (BPJ) or percentile selections of data plotted as frequency distributions. The second method advocates refinement of trophic classification systems, use of models, and/or examination of system biological attributes to assess the relationships among nutrient and algal variables. The two methods described above should be based on data for the selected index period (see Chapter 4). Finally, the third method provides several published nutrient/algal thresholds that may be used (or modified for use) as criteria. A weight of evidence approach that combines one or more of the three approaches described below will produce criteria of greater scientific validity. This section also discusses how to develop criteria for streams that feed into standing receiving waters.

USING REFERENCE REACHES TO ESTABLISH CRITERIA

One approach that may be used in developing criteria is the reference reach approach. Reference reaches are relatively undisturbed stream segments that can serve as examples of the natural biological integrity of a region. There are three ways of using reference reaches to establish criteria.

1. Characterize reference reaches for each stream class within a region using best professional judgement and use these reference conditions to develop criteria.

2. Identify the 75th percentile of the frequency distribution of reference streams for a class of streams and use this percentile to develop the criteria (see Figure 8 and the Tennessee case study, Appendix A).

3. Calculate the 5th to 25th percentile of the frequency distribution of the general population of a class of streams and use the selected percentile to develop the criteria (Figure 8).

Identification of reference streams allows the investigator to arrange the streams within a class in order of nutrient condition (i.e., trophic state) from reference, to at risk, to impaired. Defining the nutrient condition of streams within a stream class allows the manager to identify protective criteria and determine priorities for management action. Criteria developed using reference reach approaches may require comparisons to similar systems in States or Tribes that share the ecoregion so that criteria can be validated, particularly when minimally-disturbed systems are rare.

Best professional judgement-based reference reaches may be identified for each class of streams within a State or Tribal ecoregion and then characterized with respect to algal biomass levels, algal community composition, and associated environmental conditions (including factors that affect algal levels such as nutrients, light, and substrate). The streams classified as reference quality by best professional judgement may be verified by comparing the data from the reference systems to general population data for each stream class. Reference systems should be minimally disturbed and should have primary parameter (i.e., TN, TP, chl *a*, and turbidity) values that reflect this condition. Factors that are affected by algae, such as DO and pH, should also be characterized. At least three minimally impaired reference systems should be identified for each stream class (see Chapter 2). Highest priority should be given to identifying reference streams for stream types considered to be at the greatest risk from impact by nutrients and algae, such as those with open canopy cover, good substrata, etc. [Conditions at the reference reach (e.g., algal biomass, nutrient concentrations) can be used in the development of criteria that are protective of high quality, beneficial uses for similar streams in the ecoregion.]

Alternatively, a reference condition for a stream class may be selected using either of two frequency distribution approaches. In both of the following approaches, an optimal reference condition value is selected from the distribution of an available set of stream data for a given stream class.

In the first frequency distribution approach, a percentile is selected (EPA generally recommends the 75th percentile) from the distribution of primary variables of known reference systems (i.e., highest quality or least impacted streams for that stream class within a region). As discussed in Chapter 3, primary variables are TP, TN, chl *a*, and turbidity or TSS. It is reasonable to select a higher percentile (i.e., 75th percentile) as the reference condition, because reference streams are already acknowledged to be in an approximately ideal state for a particular class of streams (Figure 8).

The second frequency distribution approach involves selecting a percentile of (1) all streams in the class (reference and non-reference) or (2) a random sample distribution of all streams within a particular class. Due to the random selection process, an upper percentile should be selected because the sample distribution is expected to contain some degraded systems. This option is most useful in regions where the number of legitimate "natural" reference water bodies is usually very small, such as highly developed land use areas (e.g., the agricultural lands of the Midwest and the urbanized east or west coasts). The EPA recommendation in this case is usually the 5th to 25th percentile depending upon the number of "natural" reference streams available. If almost all reference streams are impaired to some extent, then the 5th percentile is recommended.

Both the 75th percentile for reference streams and the 5th to 25th percentile from a representative sample distribution are only recommendations. The actual distribution of the observations should be the major determinant of the threshold point chosen. Figure 8 shows both options and illustrates the presumption that these two alternative methods should approach a common reference condition along a continuum of data points. In this illustration, the 75th percentile of the reference stream data distribution produces a TP reference condition of 20 μg/L. The 25th percentile of the random sample distribution produces a value of 25 μg/L. Because there is little distinction in this case, the Agency may select either 20 μg/L, 25 μg/L, or the intermediate 23 μg/L value as illustrated in Figure 8.

Each State or Tribe should similarly calculate its reference condition initially using both approaches to determine which method is most protective. The more conservative approach is recommended for subsequent reference condition calculations. A State or Tribe may choose to draw one single line vertically through the data distribution to set their criterion (the equivalent of the line drawn at the 23μg/L TP concentration shown in Figure 8). The obvious difficulty is choosing where the line is drawn. If drawn to the left of the central tendency point, most streams are in unacceptable condition and significant restoration management should occur. If the line is drawn to the right of the central tendency point, then most streams would be in acceptable condition and far less effort would be needed for

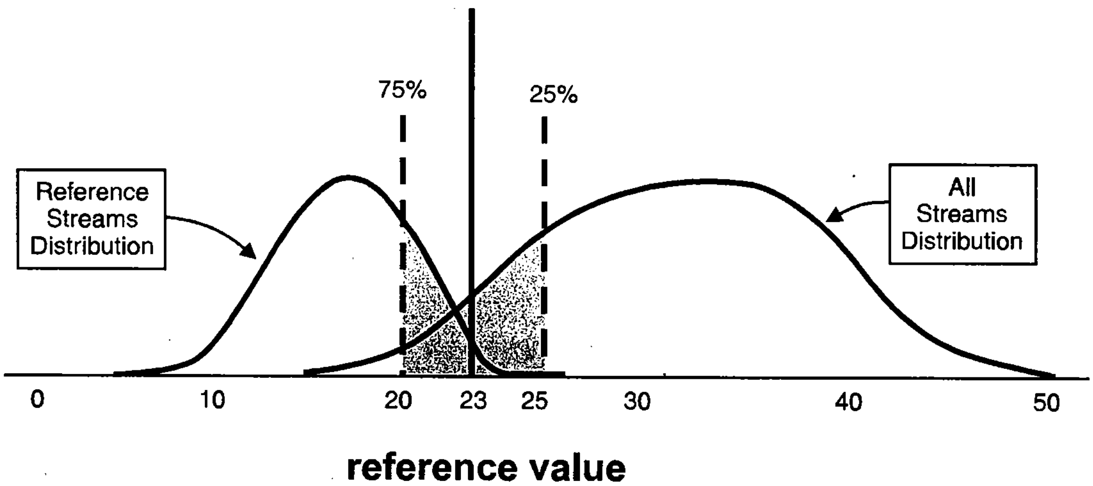


Figure 8. Selecting reference values for total phosphorus concentration (μg/L) using percentiles from reference streams and total stream populations.

restoration. The establishment of a reference condition helps to set the position of the line as objectively as possible.

It is important to understand that any line drawn through the data has certain ramifications; streams in unacceptable condition (on the right) should be dealt with through restoration. The streams to the left of the line are in acceptable condition, and should not be allowed to increase their nutrient concentrations. These streams should be protected according to the State's or Tribe's approved antidegradation policy, and through continued monitoring to assure that no future degradation occurs.

If a State or Tribe desires greater flexibility in setting their criteria, the frequency distribution can be divided into more than two segments (Figure 9). Using this approach, a criterion range is created and a greater number of stream systems fall within the criterion range. This approach divides systems into those that are of reference quality, currently in acceptable condition, or impaired. In this case, emphasis may be shifted from managing stream systems based on a central tendency (as shown above when a single line is drawn through the frequency distribution) to managing systems based on the level of impairment. This approach will also aid in prioritizing systems for protection and restoration. Stream data plotted to the right represent an increasingly degraded condition. Use of this approach requires that subsequent management efforts focus on improving stream conditions so that, over time, stream data plots shift to the left of their initial position.

State or Tribal water quality managers may also consider analyzing stream data based on designated use classifications. Using this approach, frequency distributions for specific designated uses could be examined and criteria proposed based on maintenance of high quality systems that are representative of each designated use.

In summary, frequency distributions can be used to aid in setting criteria. The number of divisions used has significant implications with respect to system management. A single criterion forces the manager to make decisions about the number of streams that will be in unacceptable condition, with considerable ramifications from that decision. If the distribution is divided into three segments, the majority of streams will be in acceptable condition (assuming that these streams are meeting their specified designated uses and do not contribute to downstream degradation of water quality), which will minimize management requirements. The method that is used may depend on the goals of the individual State or Tribe; some may wish to set criteria that encourage all State/Tribal stream systems to be preserved or restored to reference conditions. Other managers may consider additional options, such as developing criteria specific to protect the designated uses established for local streams.

USING PREDICTIVE RELATIONSHIPS TO ESTABLISH CRITERIA

The following section provides several options that can be used to evaluate nutrient and algal relationships in stream systems. These options include use of trophic state classifications, models, and biocriteria.

Trophic State Classification

One challenge associated with setting criteria is defining the relative trophic state of a stream. It is difficult to determine whether a stream is excessively eutrophic if its trophic state is not known relative

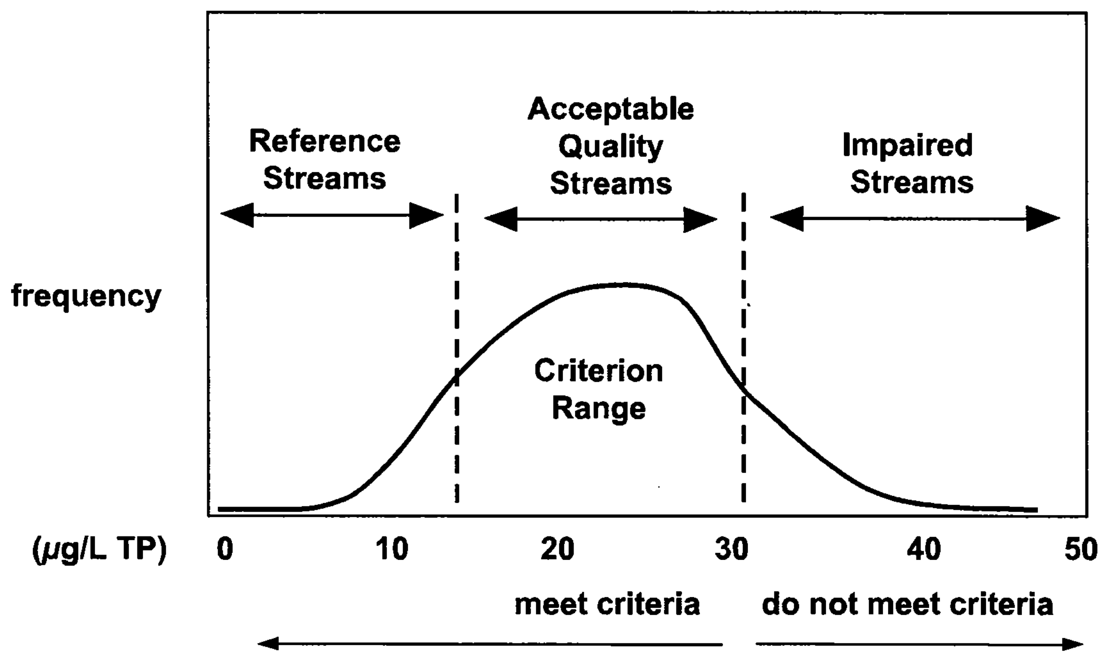


Figure 9. Frequency distribution divided into three segments that represent (from left to right) high-quality reference streams, acceptable quality streams, and impaired streams.

to other streams. There is no generally accepted system for classifying the trophic states of streams (Dodds et al. 1998). The only proposed system divides data plotted as cumulative frequency diagrams into oligotrophic (lower third), mesotrophic (middle third), and eutrophic (upper third) categories (see Chapter 2) (Dodds et al. 1998). This approach is similar to the reference reach method described in the previous section. More data are necessary to determine the applicability of such a classification scheme to streams from different ecoregions.

Models

A few models establish correlations between TN/TP and benthic algal biomass in streams (e.g., Lohman et al. 1992; Dodds et al. 1997; Bourassa and Cattaneo 1998; Chételat et al. 1999; Biggs 2000). Such models estimate algal biomass as a function of water column nutrients (as has often been done for lakes and reservoirs).

A regression model linking TP to river phytoplankton has been published (Van Nieuwenhuyse and Jones 1996). This model can be used to set TP criteria. The TP levels can in turn be used to calculate corresponding TN concentrations using the Redfield ratio (Harris 1986). This model captures additional variance when watershed area is considered (as discussed in Chapter 6).

Finally, it is necessary to relate instream TN and TP concentrations to nonpoint and point sources of nutrients. Models allowing prediction of nutrient loading in streams are needed. A method for determining instream TN and TP concentrations based on loading from point sources has been developed for use in the Clark Fork River (Dodds et al. 1997). Simple correlation techniques using data available from various regions may yield a nutrient and chlorophyll relationship that can be used to predict what management strategies are necessary to bring nutrients from point sources, and consequently algal biomass, to target levels.

Biocriteria

Biocriteria involve the use of biological parameters to establish nutrient impairment in streams. There are two ways to use biocriteria to establish water quality criteria. The first approach involves the protection and restoration of ecosystem services, which is almost exclusively related to biological features and functions in aquatic ecosystems. Although it is recognized that chemical and physical factors play a critical role in the algal-nutrient relationship, it is felt that the effect of nutrients on algae and other components of aquatic ecosystems is critical. This is why ecoregional and waterbody-specific nutrient criteria are recommended and chl *a* and Secchi depth/turbidity, arguably biocriteria, are required. The second approach is based on the concept that attributes of biological assemblages vary less in space and time than most physical and chemical characteristics. Thus, fewer mistakes in assessment may occur if biocriteria are employed in addition to physical and chemical criteria.

Multimetric indices are a special form of biocriteria in which many metrics are used to summarize and communicate in one number the state of a complex ecological system. Multimetric indices for macroinvertebrates and fish are used successfully as biocriteria in many States. A multimetric index of trophic status could be developed to complement N, P, and chl *a* criteria (see Section 6.2, Developing Multimetric Indices to Complement Nutrient Criteria).

The same approaches used to establish nutrient and algal criteria could be employed to establish criteria for other biological attributes, such as a Diatom Index of Trophic State (DITS). Frequency distributions of reference conditions or a random sample of streams would provide a target for management and restoration efforts. Alternatively, dose-response relations (predictive models) between biocriteria and nutrients could be used to set nutrient and biocriteria, based on a desired level of biotic integrity or other valued ecosystem component.

A fourth approach is also possible when characterizing the responses of many biological attributes to nutrients. Some of these factors change linearly with increasing nutrient concentrations, for a number of reasons, and some factors change non-linearly. Non-linear changes in metrics indicate thresholds along environmental gradients where small changes in environmental conditions cause relatively great changes in a biological attribute. These thresholds are valuable for setting nutrient criteria, but changes in these metrics are not necessarily the best indicators of biotic integrity. They can for example, remain relatively constant as human disturbance increases until a stress threshold is reached. Alternatively, during restoration, they may not respond to remediation until a lower threshold is reached. Thus, metrics or indices that change linearly (typically higher-level community attributes such as diversity or a multimetric index) provide better variables for establishing biocriteria because they respond to environmental change along the entire gradient of human disturbance. However, parameters changing non-linearly along environmental gradients are valuable for determining where along the environmental gradient the physical and chemical criteria should be set and, correspondingly, where to establish other biocriteria.

USING PUBLISHED NUTRIENT THRESHOLDS OR RECOMMENDED ALGAL LIMITS

In addition to using the 'reference reach' concept or applying predictive relationships to establish criteria for trophic state variables, other methods to consider include using thresholds and criteria already recommended in the literature. These approaches might be used as limits if identifying reference reaches proves difficult or as temporary measures until reference reaches can be adequately described. The following text describes potential criteria for several nutrient-related variables. Because most of the following threshold concentrations were derived primarily for northern to mid-temperate cobble-bottom streams, caution should be exercised when applying them to streams found in other geographic areas such as southern temperate and subtropical regions. The nutrient/algal relationships described below may not be valid for sandy streams of the southeast and southwest and should be tested on intermittent and effluent-dominated systems. Literature values may be used as criteria if a strong rationale is presented that demonstrates the suitability of the threshold value to the stream of interest (i.e., the system of interest should share characteristics with the systems used to derive the threshold, published values).

Nutrients

Criteria for nutrients in streams have been set or suggested by various agencies and investigators (Table 4). However, in contrast to lake management schemes, there is much less agreement on whether to use total nutrient concentrations, soluble nutrient concentrations, or nutrient concentrations that might produce a given biomass level or an undesirable effect in gravel-bed streams. Although much of the total nutrient concentrations in the water column of streams is not immediately available (due to a high fraction of detritus, as discussed previously), total concentrations probably have more general applicability than soluble fractions. While soluble fractions are more available, they also may be held at low levels during high-biomass periods due to uptake (Dodds et al. 1997). Nevertheless, some investigators have had considerable success relating soluble nutrients to algal biomass if annual mean or seasonal values are used for nutrient concentrations. Using the Bow River as an example, mean TDP during summer was more useful than TP (Table 4).

Table 4. Nutrient (µg/L) and algal biomass criteria limits recommended to prevent nuisance conditions and water quality degradation in streams based either on nutrient-chlorophyll *a* relationships or preventing risks to stream impairment as indicated.

PERIPHYTON Maximum in mg/m²						
TN	**TP**	**DIN**	**SRP**	**Chlorophyll *a***	**Impairment Risk**	**Source**
				100-200	nuisance growth	Welch et al. 1988, 1989
275-650	38-90			100-200	nuisance growth	Dodds et al. 1997
1500	75			200	eutrophy	Dodds et al. 1998
300	20			150	nuisance growth	Clark Fork River Tri-State Council, MT
	20				*Cladophora* nuisance growth	Chetelat et al. 1999
	10-20				*Cladophora* nuisance growth	Stevenson unpubl. data
		430	60		eutrophy	UK Environ. Agency 1988
		100[1]	10[1]	200	nuisance growth	Biggs 2000
		25	3	100	reduced invertebrate diversity	Nordin 1985
			15	100	nuisance growth	Quinn 1991
		1000	10[2]	~100	eutrophy	Sosiak pers. comm.
PLANKTON Mean in µg/L						
TN	**TP**	**DIN**	**SRP**	**Chlorophyll *a***	**Impairment Risk**	**Source**
300[3]	42			8	eutrophy	Van Nieuwenhuyse and Jones 1996
	70			15	chlorophyll action level	OAR 2000
250[3]	35			8	eutrophy	OECD 1992 (for lakes)

[1]30-day biomass accrual time
[2]Total Dissolved P
[3]Based on Redfield ratio of 7.2N:1P (Smith et al. 1997)

Notwithstanding the sparse set of cases, there is an indication of some consistency for total and soluble P criteria (Table 4). In two separate data sets, the tendency for *Cladophora* to begin dominating the periphyton was observed at TP concentrations of 10-20 µg/L (Chetelat et al. 1999; Stevenson pers. comm.). This general range was also selected by the Clark Fork Tri-State Council to limit maximum biomass to levels below 150 mg chl a/m^2. Setting a criterion equivalent to 'no filamentous green algae', even if chl *a* levels exceed 150 mg/m^2, would protect aesthetic use and still may not limit fisheries production.

Using a criterion for periphytic or planktonic biomass to initially judge if nutrient concentrations are excessive, may have a practical management and enforcement appeal. Advantages are several: (1) there is general agreement among some investigators and agencies on a biomass level that minimizes risk to recreational and aquatic life uses (see Table 4), (2) problems of algal control that result in poor dose-response relationships of nutrients versus biomass (due to shading by riparian canopies or suspended sediment and grazing) are averted, and (3) TMDLs and resultant controls would be required only for situations in which biomass criteria were exceeded. However, criteria for nutrients (specifically TN and TP) will ultimately be required for all stream classes within an ecoregion.

Algal Biomass

Criteria for levels of periphyton algal biomass that present a nuisance condition in streams and impact aesthetic use have been recommended by several investigators. There is surprising consistency in these values, with a maximum of about 150 mg/m^2 chl *a* being a generally agreed upon criterion (Table 4). As objective support for that criterion, percent coverage by filamentous forms was less than 20 percent, but increased with increased biomass and noticeably affected aesthetic quality (Welch et al. 1988). At this level, there were no apparent effects on DO, pH, or benthic invertebrates, which, as described earlier, occur at higher biomass levels.

Furthermore, a literature review of 19 cases indicated biomass levels greater than 150 mg/m^2 tended to occur with enrichment and when filamentous forms were more prevalent (Horner et al. 1983). As noted earlier, Lohman et al. (1992) observed that biomass rapidly recovered following flood-scour events in 12 Ozark streams when biomass exceeded the 150 mg/m^2 level at moderately to highly enriched sites. Pre-disturbance biomass did not recover as rapidly when initial levels did not exceed approximately 75 mg/m^2 at unenriched sites.

A provisional guideline of a maximum 100 mg/m^2 chl *a* and 40 percent coverage of filamentous forms was proposed for New Zealand streams to "protect contact recreation". There was insufficient evidence for protection of other uses that require specific DO and pH thresholds, which in turn vary due to atmospheric exchange (area:volume ratio) and buffering capacity (Quinn 1991).

While the 150 mg/m^2 level cannot be supported as an absolute threshold above which adverse effects on water quality and benthic habitat readily occur, it nonetheless is a level below which an aesthetic quality use will probably not be appreciably degraded by filamentous mats or any other of the adverse effects attributed to dense mats of filamentous algae (e.g., objectionable taste and odors in water supplies and fish flesh, impediment of water movement, clogging of water intakes, restriction of intra-gravel water flow and DO replenishment, DO/pH flux in the water column, or degradation of benthic habitat) (Welch 1992). Avoidance of these problems in many stream systems may be achieved with a maximum 150 mg/m^2 chl *a* criterion. As an example, control strategies were developed for the Clark Fork River, Montana, using a 100-150 mg/m^2 maximum as a criterion (see Appendix A case studies) (Watson and Gestring 1996; Dodds et al. 1997).

CONSIDERATIONS FOR DOWNSTREAM RECEIVING WATERS

More stringent nutrient criteria may be required for streams that feed into lentic or standing waters. For example, it is proposed that 35 µg/L TP concentration and a mean concentration of 8 µg/L chl *a* constitute the dividing line between eutrophic and mesotrophic lakes (OECD 1982). In contrast, data from Dodds et al. (1997) suggest that seasonal mean chlorophyll *a* values within stream systems of 100 mg/m^2 are likely at concentrations of 221 µg/L TP. Thus, unacceptable levels of chlorophyll may occur in lakes at much lower nutrient concentrations compared to streams (Dodds and Welch 2000).

7.3 EVALUATION OF PROPOSED CRITERIA

During criteria derivation, the RTAG will provide expert assessment of any proposed criteria or criteria ranges and their applicability to all streams within the class of interest. Criteria will need to be verified in many cases by comparing criteria values for a stream class within an ecoregion across State and Tribal boundaries. In addition, prior to recommending any proposed criterion, the RTAG must consider the potential for the proposed criterion to cause degradation of downstream receiving waters. In developing criteria, States/Tribes must consider the designated uses and standards of downstream waters and ensure that their water quality standards provide for the attainment and maintenance of water quality standards in downstream waters. Criteria recommended by the RTAG can be adopted by the State or Tribe as approved by EPA if there is documented evidence that no adverse effects will result downstream. However, if downstream waters are not adequately protected at the concentration level associated with the proposed criteria, then the criteria should be adjusted accordingly. Load estimating models, such as those recommended by EPA (USEPA 1999), can assist in this determination (see Section 4.2, Nutrient Load Attenuation). Water quality managers responsible for downstream receiving waters should also be consulted.

GUIDANCE FOR INTERPRETING AND APPLYING CRITERIA

After evaluating criteria proposed for each stream class, determining streams condition in comparison with nutrient criteria can be made by following the steps:

1. Calculate duration and frequency of criteria violations as well as associated consequences. This can be done using modeling techniques or correlational analysis of existing data.

2. Develop and test hypothesis to determine agreement with criteria. Analyze for alpha and beta (Type I and II) errors (see Appendix C).

3. Reaffirm appropriateness of criteria for protecting designated uses and meeting water quality standards.

The goal is to identify protective criteria and standards. Criteria should be based on ecologically significant changes as well as statistically significant differences in compiled data. Although criteria are developed exclusively on scientifically defensible methods, assignation of designated uses requires consideration of social, political, and economic factors. Thus, it is imperative that some thought be given during the criteria development process of how realistically the criteria can be implemented into standards that are accepted by the local public.

SAMPLING FOR COMPARISON TO CRITERIA

Once criteria have been selected for each indicator variable, a procedural rule to assess stream concurrence with criteria should be established. The four primary criteria variables include two causal variables (TN and TP) and two response variables (chl *a* and Secchi depth or a similar indicator of turbidity). Failure to meet either of the causal criteria should be sufficient to require remediation and typically the biological response, as measured by chl *a* and turbidity, will follow the nutrient trend. Should the causal criteria be met, but some combination of response criteria are not met, then a decisionmaking protocol should be in place to resolve the issue of whether the stream in question meets the proposed nutrient criteria.

Sampling to evaluate agreement with the standards implemented from nutrient and algal criteria will have to be carefully defined to ensure that State or Tribal sampling is compatible with the procedures used to establish the criteria. If State or Tribal observations are averaged over the year, balanced sampling is essential and the average should not exceed the criterion. In addition, no more than ten percent of the observations contributing to that average value should exceed the criterion.

A load estimating model (e.g., BASINS [see Appendix C]) may be applied to a watershed to back-calculate the criteria concentration for an individual stream from its load allocation. This approach to criteria determination may also be applied on a seasonal basis and should help States/Tribes relate their stream reach criteria with their lake or estuarine criteria. It may also be particularly important for criteria developed for streams and rivers that cross State/Tribal boundaries.

Algal Sampling for Comparison to Criteria

Once criteria for algal biomass have been established, certain sampling considerations must be addressed to obtain meaningful samples. This section discusses some of the more relevant considerations, using several questions as the basis for determining stream condition with respect to nutrients and algae.

1. How can algal criteria be applied to samples that come from only certain depths of the stream? Aesthetic criteria should be applied to the wadeable portion of large rivers, as has been done in British Columbia (Nordin 1985; see Table 4). The level necessary to protect aquatic life is likely to be system-specific and is best evaluated by determining how algal biomass affects DO, pH, and aquatic communities.

2. How large an area must exceed an algal criterion (e.g., 150 mg chl a/m^2) to be considered unacceptable? The area must be large enough to interfere with aesthetics and recreation or to cause undesirable water quality changes. Obviously, regional and site-specific testing of criteria will be necessary. The related sampling question is: how large an area should be characterized when assessing whether a reach exceeds a quantitative criterion? To ensure that a reasonably representative portion of a reach is sampled, replicate samples should be distributed over a reach at least 100 m long. Before selecting a point for sampling, a walk upstream and downstream a few hundred meters should be conducted to ensure that the preferred sampling point is not atypical of the reach being characterized.

Low altitude aerial photos taken on a sunny day in mid-to-late growing season can be used to determine the longitudinal extent of conditions similar to those at the sampling site. Floating the stream by boat can serve a similar purpose.

3. For how long must algal biomass exceed criteria to be considered unacceptable?
Attached algal biomass does not change as rapidly as water column parameters. Hence, one sample a month(from June to September) may be adequate to assess algal biomass, though weekly or bi-weekly sampling is ideal. If only two samplings can be afforded, the likely period containing the highest biomass levels should be bracketed. However, such a sampling scheme may be regarded as unacceptable if both sample values exceed aesthetic criteria. If algal biomass is high enough to cause excessive DO and pH fluctuations that violate water quality standards or that release toxins at unacceptable levels, then the time frames for those water quality violations should be used to judge the acceptability of algal biomass levels. As an example, some States or Tribes might regard the exceedance of algal biomass criteria once in 10 years (i.e., only during the 10-year low-flow) as acceptable, but more frequent exceedances may be deemed unacceptable.

4. How many replicate samples at a site are needed to obtain acceptable precision of data in order to detect differences between sites and changes over time? This depends on the variability in algal biomass in the particular system. The Kendall test with Sen slope estimate (Hirsch et al. 1982) allows the determination of the number of replicate samples needed to detect a certain percent change in annual means of a variable or a certain percent trend over a period such as 10 years (see Clark Fork River case study, Appendix A).

CRITERIA MODIFICATIONS

There may be specific cases identified by States or Tribes that require modification of established criteria, either due to unique stream system characteristics or specific designated uses approved for a stream or stream reach. Two examples of acceptable criteria modifications are presented below.

Site Specific Criteria
If a State/Tribe has additional information and data which indicate a different value or set of values is more appropriate for specific stream systems than ecoregionally-derived criteria, a scientifically defensible argument should be prepared that a "site specific" criteria modification is required. Once approved by EPA, this value can be incorporated into State or Tribal water quality standards. If no action is taken by the State or Tribe involved, EPA may propose to promulgate criteria based on the regional values and best available supporting science at the time.

Designated Use Approaches
Once a regional criterion has been established, it is subject to periodic review and calibration. Any State or Tribe in the region may elect to use the criterion as the basis for developing its own criteria to protect designated uses for specific stream classes. This is entirely appropriate as long as the criteria are as protective as the basic EPA criterion for that region. This ecoregional criterion represents EPA's "304(a)" recommendation for protection of an aquatic life use.

The Clean Water Act as amended (Pub. L. 92-500 (1972), 33 U.S.C. 1251, *et seq.*) requires all States to establish designated uses for their waters (Section 303[c]). Designated uses are set by the State. EPA's interpretation of the Clean Water Act requires that wherever attainable, standards should provide for the protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the water (Section 101[a]). Other uses identified in the Act include industrial, agricultural, and public water supply. However, no waters may be designated for use as repositories for pollutants (see 40 CFR 131.10[a]). Each water body must have legally applicable criteria or measures of appropriate water quality that protect and maintain the designated use of that water. It is therefore proper for States and Tribes to set nutrient criteria appropriate to each of their designated uses in so far as they are as protective as the regional nutrient criteria established for those classes of waters.

IMPLEMENTATION OF NUTRIENT CRITERIA INTO WATER QUALITY STANDARDS

Criteria, once developed and adopted into water quality standards by a State or Tribe, are submitted to EPA for review and approval (see 40 CFR 131). EPA reviews the criteria (40 CFR 131.5) for consistency with the requirements of the Clean Water Act and 40 CFR 131.6, which requires that water quality criteria be sufficient to protect the designated use (40 CFR 131.6[c] and 40 CFR 131.11). The procedures for State/Tribal review and revision of water quality standards, EPA review and approval of water quality standards, and EPA promulgation of water quality standards (upon disapproval of State/Tribal water quality standards) are found at 40 CFR 131.20 -22 (see Figure 1, Chapter 1). The Water Quality Standards Handbook (EPA 1994) provides guidance for the implementation of these regulations.

APPENDIX D. ACRONYM LIST AND GLOSSARY

ACRONYMS

AASF	Adopt-A-Stream Foundation
AFDM	Ash-Free Dry Mass
AFDW	Ash-Free Dry Weight
AGP	Algal Growth Potential
AI	Autotrophic Index
ANOVA	Analysis of Variance
APA	Alkaline Phosphatase Activity
B-IBI	Benthic Macroinvertebrate Index of Biological Integrity
BMP	Best Management Practice
BOD	Biochemical Oxygen Demand
BPJ	Best Professional Judgement
BuRec	U.S. Department of the Interior, Bureau of Reclamation
CENR	Committee on Environment and Natural Resources
CE-QUAL-RIV1	Hydrodynamic and Water Quality Model for Streams
CFR	Code of Federal Regulations
CGP	Construction General Permit
CLP	Clean Lakes Program
COE	Corps of Engineers
CPP	Continuing Planning Process
CREP	Conservation Reserve Enhancement Program
CRP	Conservation Reserve Program
CSO	Combined Sewer Overflow
CZARA	Costal Zone Act Reauthorization Amendment
DDT	Dichlorodiphenyltrichloroethane
DEQ	Department of Environmental Quality
DIN	Dissolved Inorganic Nitrogen
DITS	Diatom Index of Trophic Status
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
DWPC	Division of Water Pollution Control
ECA	Ecological Community Analysis
ECARP	Environmental Conservation Acreage Reserve Program
EDAS	Ecological Data Application System
EMAP	Environmental Monitoring and Assessment Program
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)
EQIP	Environmental Quality Incentives Program
FGA	Filamentous Green Algae
FIP	Forestry Incentives Program
GIS	Geographical Information Systems
HAB	Harmful Algal Bloom
HBN	Hydrologic Benchmark Network
HSFP	Hydrologic Simulation Project FORTAN

HUC	Hydrologic Unit Code
IBI	Index of Biological Integrity
LDC	Legacy Data Center
MITS	Multimetric Index of Trophic Status
N	Nitrogen
NASQAN	National Stream Quality Accounting Network
NAWQA	National Water-Quality Assessment
NIST	National Institute of Standards and Technology
NOAA	National Oceanic and Atmospheric Association
NPDES	National Pollutant Discharge and Elimination System
NPS	Nonpoint Source
NPSM	Nonpoint Source Model
NRCS	Natural Resources Conservation Service
NSCEP	National Service Center for Environmental Publications
NSS	National Stream Survey
NSWS	National Surface Water Survey
NTU	Nephelometric Turbidity Units
NWIS	National Water Information System
ONRW	Outstanding National Resource Waters
P	Phosphorus
PAR	Photosynthetically-active Radiation
PCS	Permit Compliance System
P/R	Productivity/Respiration
QA	Quality Assurance
QC	Quality Control
QUAL2E	Enhanced Stream Water Quality Model
RAD	Reach Address Database
RCC	River Continuum Concept
RF3	Reach File 3
RTAG	Regional Technical Assistance Groups
SAV	Submerged Aquatic Vegetation
SRP	Soluble Reactive Phosphorus
STORET	Storage and Retrieval
TAB	Total Algal Biomass
TDP	Total Dissolved Phosphorus
THM	trihalomethane
TIA	Total Impervious Area
TKN	Total Kjeldahl Nitrogen
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
TVA	Tennessee Valley Authority
TWINSPAN	Two Way Indicator Species Analysis
USGMA	Unweighted Pair Group Method Using Arithmetic Averages

USGS	United States Geologic Survey
VNRP	Voluntary Nutrient Reduction Plan
WASP	Water Analysis Simulation Program
WES	Waterways Experiment Station
WHIP	Wildlife Habitat Incentives Program
WLA	Waste Load Allocation
WQBEL	Water Quality Based Effluent Limits
WQN	Water Quality Networks
WQS	Water Quality Standards
WRS	Wetlands Reserve Program
χ^2	Chi Square

GLOSSARY

algal biomass
The weight of living algal material in a unit area at a given time (Wetzel 1983).

allochthonus
An organism or substance foreign to a given ecosystem (Atlas and Bartha 1993); describes organic matter reaching an aquatic community from the outside in the form of organic detritus or organic matter adsorbed to sediment (Wetzel 1983).

ash-free dry weight
An algal biomass measurement that measures the standing crop of algae to estimate net production (see Appendix B) (APHA 2000).

autochthonus
Microorganisms and/or substances indigenous to a given ecosystem; the true inhabitants of an ecosystem; refering to the common nicrobiota of the body or soil microorganisms that tend to remain constant despite fluctuations in the quantity of fermentable organic matter (Atlas and Bartha 1993); describes organic matter originating within a waterbody / aquatic community (Wetzel 1983).

autotrophic index (AI)
A means of determining the trophic nature of the periphyton community; calculated by dividing the biomass (ash-free weight of organic matter) by chlorophyll *a*. High AI values indicate heterotrophic associations or poor water quality (APHA 2000).

benthos/benthic
The assemblage of organisms associated with the bottom, or the solid-liquid interface of the aquatic system. Generally applied to organisms in the substrata (Wetzel 1983).

biocriteria
(biological criteria) Narrative or numeric expressions that describe the desired biological condition of aquatic communities inhabiting particular types of waterbodies and serve as an index of aquatic community health. (USEPA 1994).

BOD
Biochemical Oxygen Demand. Oxygen required to break down organic matter and to oxidize reduced chemicals (in water or sewage) (APHA 2000).

chlorophyll *a*
A complex molecule composed of four carbon-nitrogen rings surrounding a magnesium atom; constitutes the major pigment in most algae and other photosynthetic organisms; is used as a reliable index of algal biomass (Darley 1982).

Cladophora
A common nuisance filamentous green alga (Dodds et al. 1997).

community metabolism
The relationship between gross community production and total community respiration (Odum 1963).

criteria
Elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use (USEPA 1994).

cultural enrichment
Human activities that result in increased nutrient loads to a waterbody.

designated uses
Uses defined in water quality standards for each water body or segment whether or not the use is being attained (USEPA 1994).

detritus
Unconsolidated sediments comprised of both inorganic and dead and decaying particulate organic matter inhabited by decomposer microorganisms (Wetzel 1983).

eutrophic
Abundant in nutrients and having high rates of productivity frequently resulting in oxygen depletion below the surface layer (Wetzel 1983).

eutrophication
The increase of nutrients in [waterbodies] either naturally or artificially by pollution (Goldman and Horne 1983).

existing uses
The use that has been achieved for a waterbody on or after November 28, 1975 (USEPA 1994).

flowpath
Conveys water between points in the stream system. Examples of flow paths are a stream channel, canal, storm sewer, or reservoir (http://il.water.usgs.gov/proj/feq/feqdoc/chap3_1.html).

heterotrophic
Describes organisms that need organic compounds to serve as a source of energy for growth and reproduction (Atlas and Bartha 1993).

hypolimnetic
Characteristic of the hypolimnion, the deep, cold, relatively undisturbed stratum of a lake (Wetzel 1983).

hydrologic unit codes (HUC)
An 8-digit code, determined by the U.S. EPA, that is used as a standard method for watershed identification throughout the United States.

hyporheic zone
The subsurface zone where stream water flows through short segments of its adjacent bed and banks (Winter et al. 1998).

lentic
Relatively still-water environment (Goldman and Horne 1983).

lotic
Running-water environment (Goldman and Horne 1983).

macrophyte (also known as SAV-Submerged Aquatic Vegetation)
Larger aquatic plants, as distinct from the microscopic plants, including aquatic mosses, liverworts, angiosperms, ferns, and larger algae as well as vascular plants; no precise taxonomic meaning (Goldman and Horne 1983).

macroinvertebrate
Small benthic organisms which are retained on sieves with a mesh size $\geq$2 mm (Thorp and Covich 1991).

mesotrophic (2-4)
Having a nutrient loading resulting in moderate productivity (Wetzel 1983).

morphological characteristics (2-2)
The morphological characteristics of a waterbody are the characteristics that comprise the shape of the waterbody. In stream systems, morphology usually refers to the shape of the stream channel.

NPDES
National Pollutant Discharge Elimination System. The EPA program that regulates point source discharges through the issuance of permits to discharges and enforcement of the terms and conditions of those permits.

oligotrophic (2-4)
Trophic status of a waterbody characterized by a small supply of nutrients (low nutrient release from sediments), low production of organic matter, low rates of decomposition, oxidizing hypolimnetic condition (high DO) (Wetzel 1983).

parafluvial
Sediments within the active channel, outside the wetted stream; lateral sandbars (Holmes et al. 1994).

periphyton
Associated aquatic organisms attached or clinging to stems and leaves of rooted plants or other surfaces projecting above the bottom of a water body (USEPA 1994).

primary production
Quantity of new organic matter created by photosynthesis or chemosynthesis, or stored energy which that material represents (Wetzel 1983).

probability sampling
A sampling process wherein randomness is a requisite (Hayek 1993).

production/respiration ratio
The primary production to respiration ratio is a measure of community or whole system metabolism. This measurement can be used to assess ecosystem health and determine if the system is heterotrophically or autotrophically dominated.

Q10
The estimated discharge of ten year flood (USEPA 1994).

random sampling
Generic type of probability sampling, randomness can enter at any stage of the sampling process (Hayek 1993).

RTAG (Regional Technical Assistance Group)
Group of technical experts assembled at the EPA Regional level to assist in establishing criteria for States, Tribes and nutrient ecoregions.

reference conditions
Describe the characteristics of water body segments least impaired by human activities. As such, reference conditions can be used to describe attainable biological or habitat conditions for water body segments with common watershed/catchment characteristics within defined geographical regions.

riparian
Riverside, usually referring to vegetation (riparian vegetation) (Goldman and Horne 1983).

Secchi disk
A white or black and white disk used to measure transparency of a waterbody. The Secchi disk transparency is measured as the mean depth of the point where a weighted white (or black and white) disk, 20 cm in diameter, disappears when viewed from the shaded side of a vessel, and that point where the disk reappears upon raising it after it has been lowered beyond visibility (Wetzel 1983).

secondary production
New organic material created by an organism that uses organic substrates (i.e. uses material from primary producers) (Wetzel 1983)

seston/sestonic
organic matter suspended in the water column generally comprised of phytoplankton, bacteria and fine detritus (Thorp and Covich 1991).

STORET
EPA's computerized water quality database that includes physical, chemical, and biological data measured in water bodies throughout the United States (USEPA 1994).

Stratification, stratified random sampling
Type of probability sampling where a target population is divided into relatively homogenous groups or classes (strata) prior to sampling based on factors that influence variability in that population (Hayek 1993). In stratified sampling, a heterogenous environment is divided into homogenous strata or parts. Analysis of variance can be used to identify statistically different parameter means among the sampling strata or classes. The strata are the analysis of variance treatments (Poole 1972).

TMDLs
Total maximum daily loads (TMDLs) are defined by calculating the assimilative capacity of a waterbody for a substance (e.g. total phosphorus) and identifying the sources to determine the maximum load the waterbody is capable of carrying without causing detrimental effects.

trophic state
The trophic status of a waterbody (Carlson 1977).

TSS (total suspended solids)
Particulate matter suspended in the water column.

turbidity
Cloudiness or opaqueness of a suspension. In our context, refers to the amount of suspended matter in the water column, usually measured in nephelometric turbidity units (Atlas and Bartha 1993).

TVSS (total volatile suspended solids)
Volatile particulate matter suspended in the water column.

watershed
The area of land that drains water, sediment, and dissolved materials to a common outlet at some point along a stream channel. In American usage, *watershed* is synonymous with the terms *drainage basin* and *catchment* (Dunne and Leopold 1978).