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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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CONTENTS

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CONTRIBUTORS

Sharon Buck (U.S. Environmental Protection Agency)*
Gregory Denton (Tennessee Department of Environment and Conservation)
Walter Dodds (Kansas State University)*
Jen Fisher (U.S. Environmental Protection Agency)*
David Flemer (U.S. Environmental Protection Agency)
Debra Hart (U.S. Environmental Protection Agency)*
Amanda Parker (U.S. Environmental Protection Agency)*
Stephen Porter (U.S. Geological Survey)
Sam Rector (Arizona Department of Environmental Quality)
Alan Steinman (South Florida Water Management Districts)
Jan Stevenson (Michigan State University)*
Jeff Stoner (U.S. Geological Survey)
Danielle Tillman (U.S. Environmental Protection Agency)
Sherry Wang (Tennessee Department of Environment and Conservation)
Vicki Watson (University of Montana)*
Eugene Welch (University of Washington)*

*Denotes primary authors

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This document was peer reviewed by a panel of expert scientists. The peer review charge focused on evaluating the scientific validity of the process for developing nutrient and algal criteria described in the guidance. The peer review panel comprised Drs. Elizabeth Boyer (State University of New York), Nina Caraco (Institute of Ecosystem Studies), Gary Lamberti (University of Notre Dame), Judy Meyer (University of Georgia), and Val Smith (University of Kansas). Edits and suggestions made by the peer review panel were incorporated into the final version of the guidance.

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.

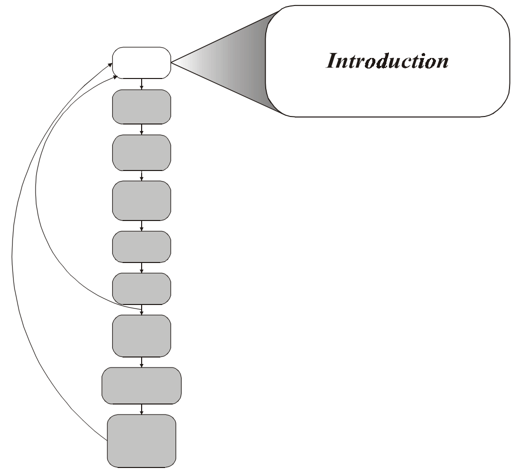
Chapter 7 presents several approaches that water quality managers can use to select numeric criteria for the rivers and streams in their State/Tribal ecoregions. The approaches that are presented include: the use of reference streams, applying predictive relationships to select nutrient concentrations 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.

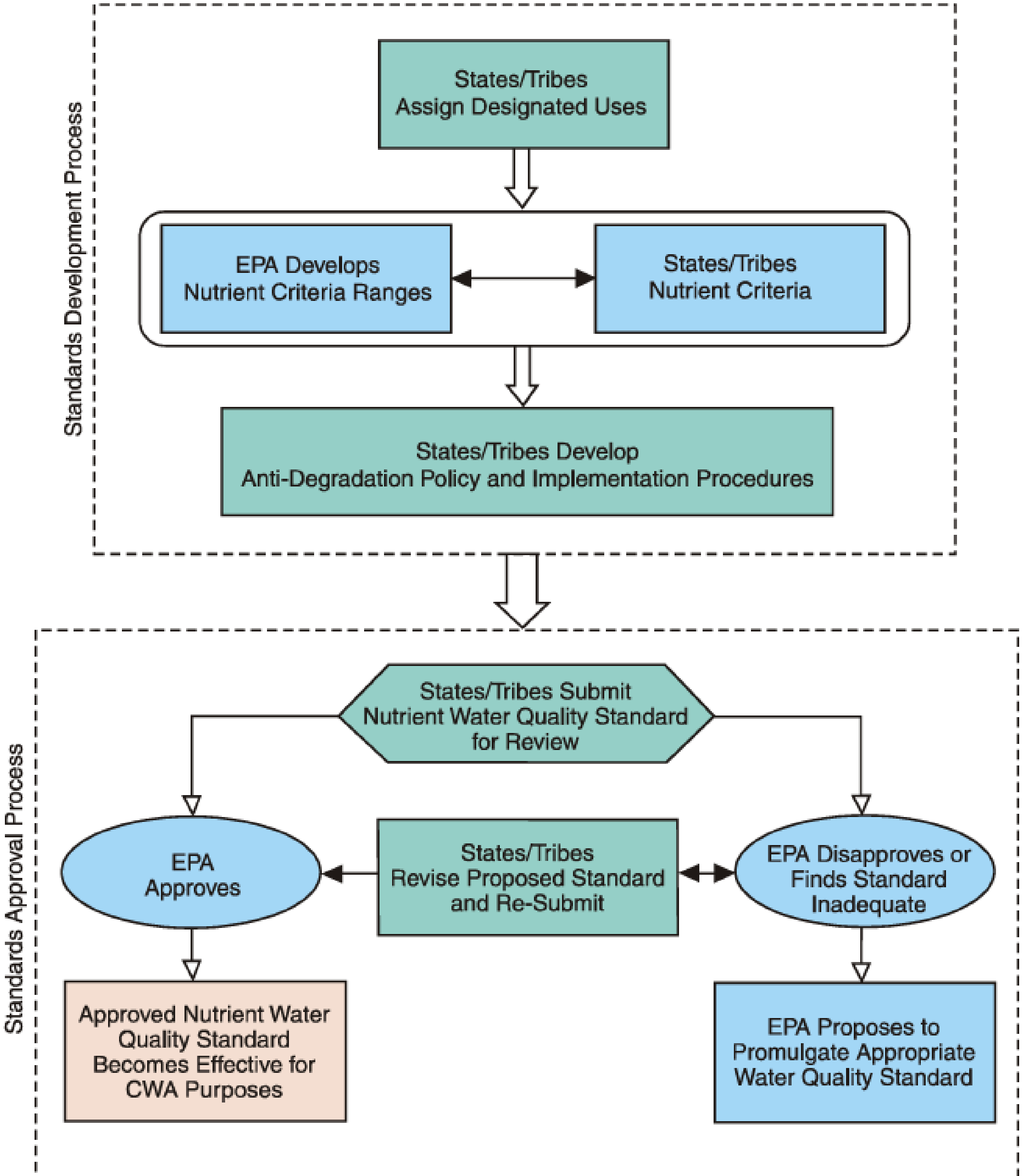


Figure 1. Developing water quality standards for nutrients.

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

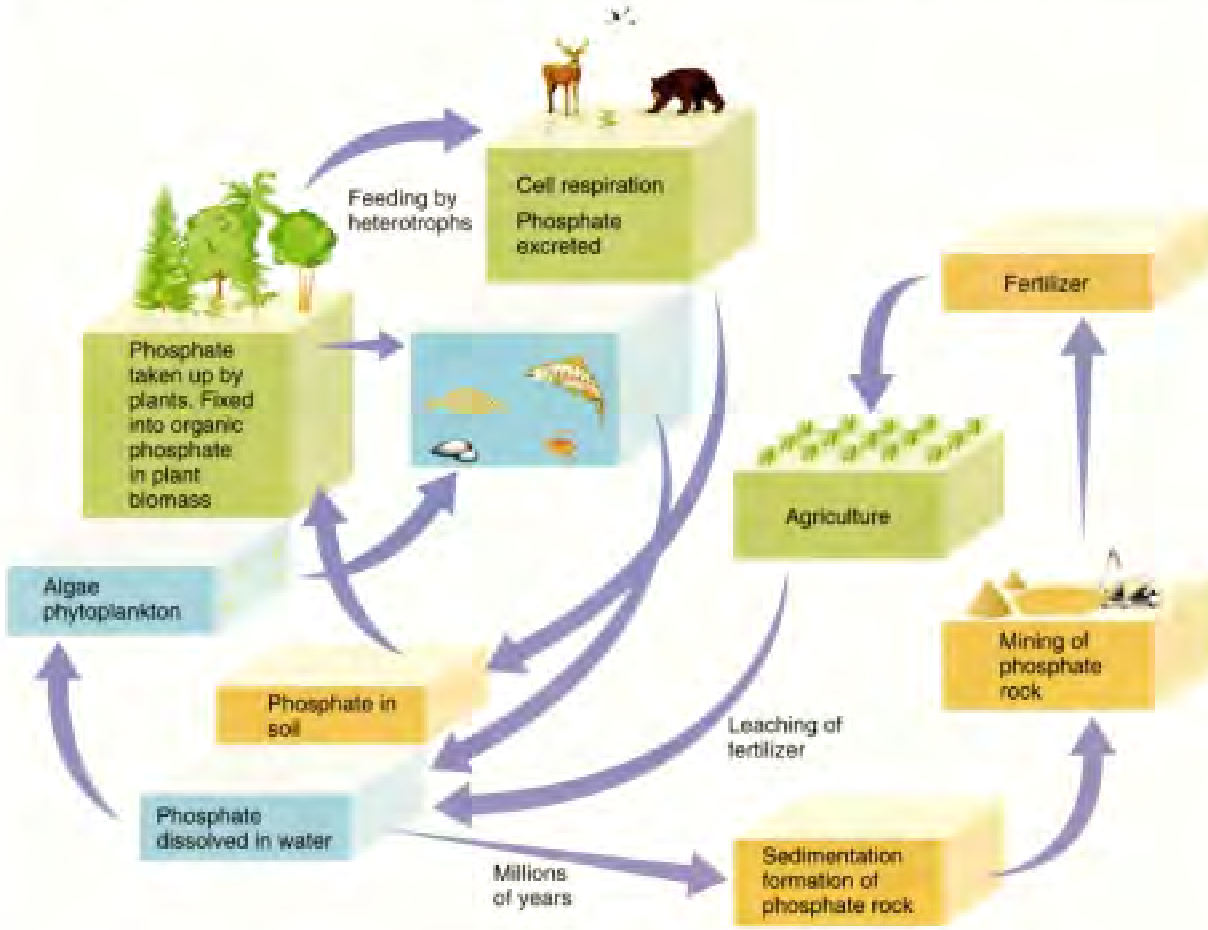


Figure 2a. The phosphorus cycle.

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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

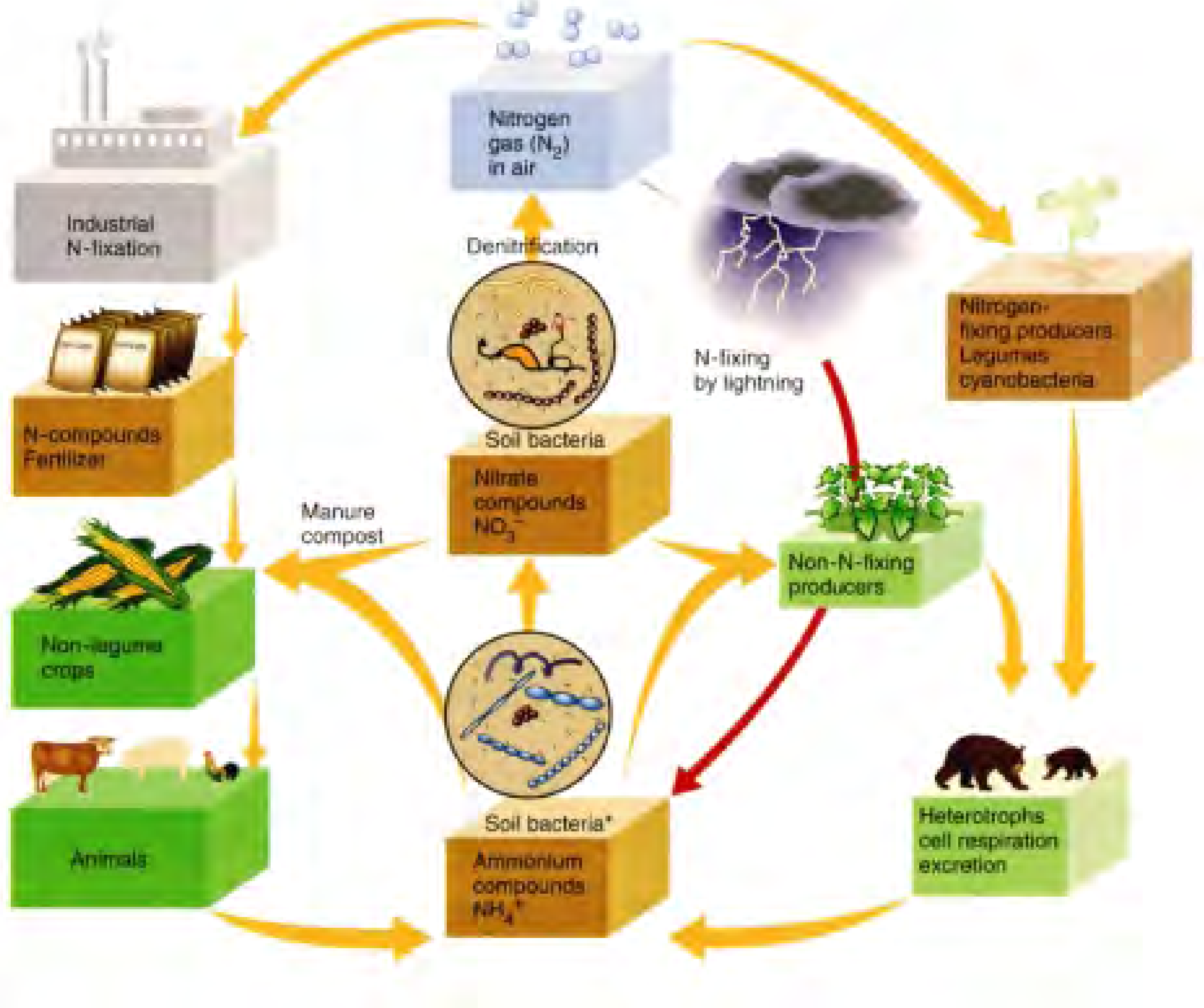

Figure 2b. The nitrogen cycle.

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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.

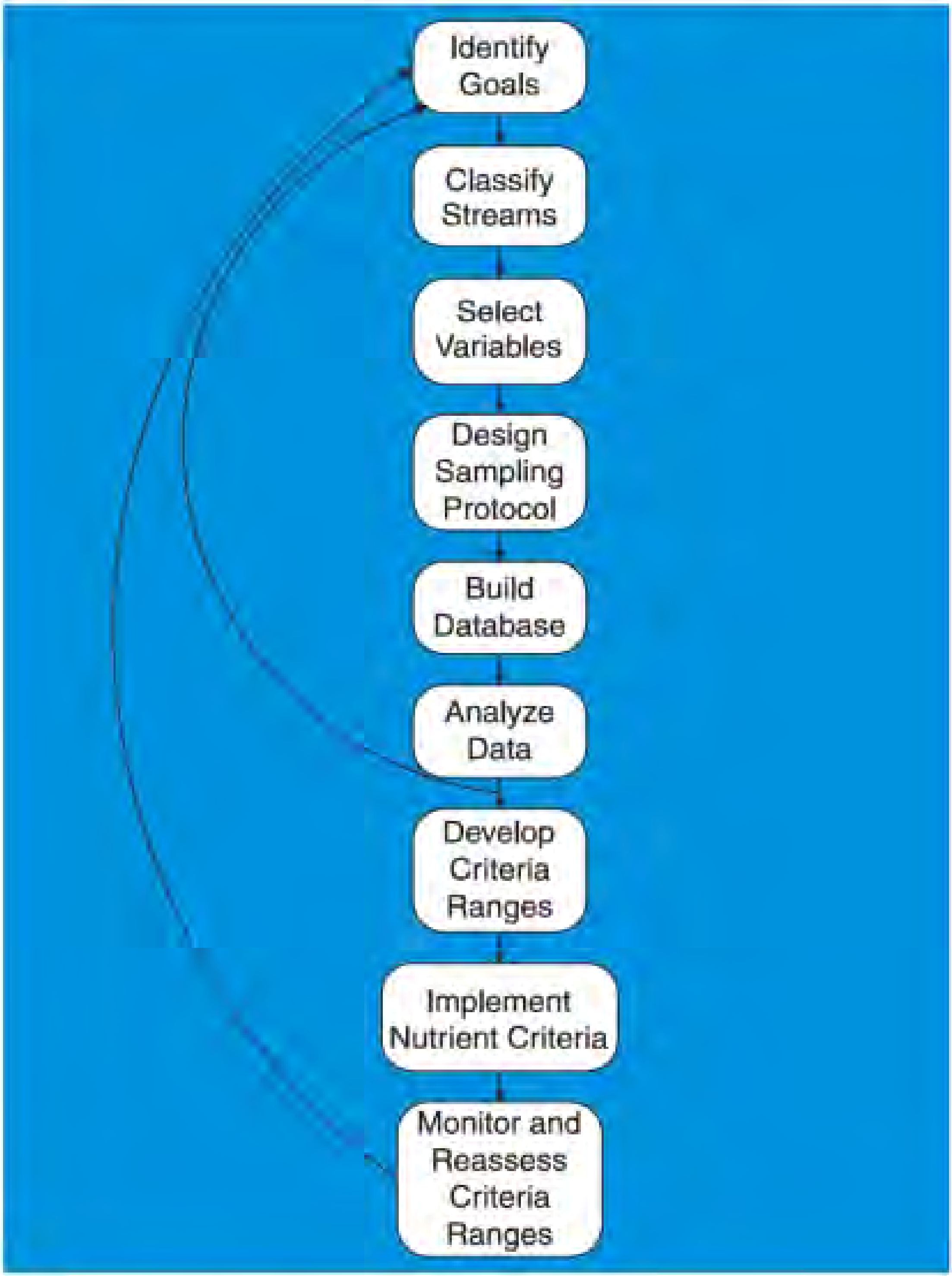


Figure 3. Criteria development flow chart.

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.

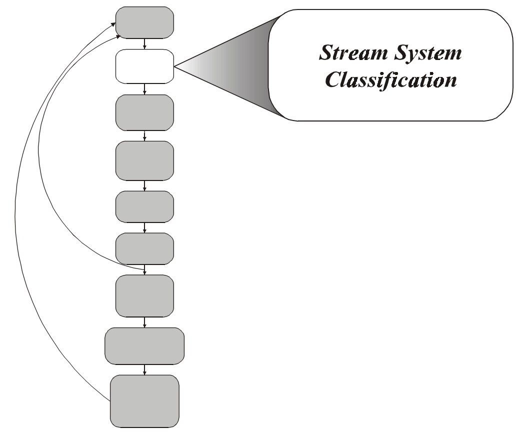
1.7 DOCUMENT STRUCTURE

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.

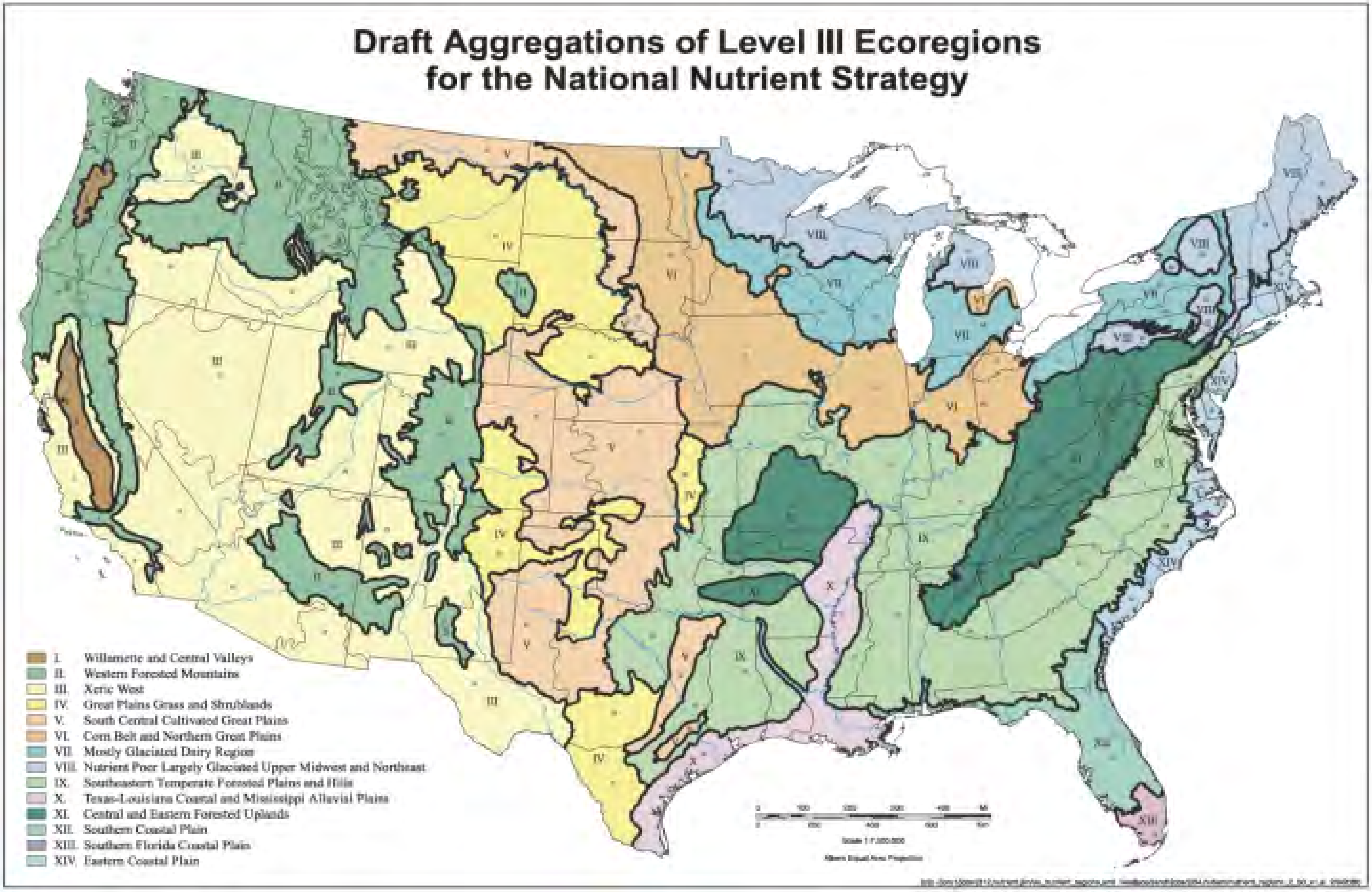


Figure 4. Fourteen nutrient ecoregions as delineated by Omernik (2000). Ecoregions were based on geology, land use, ecosystem type, and nutrient conditions.

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

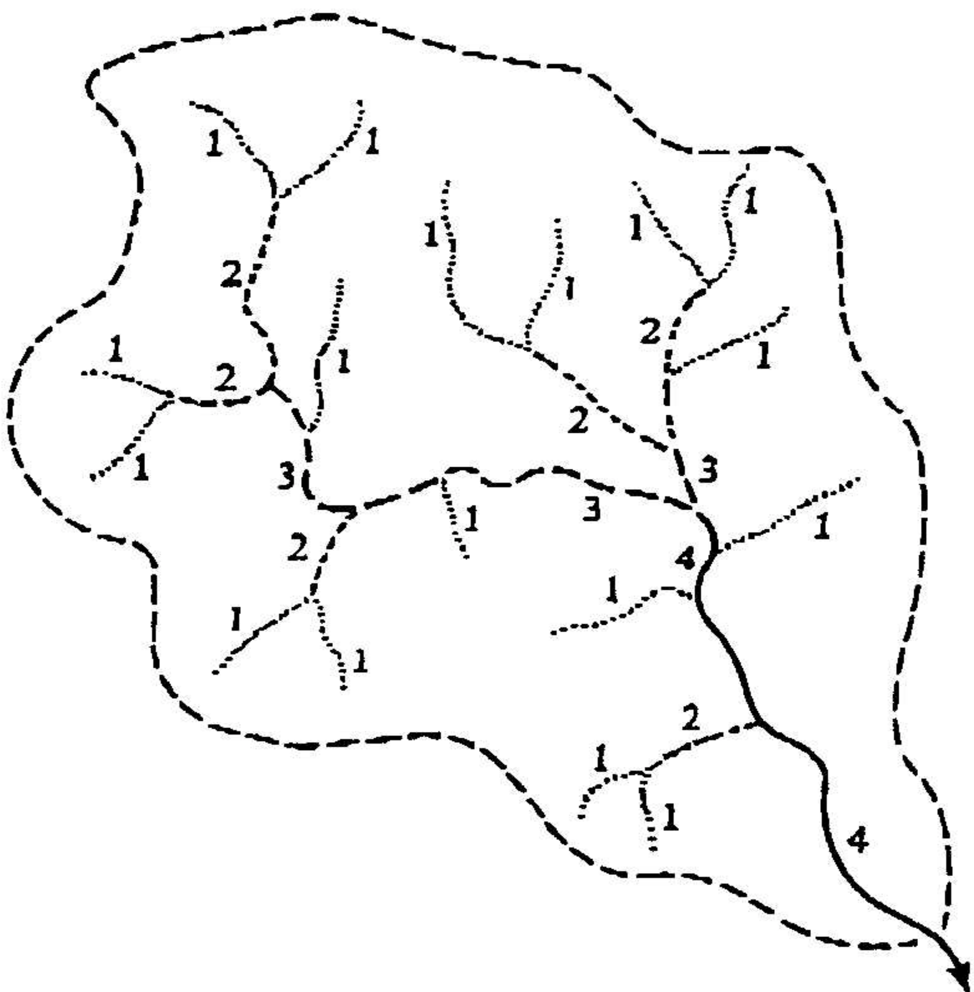


Figure 5. Stream ordering of a watershed basin network using the Strahler method. (Adapted from Strahler [1964]).

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

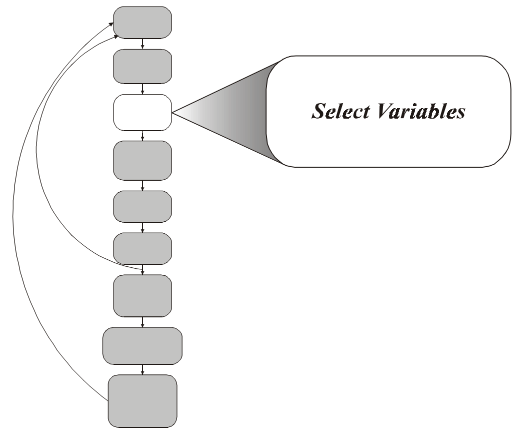
$^{+}$Data from Dodds et al. (1998); $^{++}$data from Van Nieuwenhuyse and Jones (1996); $^{+++}$data from Omernik (1977).

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.

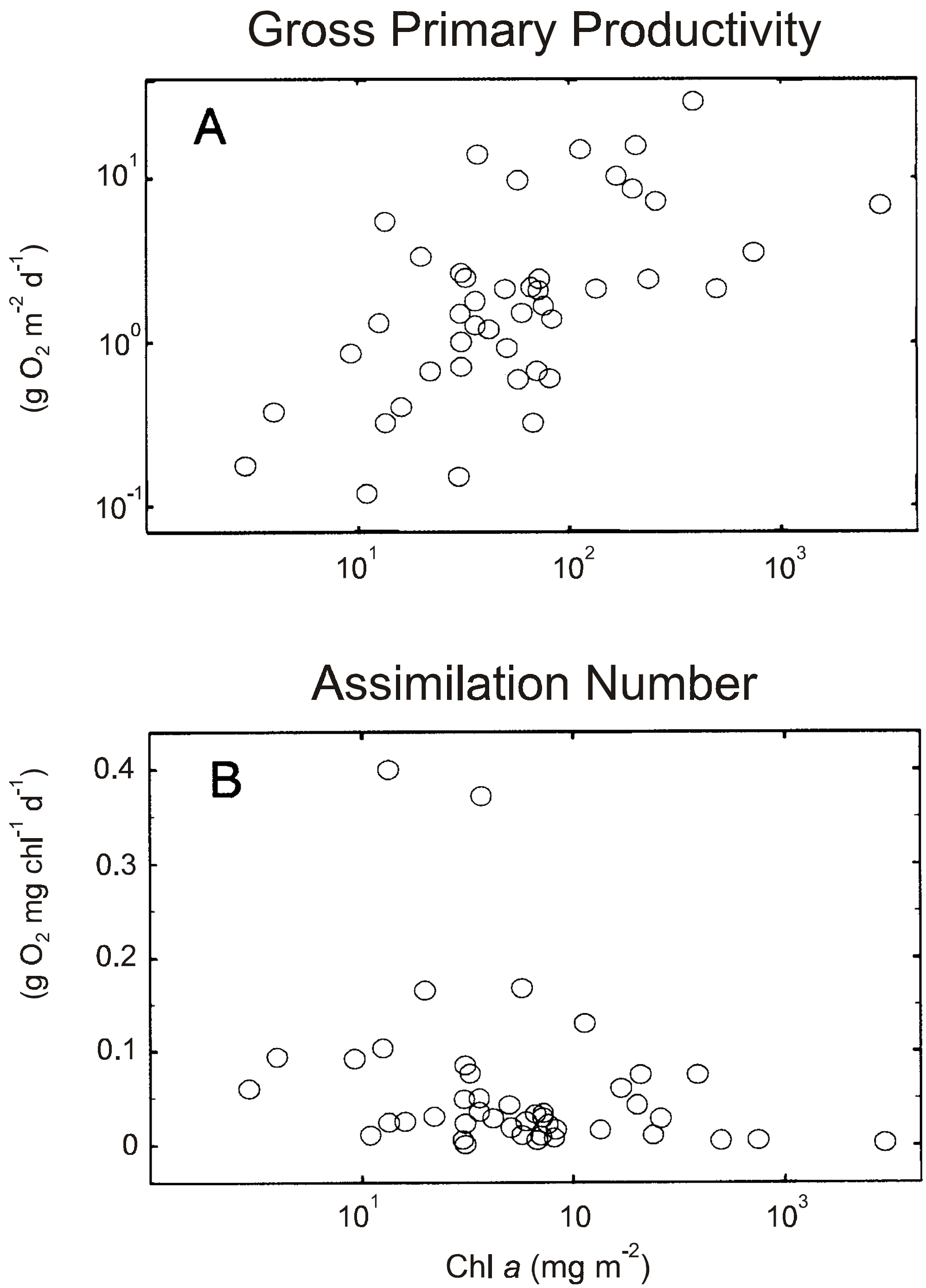


Figure 6. Integrated daily productivity related to biomass as chlorophyll *a* (data compiled by Dodds from published literature; many of the data from Bott et al. 1985).

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

<u>Conductivity</u>

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.

<u>Dissolved Organic Carbon</u>

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

<u>Temperature</u>
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

<u>Algal Biomass as Ash-Free Dry Mass</u>
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).

<u>Algal Biomass - % Cover of Bottom by Nuisance Algae</u>
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.

<u>Macrophytes</u>

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 communities 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.

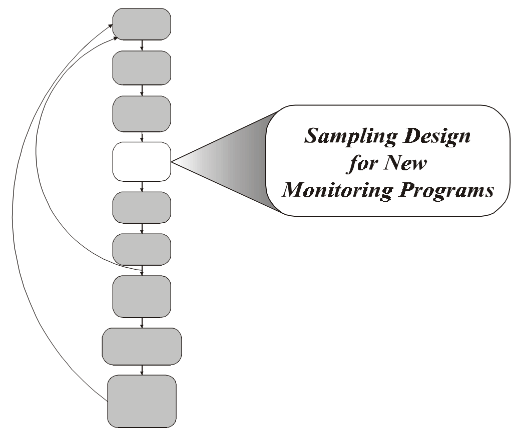
Current velocity, sediment type, and light availability to a large extent determine the plant types that occur in rivers (Hynes 1969; Goldman and Horne 1983; Chambers et al. 1999).

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 4. Sampling Design for New Monitoring Programs



4.1 INTRODUCTION

The purpose of this chapter is to provide technical guidance on designing effective sampling programs for reconnaissance. Appropriate data describing stream nutrient and algal conditions are lacking in many places. 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.

Nutrient monitoring programs are used to better define nutrient and algal relationships within stream systems. At the broadest level, monitoring data should detect:

1. Seasonal patterns in nutrient levels and their relationship to algal biomass levels;

2. The assimilation capacity of the system for nutrients: i.e., how much nutrient loading can be assimilated without causing unacceptable changes in water quality or the algal community (biomass and composition);

3. Whether nutrient concentrations are increasing, decreasing, or staying the same over time.

This Chapter provides discussion on issues to consider with regard to monitoring nutrients and their effects in stream systems. The various forms of nutrients to consider for sampling are discussed in Chapter 3. Field sampling and laboratory methods for nutrient assessment are described in Appendix B.

Monitoring programs are often poorly and inconsistently funded or are improperly designed and carried out, making it difficult to collect a sufficient number of samples over time and space to identify changes in water quality or estimate average conditions with statistical rigor. This Chapter provides a procedural approach for assessing water quality condition and identifying impairment by nutrients and algae in stream reaches. The approaches described below present sampling designs that allow one to obtain a significant amount of information with relatively minimal effort. Probabilistic and stratified random sampling begin with large-scale random monitoring designs that are reduced as nutrient and algal conditions are characterized. The tiered approach to monitoring begins with coarse screening and proceeds to more detailed monitoring protocols as impaired and high-risk systems are identified and targeted for further investigation.

Water quality variables other than the primary variables discussed in Chapter 3, e.g., DO, pH, TSS, etc., should be critically selected in a monitoring design to obtain the most cost-effective information required to assess river system nutrient and algal conditions. Sampling should be designed to answer questions such as: how, when, where and at what levels do nutrient concentration and algal biomass contribute to unacceptable water quality conditions (e.g., offensive odors, aesthetic impairment, degraded habitat for aquatic life, diurnal decreases in DO and pH increases)? These questions are interrelated, and a well-designed program that monitors the primary variables (TN, TP, chl *a*, turbidity) with other water quality variables can contribute to answering them.

4.2 SAMPLING PROTOCOL

CONSIDERATIONS FOR SAMPLING DESIGN

Developing nutrient criteria and monitoring the success of nutrient management programs involve important considerations for sampling design. Initially, the relationships between critical response variables and nutrient concentrations need to be established. Next, reference reaches should be sampled and assessed for specific classes of streams. **Nutrient concentrations and algal biomass levels in reference reaches should define the ecological state that could be attained if impaired reaches were restored.** In some streams and rivers, nutrient levels may be naturally high if bedrock, soils, or wetlands are nutrient-rich sources in the region. However, human actions can exacerbate nutrient enrichment regardless of the natural nutrient condition.

Reach/stream selection for establishing causal relationships between nutrients and algal biomass is based on the need to sample a relatively large number of streams with nutrient concentrations distributed along the entire nutrient gradient for each class of streams in a specific regional setting. Cause-response relationships can also be identified using large sample sizes and streams with low as well as medium and high nutrient concentrations. All ranges of responses should be observed along the gradient from reference condition to high levels of human disturbance. Therefore, streams should be selected based on land-use in the region so that watersheds range from minimally impaired with expected low nutrient runoff to high levels of development (e.g., agriculture, forestry, or urban) with expected high runoff.

Assessing watershed characteristics through aerial photography and the use of geographical information systems (GIS) linked to natural resource and land-use databases, can aid in identifying reference and impaired streams. Some examples of watershed characteristics which can be evaluated using GIS and aerial photography include land-use, land-cover (including riparian vegetation), soils, bedrock, hydrography, infrastructure (e.g., roads, public sewerage systems, private septic systems), and climate. Watersheds with little or no development that receive minimal anthropogenic inputs could potentially contain streams that would serve as reference sites (see section below). Watersheds with a high percentage of their area occupied by nutrient-rich soils, heavily fertilized agricultural land, and extensive unsewered development in coarse soils are likely to contain streams receiving high nutrient loads that could potentially be considered ‘at risk’ for developing nutrient and algal problems. The USDA agricultural census provides information on agricultural land use (crops, livestock, irrigation, chemicals used) at the national, state, and county levels. Data are available on their website at: http://www.nass.usda.gov/census/.

Once the watershed level has been considered, a more stream-specific investigation can be initiated to better evaluate nutrient and algal conditions. Rivers and streams need adequate light and nutrients to develop and maintain high levels of algal biomass. In addition, attached algae (periphyton) require coarse substrata (cobbles, boulders) and a flow regime that provides sufficient periods between scouring floods (at least one month) to accumulate high levels of biomass. The condition of the riparian zone needs to be considered. Riparian buffer zones may mediate the effects of nonpoint sources of nutrients and turbidity and, depending on the slope of the system, may reduce the velocity of overland runoff to a stream. Riparian wetlands may serve as both sources and sinks for nutrients varying with wetland type, seasonal flows, and degree of disturbance. The presence or absence of streamside trees can affect light limitation in a stream. Light is unlikely to limit algal growth where streamside trees have been removed or the stream is wide, shallow and clear enough to permit sufficient light to reach much of the bottom. Shaded streams may have high nutrient concentrations with no correlative response in algal growth, though the nutrient load may stimulate algal growth further downstream. The relative risk to develop nutrient and algal problems could be assessed by noting how many of the above factors that permit higher algal levels and/or nutrient concentrations are common to a stream or reach.

WHERE TO SAMPLE

Nutrient inputs can occur at a myriad of points along a river system resulting in highly variable concentrations of nutrients throughout the system. System variability and multiple nutrient input points require numerous sampling sites for assessing the nutrient condition of a river system. Monitoring stations for nutrients in streams and rivers should be located upstream and downstream from major sources of nutrients or diluting waters (e.g., discharges, development, tributaries, areas of major groundwater inputs) to quantify sources and loads.

WHEN TO SAMPLE

Nutrient and algal problems are frequently seasonal in streams and rivers, so sampling periods can be targeted to the seasonal periods associated with nuisance problems. Nonpoint sources may cause increased nutrient concentrations and turbidity or nuisance algal blooms following periods of high runoff during spring and fall, while point sources of nutrient pollutants may cause low-flow plankton blooms and/or increased nutrient concentrations in pools of streams and in rivers during summer. In most state monitoring programs, sampling is only conducted once during the season when greatest impacts are expected. If only a one-time sampling is possible, then sampling between two to four (2-4) weeks after a storm or high flow event has disturbed algal assemblages (Stevenson and Bahls 1999) is recommended. Two to four weeks will allow sufficient time for algal biomass recovery in streams where algal biomass predominantly consists of diatoms or micro-algae. Alternatively, sampling should be conducted during the growing season at the mean time after flooding for the system of interest. In streams where macro-algae or macrophytes comprise the dominate photosynthetic biomass, recovery of photosynthetic biomass may take one or more growing seasons following a major high-flow event. However, if a high-flow event does not move anchoring substrata, the flow event will only have a nominal effect on photosynthetic biomass. High flow events late in the growing season when algal and macrophyte filaments and fronds are more prone to slough, may cause a reduction in the photosynthetic biomass. A one-time sampling approach may be adequate for indicators of nutrient status, designated use, and biotic integrity. However, criteria and biological or ecological indicator development (see Assessing Algal Biomass below) may require more frequent sampling to observe nutrient conditions that relate to peak algal biomass (Biggs 1996; Stevenson 1996; Stevenson 1997b).

Nutrient concentrations vary with climate-driven changes in flow. Algal blooms, both benthic and planktonic, can develop rapidly and then may dissipate as nutrient supplies are depleted or flow increases. Thus sampling through the season of potential blooms may be necessary to observe peak algal biomass and to characterize the nutrient conditions that caused the bloom. Sampling through the season of potential problems is important for developing cause-response relationships (with which biological and ecological indicators can be developed) and for characterizing reference conditions. Keep in mind that there is a time-lag between nutrient enrichment and algal response. Therefore to characterize algal response to a specific enrichment event, nutrient sampling should be conducted prior to algal sampling. Samples for nutrients should also be collected during the season of lowest algal levels (at least 3 samplings spread over the period) to determine current background levels of algal biomass; avoid the problem of algal uptake attenuating nutrient concentrations, and help provide an estimate of maximum nutrient concentration. Many nutrient monitoring programs are based on quarterly sampling. However, quarterly samples are usually inadequate to detect long-term trends due to year-to-year variation in the window of high flows, the period of high nutrient uptake and algal growth, and the period of algal sloughing at the end of the growing season.

If few nutrient and algal data exist, then multi-year surveys on a twice monthly or monthly basis may be necessary to determine if nuisance algal problems occur. Frequent sampling is necessary because algal blooms may develop and dissipate rapidly with residual adverse effects, such as fish kills and impaired aquatic habitat. Multi-year sampling is necessary because unusually large annual variability can occur annually in the intensity of nutrient/algal problems, due to timing of weather (primarily scouring storm events or persistent low flow events with long residence time) and seasonality of algal blooms.

Ideally, water quality monitoring programs produce long-term datasets compiled over multiple years, to capture the natural, seasonal and year-to-year variations in waterbody constituent concentrations (e.g., Dodds et al. 1997; Tate 1990). Multiple-year datasets can be analyzed with statistical rigor to identify the effects of seasonality and unusual flow years (Miltner and Rankin 1998). Once the pattern of natural variation has been described, the data can be analyzed to determine the water quality conditions that degrade the ecological state of the waterbody or effect downstream receiving waters. Long-term data sets have also been extremely important in determining the cost-effectiveness of management techniques for lakes and reservoirs (Cooke et al. 1993). The same should be true for streams and rivers, if not more so (due to greater constituent variability), although management of nutrients to improve quality in streams and rivers has not been as well documented.

In spite of the documented value of long-term data sets, there is a tendency even in lake/reservoir management to intensively study a waterbody for one year before and one year after treatment. A more cost-effective approach would be to measure only the most essential indices, but to double or triple the monitoring period. Two or more years of data are needed to identify the effects of years with extreme climatic or flow conditions. Low periphyton biomass has often been observed during high-flow summers as well as the reverse, i.e., high biomass-low flow. The cause for that is not entirely clear; high flows may reduce biomass through scouring and/or dilute inputs of ground water nutrients. Whatever the cause, the effect will be “averaged out” enough to discern the overall effect of treatment (e.g., nutrient reduction or diversion) if several years of data are available to minimize the effect of the unusual flow year(s). At the very minimum, two years of data before and two after implementing nutrient management, but preferably three or more each, are recommended to evaluate treatment cost-effectiveness with some degree of certainty. If funds are limited, restricting sampling frequency and/or numbers of constituents analyzed should be considered to preserve a longer-term data set. This will allow for effectiveness of management approaches to be assessed against the high annual variability that is common in most streams. High hydrological variation in a stream from year to year, requires more years of sampling before and after mitigation procedures.

Characterizing Precision of Estimates

Estimates of dose-response relationships, nutrient and biological conditions in reference reaches, and stream conditions of a region are based on sampling. Therefore, precision and accuracy must be assessed. Determining precision of measurements for one-time assessments from single samples in a reach is often necessary. The variation associated with one-time assessments from single samples in a reach can often be determined by re-sampling a specific number of reaches during the survey. Measurement variation among replicate samples can then be used to establish the expected variation for one-time assessment of single samples. Re-sampling does not establish the precision of the assessment process, but rather identifies the precision of an individual measurement. Re-sampling frequency is often conducted for one stream reach in every block of ten reaches. However, investigators should adhere to the objectives of re-sampling (often considered an essential element of QA/QC) to establish an assessment of the variation in a one-time/sample assessment. The larger the sample size the better (smaller) will be the estimate of that variation. Often, more than one in ten samples need to be replicated in monitoring programs to provide a reliable estimate of measurement precision.

APPROACHES TO SAMPLING DESIGN

The following sections discuss two different approaches to sampling design, probabilistic and goal-oriented. Both approaches have advantages and disadvantages that under different circumstances warrant the choice of one approach over the other (Table 3). The decision as to the best approach for sample design in a new monitoring program must be made by the water quality resource manager or management team after carefully considering different approaches.

Probabilistic Sampling

Probability sampling, where randomness is required, can be used to determine the variability of nutrient and algae levels in streams and rivers across a state or a region. Random sampling is a generic type of probability sampling where randomness can enter at any stage of the sampling process. Probabilistic sampling – a sampling process wherein randomness is a requisite (Hayek 1994) – can be used to characterize the status of nutrient conditions and biotic integrity in a region’s streams and rivers. Probabilistic designs are often modified by stratification (such as classification [Chapter 2]), by deleting "redundant" reaches, or by adding important sites. Stratification or stratified random sampling is a 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 1994). Analysis of variance can be used to identify statistically different parameter means among the sampling

Table 3. Comparison of probabilistic and goal-oriented sampling designs.

Probabilistic Sampling	Goal-oriented Sampling
• random selection of streams from entire population within a region • requires no prior knowledge of streams within the sample population • may require more resources (time and money) to randomly sample stream classes because more streams may be sampled • nutrient condition characterization for a class of streams is more statistically robust • potentially best for regional characterization of stream classes, especially if water quality conditions are not known	• targeted selection of streams based on problematic (reaches known to have nutrient/algal problems) and reference reaches • requires prior knowledge of streams within the sample population • utilizes fewer resources because only targeted streams are sampled • nutrient condition characterization for a class of streams is less statistically robust, though characterization of a targeted stream or reach may be statistically robust • potentially best for site-specific and watershed-specific criteria development when water quality conditions for the reach of interest are known • selection of sites that represent a range of nutrient conditions will facilitate establishment of nutrient-algal relationships for the systems of interest

strata or classes. The strata are then used as the analysis of variance treatments (Poole 1972). Goal-oriented sampling as described in the tiered approach in this Chapter, is not as easily analyzed by rigorous statistical analyses. Goal-oriented monitoring may be better suited to statistical analyses using basic descriptive statistics and correlational analyses.

Streams are selected for probabilistic sampling by random selection of a sample of streams from the entire population of streams within a region. Thus, all stream reaches within a region must be identified to establish the statistical population of streams; then a sample of all possible streams is selected from that population. The results of collecting and assessing water quality and biotic responses with a probabilistic sample is, presumably, an unbiased estimate of the descriptive statistics (e.g., means, variances, modes, and quartiles) of all streams in a region. Probabilistic sampling designs are commonly modified by stratifying by stream size and stream classes. Otherwise, sample statistics would be most characteristic of the numerous small streams of the dominant stream types in a region.

Many state 305b and watershed monitoring programs utilize modified probabilistic sampling designs. Stratification in many of these programs is based on identifying all stream reaches in a region (or watershed) and then selecting an "appropriate" sample of reaches from the defined population. The sample population is often modified by deleting stream reaches that are too close to other reaches to be different, thereby reducing redundant collection efforts. The selected sample of streams may also be modified by adding sites that are near known sources of impact. Estimates of ecological conditions from these kinds of modified probabilistic sampling designs can be used to characterize the nutrient status, and over time, to distinguish trends in stream nutrient condition within a region. Estimates of regional conditions are best when sites near known sources of impact are removed from the analysis and later compared to the distribution of regional nutrient conditions.

Goal-Oriented Sampling

A goal-oriented approach to sampling design may be more appropriate when resources are limited. The tiered approach described here focuses the greatest efforts on identifying and characterizing rivers and streams likely to have nutrient problems, and on relatively undisturbed streams, often called reference streams or reaches, that can serve as regional or sub-regional examples of natural biological integrity. Choosing sampling stations that best allow comparison of nutrient concentrations at reference stream or river sites of known condition can conserve financial resources. Goal-oriented sampling also includes some elements of randomness. However, the identification of systems with nutrient problems and reference conditions eliminates the need for selecting a random sample of the population for monitoring.

Goal-oriented sampling assumes some knowledge of the systems sampled. Systems with evidence of impairment are compared to reference systems that are similar in their physical structure. Sites chosen to represent a range of nutrient conditions will facilitate development of nutrient concentration-algal biomass relationships. Goal-oriented sampling requires that the reaches be characterized according to assessed nutrient and algal levels. Comparison of the monitoring data to data collected from reference stream reaches will allow characterization of the sampled streams. Reaches identified as ‘at risk’ should be evaluated through a sampling program to characterize the degree of impairment. An impaired reach is simply a reach of any length where nutrient concentrations exceed acceptable levels, or algae interfere with beneficial uses. Once characterized, the reaches should be placed in one of the following categories:

1. Impaired reaches – reaches in which nutrients or algal biomass levels interfere with designated uses;

2. High-risk reaches – reaches where nutrient concentrations are high but do not significantly impair designated uses. In high-risk streams impairment is prevented by one or a few factors that could be changed by human actions, though water quality characteristics (e.g., DO, turbidity) are already marginal;

3. Low-risk reaches – reaches where many factors contain nutrient concentrations and algal biomass levels are below problem levels and/or no development is contemplated that would change these conditions.

4. Reference reaches – reaches where nutrient concentrations and algal biomass levels most closely represent the pristine or minimally impaired condition.

Once stream reaches have been classified based on their physical structure (see Chapter 2) and placed into the above categories, specific reaches need to be selected for monitoring. At this point, randomness is introduced; stream reaches should be randomly selected within each class and risk category for monitoring.

Monitoring efforts are often prioritized to best utilize limited resources. Impaired and high-risk streams should be monitored more intensively than low-risk streams. Impaired streams should be monitored to evaluate, implement, and assess management activities to reduce algal biomass and improve water quality. High-risk streams should be monitored to assure that no further degradation takes place. Low-risk streams can be monitored less frequently, but should be monitored frequently enough to identify any increase in nutrients or algae, and/or change of water quality. Reference reaches should be monitored frequently enough to make robust comparisons with impaired and high-risk stream reaches. In addition, monitoring of changes in the watershed can help identify areas where changes are likely to result in degradation of nutrient condition. Human activities within a watershed that can increase the risk of nutrient and/or algal problems include 1) stabilization of flows (reduces scour); 2) reduction of flows (increases light, reduces dilution of nutrients); 3) removal of streamside vegetation (increases light, may decrease depth of stream; and increases the flux of nutrients from the stream hillslope due to reduced uptake from plant roots); 4) discharge of nutrient rich waste water; 5) construction of unsewered residential development (especially in thin coarse soils); 6) over fertilization of agricultural land; 7) development that increases the percent of impervious surface in the watershed; and hence nutrient runoff; and 8) discharge of toxins or release of exotic species that reduce grazer populations.

IDENTIFYING AND CHARACTERIZING REFERENCE STREAM REACHES

Potential reference streams should be characterized to allow for the identification of appropriate reference streams and reference stream reaches. Classification of streams, as discussed in Chapter 2, will allow appropriate reference reaches to be identified for specific regions and stream types. Stream classification should be supplemented with information on return frequency of flows. Reference streams or reaches may not be available for all stream classes. In this case, data from systems that are as close as possible to the assumed unimpaired state of rivers and streams in that class should be sought from States or Tribes within the same nutrient ecoregion.

The identification of reference *reaches* as opposed to reference *streams* is an important distinction (see Chapter 7, Section 7.2). Identification of impaired and reference streams would be relatively simple if an entire stream had all the same physical characteristics and risk factors. However, only one specific portion of a stream length, a *reach*, may have all the characteristics necessary to produce algal problems. It may not be possible to find an entire stream that has little or no impacts anywhere in its watershed. Therefore, stream *reaches* should be targeted, but their watersheds should also be kept in mind. The stream bed, banks, and riparian zone of a reference reach should be in a fairly natural state, and its watershed as undeveloped as possible. States/Tribes should endeavor to protect such reference reaches from future development.

Streams for reference-reach sampling should be selected based on low levels of human alteration in their watersheds and aquatic habitat. Selecting reference reaches usually involves assessment of land-use within watersheds, and visits to streams to ground-truth expected land-use and check for unsuspected impacts. Sometimes ecological impairment that was not apparent from land-use and local habitat conditions may be identified. Again, sufficient sample size is important to characterize the range of conditions that can be expected in the least impacted systems of the region (see TN case study in Appendix A).

Reference reaches should be identified for each nutrient ecoregion in the State or Tribal lands and then characterized with respect to nutrient concentrations, algal biomass levels, algal community composition and associated environmental conditions including turbidity, light, and substrata as well as factors that are affected by algae, such as DO and pH. For each ecoregion in a state, a minimum of three low impact reference systems should be identified for each stream class. Highest priority should be given to identifying reference streams for those stream types considered to be at the greatest risk from nutrients and algae. Reference stream reaches are often less accessible than reaches adversely affected by nutrient and algal impairments. However, sampling need not be as frequent in reference reaches, except to validate models of algal response to nutrient loads for such reaches.

Continuation of Less Intensive Monitoring of High-Risk Reaches

The continuation of monitoring of high-risk reaches should focus on factors likely to increase nutrient concentrations or limit algal growth and on any actions that might alter those factors. For example, if light is limiting, it may be most appropriate to evaluate the potential impact of the removal of streamside trees or of the manipulation of water levels which may kill streamside trees. Stabilization of flows results in the decline of flood-dependent vegetation. Increased grazing levels can reduce streamside trees degrade banks, altering the depth and width of the stream. State/Tribal water quality agencies should encourage adoption of local riparian protection plans where light is limiting to minimize nutrient-caused water quality problems.

If scouring flows limit algal accrual and significantly dilute nutrient loading, a closer evaluation of plans that could manipulate flows (by diversion, damming or altering management at existing structures) is warranted. State/Tribal water quality agencies should inform agencies that regulate water development of the potential impacts of flow manipulation.

Development plans in the watershed should be evaluated where nutrients are limiting (see Defining the Limiting Nutrient, Section 6.2). Changes in point sources can be monitored through the NPDES permit program. Changes in nonpoint sources can be evaluated through the identification and tracking of wetland loss and/or degradation, increased residential development, increased tree harvesting, and shifts to more intensive agriculture with greater fertilizer use or increases in livestock numbers. Local planning agencies should be informed of the risk of increased nutrient loading and encouraged to guide development accordingly. Nutrient levels often gradually increase due to many growing nonpoint sources. Hence, in-stream nutrient monitoring is warranted in nutrient-limited, high-risk reaches if sufficient resources remain after meeting the needs of impaired reaches. Seasonal nutrient levels should be more stable in streams with low algal biomass than in streams with high algal biomass because nutrient concentrations would not be depleted in such streams. Sampling during growing season baseflow and nongrowing season baseflow should provide a limited, yet useful, assessment of trends in nutrient levels from year-to-year.

Whenever development plans appear likely to alter factors that were limiting algae growth in a high-risk reach, instream monitoring should be initiated at a level similar to that described for impaired reaches in order to enhance the understanding of baseline conditions.

OTHER CONSIDERATIONS FOR MONITORING NUTRIENTS

Assimilative Capacity

The assimilative capacity of a stream for nutrients depends on its physical and biological nature. Assimilative capacity is the load of nutrients entering a river system at which nutrient and algal biomass levels remain low enough such that excessive diurnal fluctuations of DO concentrations and pH levels will not occur, recreation and aesthetics will not be negatively impacted, irrigation ditches will not be clogged with algae, and biotic criteria will be consistently met. Such nutrient loads are difficult to predict because nutrients are stored in many forms and released under a variety of conditions, and because the levels of nutrients and algae causing impaired conditions may vary from system to system.

The simplest model applied has been to apply an exponential decline in instream nutrient concentrations below point sources and tributaries, with the rate of decline derived from monitored data. This approach does not quantify mechanisms (such as sedimentation, uptake, dilution by groundwater and denitrification), that can lead to nutrient losses. Such an approach was applied on the Clark Fork River (Dodds et al. 1997) to model the influence of lowered inputs from point sources on instream nutrient concentrations.

Nutrient Load Attenuation

A given nutrient load may produce a few kilometers (km) containing unacceptable algal biomass followed by a section of river containing acceptable levels because a river's load is attenuated by retention in algae and sediment. The total length of river containing unacceptable algae biomass levels may change from year-to-year due to changing nutrient loads or changes in other factors (e.g., flow, dilution) that may limit algae growth (see Section 6.2). This phenomenon was illustrated following nutrient control in the Bow River, Alberta, where TDP remained high (25 µg/L) for several km downstream from the treated wastewater source. High TDP in the portions of the stream closest to the point source release resulted in no change in algal biomass, while algae decreased farther downstream as TDP decreased (see Bow River case study, Appendix A). The length of river containing unacceptable algal biomass levels may be hypothetically estimated by the following equation described in Welch et al. (1989).

$$D_c = Q*r*(SRP_i - SRP_c)/[(P/\text{chl } a \text{ day})*B_n*T*W*10^3 \text{ m/km}])$$

where SRP is in µg/L (mg/m^3) producing the threshold nuisance biomass (150 mg chl/m^2) in the growth period (nominally ~ 1-4 mg/m^3 in channel experiments [Walton et al. 1995]); Q is the daily flow in m^3/day; r accounts for the recycle (~ 1.5, after Newbold et al. 1981); SRP_i is the influent concentration (ambient river and groundwater in mg/m^3) to the segment; SRP_c is the critical concentration, above which nuisance algal growth occurs; P/chl *a*-day is the average uptake by periphyton with nominal value of 0.2; B_n is the nuisance threshold biomass of 150 mg chl *a*/m^2; T is the factor for trophic (consumer) retention (~ 1.2 representing a 20% conversion); and W is average stream width in meters.

This equation is simply the ratio of SRP mass available for uptake in excess of the critical level and the expected demand for SRP by periphyton in an enriched stream reach in which the threshold nuisance biomass is attained. The basis of the formulation is that periphytic biomass will not be reduced unless SRP is less than the critical concentration (SRP_c) during low-flow, maximum growth conditions, which has been shown to be quite low in channel experiments (Walton et al. 1995). Low values for the critical P concentration were supported by the Bow River case study (see Appendix A). The length of river with unacceptable algal biomass levels increases as the criterion decreases. The important recycle rate in the equation is a nominal value taken from uptake studies in a natural stream and could be highly variable. More definite predictions of limiting nutrient content and algal biomass changes downstream from a point source requires a dynamic model for algal biomass, such as:

$$dB/dt = (u * L * Bi) - (S + G)$$

where u = nutrient uptake rate in 1/day, L = dimensionless light factor, Bi = periphyton biomass from previous time step in mg chl/m^2, S = sloughing loss in mg chl/m^2-day and G = grazing loss in mg chl/m^2-day (after Elswick 1998).

Estimating nutrient loads to a stream is at least as complex as a detailed nutrient source study for a lake and requires the tracking of nutrient sources upstream and upgradient. In some cases, loading estimates of stream and river systems may be back calculated from the loading estimate for the receiving waterbody. That is, the partition of the nutrient load to a receiving waterbody (lake or estuary) identified as belonging to a particular stream may be used as an estimate of the total load for that stream or reach. Loading is often estimated using a calibrated model that predicts nutrient loads from hydrologic inputs or other parameters if nutrient data are inadequate to calculate load.

The USGS has developed a set of spatially referenced regression models for evaluating nutrient loading in a watershed. The modeling approach is referred to as SPARROW (SPAtially Referenced Regressions On Watershed attributes), a statistical modeling approach that retains spatial referencing for illustrating predictions, and for relating upstream nutrient sources to downstream nutrient loads (Preston and Brakebill 1999) (See Appendix C). Stream-load estimates at gaged monitoring sites are generated from stream-discharge and water quality data by utilizing a log-linear regression model called ESTIMATOR. The ESTIMATOR model estimates daily concentration values based on flow, season, and temporal trend terms (Preston and Brakebill 1999) (see Appendix C).

Better Assessment Science Integrating Point and Nonpoint Sources, or BASINS, is a tool developed by EPA to facilitate water quality analysis on a watershed level for specific waterbodies or stream segments. BASINS was designed to integrate national water quality data, modeling capabilities, and (GIS) so that regional, State, local and Tribal agencies can easily address the effects of both point and nonpoint source pollution and perform sophisticated environmental assessments (http://www.epa.gov/ost/BASINS/).

Models should be used with caution. Models can be used incorrectly and, therefore, can be less accurate than loads calculated from data. Regardless of the method used for calculating loads, subsequent changes in the watershed may alter the relationship between hydrologic and nutrient inputs requiring loads to be re-calculated to reflect those changes.

Assessing Algal Biomass

This section focuses on assessing attached algal biomass and how to obtain a meaningful, representative algal biomass sample. Sampling strategies will vary with objectives of programs. Algal sample collection techniques for streams and laboratory methods for the analysis of chlorophyll, AFDM, and other measures of biomass are discussed in Appendix B.

If the goal of sampling is to develop a relationship between nutrients and algal problems for the rivers of a region or to assess status and trends in nutrient-related problem areas of a region (i.e. probabilistic sampling), then one representative estimate of algal assemblage characteristics is all that can be used in an analysis. In most cases, the desired estimate is a mean algal biomass measure for a reach that can be obtained with composite sampling (explained below). However, spatial extent and temporal duration of blooms or nuisance growths may also be important parameters to characterize. More than one sample (or estimate) from a site would result in pseudoreplication (Hurlburt 1984) and would be unacceptable for data analyses which require independent observations of conditions (biotic and nutrient) at each site.

Variability in attached algal biomass estimates due to spatial variability can be reduced by collecting composite samples and by sampling in targeted habitats where algal biomass is relatively uniform (e.g., riffles). Composite sampling calls for combining subsamples from many substrata into a single sample, thus incorporating spatial variability into the one sample. The targeted habitat is usually defined as the habitat in which nuisance problems are greatest, typically the riffles during higher flow seasons and pools during low-flow seasons. Variability in algal biomass assessments should decrease with increasing numbers of riffles and area of stream assessed. Therefore, composite samples should be collected over the entire study reach.

Large scale assessments are particularly important for patchy filamentous algae, which may be best assessed using rapid periphyton surveys (in-stream, visual assessments of periphyton biomass; see Stevenson and Bahls 1999). Streams and rivers shallow enough to be wadeable during the period when nuisance problems are greatest may be sampled randomly across the entire width of the stream. If variability is still too great, the focus of assessments could be reduced to an indicator zone (an area having a high potential for nuisance algal growth) with a narrow range of water velocity, depth, and substratum size. For rivers with unwadeable depths, sampling attached algae is commonly confined to the wadeable portions because deeper portions may not have enough light for dense benthic algal growth. However, SCUBA has been used to sample benthic algae in large rivers (Lowe 1974).

In streams and rivers where nuisance algal problems arise from planktonic algal blooms during low-flow conditions, sources of variability in algal biomass (and related factors like low DO) tend to be due to temporal as opposed to spatial variability. Repeated plankton sampling during the low-flow period is strongly recommended to relate nutrients to peak plankton biomass and potential problems of low DO or noxious (toxic, taste, and odor causing) algal blooms. If the goal of estimating algal biomass at a problem site is to compare estimates of biomass to a criterion, then replicate sampling of at least four samples at that site is recommended to characterize the mean and variance in observations. If the goal of sampling is to develop a relationship between nutrients and algal problems for the rivers of a region, or to assess status and trends for nutrient-related problems, then replicate sampling is not as important as accounting for temporal variability and sampling more sites.

Relating nuisance algal problems to nutrient concentrations during stream low-flow conditions can be complicated by a number of factors. Algal problems may be due to a combination of planktonic algae blooming throughout pools and benthic algae along margins of pools. Planktonic algae may settle into sediments of pools and may generate oxygen demand from those sediments. Thus, thorough sampling designs should be employed that consider both spatial and temporal variability in algal biomass and associated nutrients to ensure development of accurate and precise relationships between nuisance algal problems and nutrients.

Attached algal biomass can vary greatly in time as well as space within the same stream. Temporal variability in algal biomass can be addressed by repeated sampling during periods when high algal biomass is most likely a problem. Alternatively, algal biomass can be sampled during periods of peak biomass following flood disturbances. This period of peak biomass may endure from one week to two months, depending upon nutrient concentrations in streams and the severity of flood events. Repeated assessment of algal biomass in streams can be facilitated by using rapid periphyton surveys to reduce sampling and laboratory assay costs (see Stevenson and Bahls 1999). Even though many measurements are being made through time, only one measurement per site can be used to develop biomass-nutrient relationships because of site-specific dependence and problems of repeated measures from the same site (Green 1979; Sokal and Rohlf 1998).

In some cases, the goal of assessment might be to estimate algal biomass at a problem site to compare estimates of biomass to a criterion. In this case, replicate sampling of at least four or many more samples at a site is recommended to characterize the mean and variance in the mean with replicate samples from a site. If the variability in algal biomass is similar to that in the Clark Fork River (see Appendix A case study), as many as 20 replicate samples may be required to detect small changes, which may be important to monitor restoration efforts.

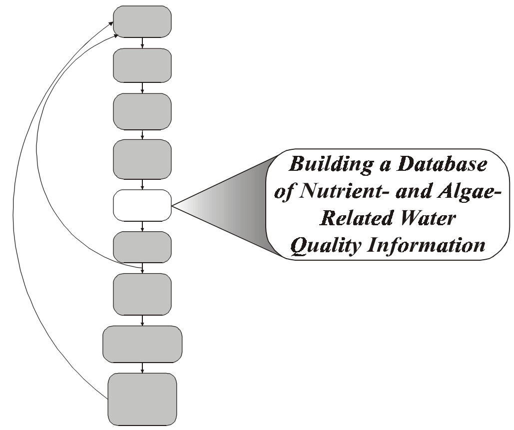
INVOLVEMENT OF CITIZEN MONITORING PROGRAMS

Citizen input can be used to assist in identifying and prioritizing potential problem streams. For example, citizens can be asked (through the use of surveys) to identify streams in which they have observed algal biomass levels that interfere with human uses or impair aesthetic enjoyment. They can also be asked to provide their evaluation of which streams have been affected most and which uses have been impaired to the greatest degree.

While state water quality agencies will likely take the lead in monitoring impaired reaches, citizen monitors may provide much of the monitoring on high-risk reaches. If properly trained and directed, citizen volunteers can be valuable in algal and nutrient monitoring. Citizens, with training, can visually assess algal levels, collect algal samples and freeze them for analysis by an approved laboratory, and may also help in the initial characterization of streams. Citizen monitors can frequently provide more complete flow records by visiting gauges more often than state personnel. Once advised that a stream is high-risk and that the limiting factors have been identified, citizens can help monitor development plans that might affect those factors. Involvement in monitoring programs may lead citizens to effective participation in local planning.

Many excellent resources are available for training citizen monitors. EPA has a volunteer monitoring coordinator (Alice Mayio—E-mail: alice.mayio@epamail.epa.gov) and a web site that lists many resources http://www.epa.gov/OWOW/monitoring/volunteer/spring94/ppresf04.html). Numerous non-governmental organizations, such as the Izaak Walton League, have developed citizen monitoring manuals. One of the best is the Streamkeeper's Guide by the Adopt-a-Stream Foundation (600-128th St. SE, Everett, WA 98208, phone 206-316-8592; web site: http://www.streamkeeper.org/).

Chapter 5. Building a Database of Nutrient- and Algae-Related Water Quality Information



5.1 INTRODUCTION

A database of relevant water quality information can be an invaluable tool to States and Tribes as they develop nutrient criteria. Existing data may provide considerable information that is specific to the region where criteria are to be set. First the data must be located, then the suitability of the data (type and quality) ascertained before they are to be used for historical reconstruction of water quality parameters. It is also important to determine how the data were collected to make future monitoring efforts compatible with earlier approaches.

Databases operate much like spreadsheet applications, but have greater capabilities. While spreadsheets analyze and graphically display small quantities of data, databases store and manage large quantities of data and allow viewing and exporting of data sorted in a variety of ways. Databases can be used to organize existing information, store newly gathered monitoring data, and manipulate data as criteria are being developed. Databases can sort data for export into statistical analyses programs, spreadsheets, and graphics programs. This chapter will discuss the role of databases in nutrient criteria development, and provide a brief review of existing sources of nutrient-related water quality information for streams and rivers.

5.2 DATABASES AND DATABASE MANAGEMENT

This section describes general database structure and provides detailed information on relational and GIS databases. A database is a collection of information related to a particular subject or purpose. Databases are arranged so that they divide data into separate electronic repositories called tables. Data in tables can be viewed and edited, and new data can be added. A single datum is stored in only one table, but can be viewed from multiple locations. Updating one view of a datum will update it in all the various viewable forms. Each table should contain a specific type of information. Data from different tables can be viewed simultaneously according to the user-defined table relationships. That is, the relationship among data in different tables can be defined so that more than one table can be queried or reported, and accessed in a single view. Data stored in tables can be located and retrieved using queries. A query allows the user to find and retrieve only the data that meets user-specified conditions. Queries can also be used to update or delete multiple records simultaneously and to perform built-in or custom calculations of data. Data in tables can be analyzed and printed in specific layouts using reports. Data can be analyzed or presented in a specific way in print by creating a report.

To facilitate data manipulation and calculations, it is highly recommended that historical and present-day data be transferred to a relational database. A relational database is a collection of data items organized as a set of formally-described tables from which data can be accessed or reassembled in many different ways without having to reorganize the database tables. Each table (which is sometimes called a relation) contains one or more data categories in columns. Each row contains a unique instance of data for the categories defined by the columns. The organization of data into relational tables is known as the logical view of the database. In other words, the logical view is the form in which a relational database presents data to the user and the programmer (www.whatis.com/relation.htm). Relational databases are powerful tools for data manipulation and initial data reduction. They allow selection of data by specific, multiple criteria, and definition and redefinition of linkages among data components.

GISs are geo-referenced relational databases that have a geographical component (i.e., spatial platform) in the user interface. Spatial platforms associated with a database allow geographical display of sets of sorted data. Databases with spatial platforms are becoming more common. The system is based on premises that "pictures are worth thousands of words" and most data can be related to a map or other easily understood graphic. GIS platforms such as ArcView™, ArcInfo™, and MapInfo™ are frequently used to integrate spatial data with monitoring data for watershed analysis.

NATIONAL NUTRIENTS DATABASE

The Nutrient Criteria Program has initiated development of a national relational database application that will be used to store and analyze nutrient data. The ultimate use of these data will be to derive ecoregion- and waterbody-specific numeric nutrient criteria ranges. Initially, EPA is developing a Microsoft Access™ application which will ultimately be populated with STORET Legacy Data, USGS NAWQA, NASQAN and Benchmark data, and other relevant nutrient data from universities, States/Tribes, and additional data rich entities. EPA is also developing a compatible, interactive system in an Oracle™ environment which allows for easy web-accessibility, geo-referencing/GIS compatibility, and data analysis on both a State/Tribal, regional, and national basis. The total amount of existing nutrient data nationally is large (>20 gigabytes), and it is anticipated that more data will be entered into the system. The Oracle™ application can easily manage large quantities of data and will provide ample room for expansion as more data are collected. Both the Access™ and the Oracle™ database applications are being designed for compatibility with EPA's latest edition of STORET to avoid duplication of effort for users of STORET and the Nutrients database application. Considerable efforts are also being made to assure compatibility with other database systems (e.g., WQS and RAD) currently being developed in EPA's Office of Water. The Microsoft Access™ application will be available in January 2000; the Oracle™ application will be online in the spring of 2000.

5.3 COLLECTING EXISTING DATA

In some States/Tribes, historical data on streams and rivers are already available. These data can be used to identify reference streams and begin development of potential nutrient criteria. Data should be compiled in a format that is easily imported into database and spreadsheets. Ideally, data will be compiled in the Nutrients Database described above. Potential data sources for river and stream nutrient data that will be useful for developing criteria are discussed below. These data sources contain extensive water quality data, however, data collection should not be limited to these sources. Collection of scientifically sound water quality data from any reliable source is encouraged.

POTENTIAL DATA SOURCES

Potential sources of data include water quality monitoring data from Federal, State, Tribal and local water quality agencies; university studies; and volunteer monitoring information. The data sources described in this section do not encompass the full extent of available data sources. Many State/Tribal, and Federal programs that are regional or site-specific are excellent data sources, but are not included in this discussion.

EPA Water Quality Data

EPA has many programs of national scope that focus on collection and analysis of water quality data. The following presents information on several of the databases and national programs that may be useful to water quality managers as they compile data for criteria development.

STORET

STOrage and RETrieval system (STORET) is EPA's national database for water quality and biological data. EPA's original STORET System, operated continuously since the 1960s, was historically the largest repository of water quality data in the nation. This legacy mainframe-based system will cease to exist in the year 2000. In its place, EPA will support two independent, web-accessible databases. The older database, called the STORET Legacy Data Center (LDC) is the repository of all data held in EPA's original STORET System as of the end of 1998. The newer, modernized database, simply called STORET, is designed as a replacement for the original STORET System. It is the repository for more current data, and offers major improvements in database content and quality control documentation.

Interested parties may view both databases on the World-Wide-Web, where the capability will exist to produce printed reports and download data files. Queries for data via the web will be designed for use by the general public and will require no special training or software. The web site will be announced in the first quarter of FY2000.

STORET (the new STORET system) is a compendium of data supplied by Federal, State, and local organizations which evaluates environmental conditions in the field. The data in STORET is organized by both geographic location and data ownership. Every field study site is identified by at least one latitude/longitude and, where appropriate, also by State/Province, County, drainage basin, and stream reach. Monitoring activities recorded include field measurements, habitat assessments, water and sediment samples, and biological population surveys. Records cover the complete spectrum of physical properties, concentrations of substances, and abundance and distribution of species observed during biological monitoring. STORET is designed for maximum compatibility with commercial software, including Geographic Information Systems such as the ESRI ArcView package, and statistical packages such as PC SAS. STORET download files import easily into all standard spread sheet packages.

Further information about STORET may be obtained by e-mailing STORET@epa.gov, or telephoning toll-free at 1-800-424-9067.

National Surface Water Survey (NSWS)

EPA's National Surface Water Survey consists of two parts: the National Lake Survey and the National Stream Survey. The purpose of the National Lake Survey is to quantify, with known statistical confidence, the current status, extent, and chemical and biological characteristics of lakes in regions of the United States that are potentially sensitive to acidic deposition. The purpose of the National Stream Survey (NSS) is to determine the percentage, extent, and location of streams in the United States that are presently acidic or have low-acid neutralizing capacity and may, therefore, be susceptible to future acidification, as well as to identify streams that represent important classes in each region for possible use in more intensive studies or long-term monitoring. The NSS provides an overview of stream water quality chemical characterisitics in regions of the United States that are expected, on the basis of previous alkalinity data, to contain predominantly low-acid neutralizing capacity waters (EPA website http://www.epa.gov/ceisweb1/ceishome/ceisdocs/usguide/prog(56).htm).

Environmental Monitoring and Assessment Program (EMAP)

The Environmental Monitoring and Assessment Program is an EPA research program designed to develop the tools necessary to monitor and assess the status and trends of national ecological resources (see EMAP Research Strategy on the EMAP website: www.epa.gov/emap). EMAP's goal is to develop the scientific understanding for translating environmental monitoring data from multiple spatial and temporal scales into assessments of ecological condition and forecasts of future risks to the sustainability of the Nation's natural resources. EMAP's research supports the National Environmental Monitoring Initiative of the Committee on Environment and Natural Resources (CENR) (www.epa.gov/emap/). Data from the EMAP program can be downloaded directly from the EMAP website (www.epa.gov/emap/). The EMAP Data Directory contains information on available data sets including data and metadata (language that describes the nature and content of data). Current status of the data directory as well as composite data and metadata files are available on this website.

Clean Lakes Program (CLP)

The EPA Clean Lakes Program was initiated to assess water quality in impaired public lakes and reservoirs and to restore these systems where appropriate. CLP included a monitoring and assessment component to identify the efforts needed to restore water quality. Lakes in this program were selected because they were perceived to have water quality impairment. Major tributaries into lakes and reservoirs included in this program were sampled on a regular basis. EPA encouraged States in its May 1996 section 319 nonpoint source guidance to use section 319 funds to fund eligible activities that might have been funded in previous years in the CLP under Section 314. Data from this program may be useful for positioning river and stream systems on a nutrient gradient continuum, but are unlikely to provide data for reference stream reaches. Information about EPA's CLP can be found at the website: http://www.epa.gov/owowwtr1/lakes/lakes.old.html.

Ecological Data Application System (EDAS)

EDAS is EPA's program-specific counterpart to STORET. EDAS was developed by EPA's Office of Water to manipulate data obtained from biological monitoring and assessment and to assist States/Tribes in developing biocriteria. It contains built-in data reduction and recalculation queries that are used in biological assessment. The EDAS database is designed to enable the user to easily manage, aggregate, integrate, and analyze data to make informed decisions regarding the condition of a water resource. Biological assessment and monitoring programs require aggregation of raw biological data (lists and enumeration of taxa in a sample) into informative indicators. EDAS is designed to facilitate data analysis, particularly the calculation of biological metrics and indexes. Pre-designed queries that calculate a wide selection of biological metrics are included with EDAS. Future versions of EDAS will include the capability to upload data to, and download data from, the distributed version of modernized STORET. EDAS is not a final data warehouse, but is a program or project-specific customized data application for manipulating and processing data to meet user requirements. The EDAS application is currently under development; more information will be available at a later date through the EPA website.

USGS (U.S. Geological Survey) Water Data

The USGS has national and distributed databases on water quantity and quality for waterbodies across the nation. Much of the data for rivers and streams are available through the National Water Information System (NWIS). These data are organized by state, Hydrologic Unit Codes (HUCs), latitude and longitude, and other descriptive attributes. Most water quality chemical analyses are associated with an instantaneous streamflow at the time of sampling and can be linked to continuous streamflow to compute constituent loads or yields. The most convenient method of accessing the local data bases is through the USGS State representative. Every State office can be reached through the USGS home page on the Internet at URL http://www.usgs.gov/wrd002.html.

HBN and NASQAN

USGS data from several national water quality programs covering large regions offer highly controlled and consistently collected data that may be particularly useful for nutrient criteria analysis. Two programs, the Hydrologic Benchmark Network (HBN) and the National Stream Quality Accounting Network (NASQAN) include routine monitoring of rivers and streams during the past 30 years. The HBN consisted of 63 relatively small, minimally disturbed watersheds. HBN data were collected to investigate naturally-induced changes in streamflow and water quality and the effects of airborne substances on water quality. The NASQAN program consists of 618 larger, more culturally influenced watersheds. NASQAN data provides information for tracking water-quality conditions in major U.S. rivers and streams. The watersheds in both networks include a diverse set of climatic, physiographic, and cultural characteristics. Data from the networks have been used to describe geographic variations in water-quality concentrations, quantify water-quality trends, estimate rates of chemical flux from watersheds, and investigate relations of water quality to the natural environment and anthropogenic contaminant sources. Since 1995, the NASQAN Program has focused on monitoring the water quality of four of the Nation's largest river systems—the Mississippi (including the Missouri and Ohio), the Columbia, the Colorado, and the Rio Grande. NASQAN currently operates a network of 40 stations in which the concentration of a broad range of chemicals—including pesticides and trace elements—and stream discharge are measured.

Alexander and others (1996) assembled much of the historical water-quality and streamflow data collected by the NASQAN and HBN on two CD-ROMs, including supporting documentation and quality assurance information (see Internet URL http://wwwrvares.er.usgs.gov/wqn96/). These data are collectively referred to as Water-Quality Networks (WQN). The CD-ROMs are designed to allow users to efficiently browse text files and retrieve data for subsequent use in user-supplied software including spreadsheet, statistical analysis, or geographic information systems. The data may be extracted from one of the CD-ROMs (the "DOS disc") using the supplied DOS-based software, and output in a variety of formats. This software allows the user to search, retrieve, and output data according to user-specified requirements. Alternatively, the ASCII form of the WQN data may be accessed on a second CD-ROM (the "ASCII disc") from user-supplied software including a Web browser, spreadsheet, or word processor.

A comprehensive review of sources, concentrations, and loads of nutrients in the Mississippi River Basin was completed by USGS under the Committee of Environmental Natural Resources. The review focused on analyzing issues related to the Gulf of Mexico hypoxia. Much of Topic 3, Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin, includes data and analysis that could be useful for the development of nutrient criteria in large river systems, such as the Mississippi River. Results of this effort, which was led by the National Oceanic and Atmospheric Administration, have been published and can also be found at the Internet site http://www.nos.noaa.gov/products/pubs_hypox.html.

NAWQA

The USGS National Water-Quality Assessment (NAWQA) Program is building a third national database of stream quality information from data collected and analyzed for more than 50 river basins and aquifer systems, called Study Units, across the Nation. NAWQA studies are based on a complex sampling design that targets specific land uses and hydrologic conditions in addition to assessing the most important aquifers and large streams and rivers in each area studied. Gilliom and others (1995) describe the NAWQA sampling design in detail. A comprehensive data screening, computer retrieval, and review of existing data on nutrients in streams was completed for each of the first 20 Study Units (Mueller et al. 1995). A major component of the sampling design for streams is to target specific watersheds influenced primarily by a single dominant land use (agricultural or urban) that is important in a particular area of the United States. Some of the watersheds were selected as undeveloped areas relative to the rest of the Study Unit to use in comparative analysis of land-use effects on water quality. Water-quality data collection during 1992-1996 include analyses of eight nutrient species from about 8,500 samples of streams and rivers in the first 20 Study Units. A data set used for national synthesis of water quality has been compiled and can be viewed and downloaded via the Internet URL http://wwwrvares.er.usgs.gov/nawqa/nutrient.html. Mueller and others (1997) describe quality control of the NAWQA stream data and Mueller (1998) provides a rigorous assessment of the quality of these data.

WEBB

The Water, Energy, and Biogeochemical Budgets (WEBB) program was developed by USGS to study water, energy, and biogeochemical processes in a variety of climatic/regional scenarios. Five ecologically diverse watersheds, each with an established data history, were chosen. This program may prove to be a rich data source for ecoregions in which the five watersheds are located. Many publications on the WEBB project are available. See the USGS website for more details (http://water.usgs.gov/nrp/webb/about.html).

USDA

Agricultural Research Service (ARS)

ARS houses Natural Resources and Sustainable Agricultural Systems, which has seven national programs to examine the effect of agriculture on the environment. The program on Water Quality and

Management addresses the role of agriculture in nonpoint source pollution through research on Agricultural Watershed Management and Landscape Features, Irrigation and Drainage Management Systems, and Water Quality Protection and Management Systems. Research is conducted across the country and several models and databases have been developed. Information on research and program contacts is listed on the website (http://www.nps.ars.usda.gov/programs/nrsas.htm).

Forest Service

The Forest Service has designated research sites across the country, many of which are Long Term Ecological Research (LTER) sites. Many of the data from these experiments are available in the USFS databases located on the website (http://www.fs.fed.us/research/). Most of the data are forest-related, but may be of use for determining land uses and questions on silviculture runoff.

National Science Foundation (NSF)

The National Science Foundation funds projects for the LTER Network. The Network is a collaboration of over 1,100 researchers investigating a wide range of ecological topics at 24 different sites nationwide. The LTER research programs are not only an extremely rich data source, but also a source of data available to anyone through the Network Information System (NIS), the NSF data source for LTER sites. Data sets from sites are highly comparable due to standardization of methods and equipment. Data can be accessed from the website http://www.lternet.edu/research/data/nis/.

U.S. Army Corps of Engineers (COE)

The U.S. Army Corps of Engineers is responsible for more than 750 reservoirs. Many have extensive monitoring data that could contribute to the development of nutrient criteria for tributaries to those reservoirs. The COE focuses more on water quantity issues than on water quality issues. As a result, much of the river and stream system data collected by the COE does not include nutrient or algal constituents. Nonetheless, the COE does have a large water sampling network and supports USGS and EPA monitoring efforts in many programs. A list of the water quality programs that the COE actively participates in was compiled in 1997. This information can be found at the website: http://cw71.cw-wc.usace.army.mil/wqinfo/wq98sem/ANNWQMGT.HTM.

U.S. Department of the Interior, Bureau of Reclamation (BuRec)

The Bureau of Reclamation manages many irrigation and water supply reservoirs in the West, some of which may have operational data available. These data focus on water supply information and limited water quality data. However, real time flow data are collected for rivers supplying water to BuRec, which may be useful for the flow component of criteria development. These data can be gathered on a site-specific basis from the BuRec website: www.usbr.gov. Extensive remote sensing data are available from the website: http://wais.rsgis.do.usbr.gov/html/rsgig_wq.html.

State/Tribal Monitoring Programs

Most states monitor some subset of stream and river systems within their borders for algal and nutrient variables. Data collected by State/Tribal water quality monitoring programs can be used for nutrient criteria development. These data should be available from the agencies responsible for monitoring.

Volunteer Monitoring Programs

Many States have volunteer water quality monitoring programs. Some programs are state-sponsored, while others are independent organizations such as Adopt-A-Stream. Citizens in many areas donate their time, money, or experience to aid State, Tribal, and local governments in collecting water quality data. Volunteers analyze water samples for DO (dissolved oxygen), nutrients, pH, temperature, and a host of other water constituents; evaluate the health of stream habitats and aquatic biological communities; note stream-side conditions and land uses that may affect water quality; catalogue and collect beach debris; and restore degraded habitats.

State and local agencies may use volunteer data to screen for water quality problems, establish trends in waters that would otherwise be unmonitored, and make planning decisions. Volunteers benefit from learning more about their local water resources and identifying what conditions or activities might contribute to pollution problems. As a result, volunteers frequently work with clubs, environmental groups, and State/Tribal or local governments to address problem areas.

The EPA supports volunteer monitoring and local involvement in protecting our water resources. EPA support takes many forms including: sponsoring national and regional conferences to encourage information exchange among volunteer groups, government agencies, businesses, and educators; publishing sampling methods manuals for volunteers; producing a nationwide directory of volunteer programs; and providing technical assistance (primarily on quality control and lab methods) and Regional coordination through the ten EPA Regional offices. In addition, grants to States/Tribes that can be used to support volunteer monitoring in lakes and for nonpoint source pollution control are managed by the EPA Regions (http://www.epa.gov/OWOW/monitoring/volunteer/epavm.html).

<u>Adopt-A-Stream</u>

The Adopt-A-Stream Foundation (AASF) is a non-profit organization that works to increase public awareness and involvement in water quality issues, stream enhancement, and environmental education. Their two main areas of focus are Environmental Education and Habitat Restoration. AASF seeks to protect streams through volunteer work, encouraging school and community groups, sports clubs, civic organization, and individuals to become “Streamkeepers.” “Adoption” of a stream requires that volunteers provide long-term care of the stream and establish stream monitoring, restoration, and community-wide environmental education activities. AASF provides education materials, classes, and tools for monitoring. Data collected through the volunteer monitoring associated with Adopt-A-Stream is usually site-specific, focusing on a single stream. However, if volunteers have been properly trained, the data collected may be useful in helping identify streams at risk for nutrient problems. The AASF website contains additional information on this organization and data they may be able to provide (http://www.streamkeeper.org/).

<u>American Heritage Rivers</u>

The American Heritage Rivers Initiative is a program launched by President Clinton to help communities restore their local waters and waterfront areas. Participation is voluntary and must be initiated by the community. To date, fourteen rivers have been designated on the basis of historical, economic, and environmental considerations. One goal of the program is to develop additional information that can be used by communities to improve any river system. Through the American Heritage Rivers website (http://www.epa.gov/OWOW/heritage/rivers.html), valuable information about our nation's rivers is easily available to everyone. Information organized geographically on flood events, population change, road networks, condition of water resources, and partnerships already at work in the area is available. Additionally, customized maps and environmental and educational assessment models will be made available through this initiative.

Electric Utilities

Many electric utilities own reservoirs for hydroelectric power generation, and are required to monitor the reservoirs' water quality. The largest of these, the Tennessee Valley Authority (TVA), has extensive chemical and biological monitoring data from most of its reservoirs from the early 1980s to the present. Data collected in conjunction with hydroelectric reservoirs must be gathered from the facility owners or managers.

Drinking Water Facilities

Many local drinking water facilities are supplied from river systems. These facilities continuously monitor some water quality parameters at the intake pipe. Nutrients are infrequently monitored by most of these facilities, but supplemental data, i.e., turbidity, pH, and flow are usually measured. These data may not provide the necessary parameters for deriving criteria, but may be very useful in combination with State/Tribal water quality monitoring data to develop criteria. Data from these facilities should be accessed locally for the waterbody of concern.

Academic and Literature Sources

Many research studies are conducted by academic institutions that may provide data useful for developing nutrient criteria. Much of the research conducted by the academic community concentrates on unimpaired or minimally impaired systems. While data collected from these sources may not be directly representative of the population of stream systems within an ecoregion, they could be useful for identifying reference conditions. Academic research also tends to be site-specific and span a limited number of years, although data for some systems may span 20 years or more. Academic research data should be available from researchers and the scientific literature.

QUALITY OF HISTORICAL DATA

The value of older historical data sets is a recurrent problem because data quality is often unknown. Knowledge of data quality is also problematic for long-term data repositories such as STORET and long-term State databases, where objectives, methods, and investigators may have changed many times over the years. The most reliable data tend to be those collected by a single agency, using the same protocol, for a limited number of years. Supporting documentation should be examined to determine the consistency of sampling and analysis protocols. Investigators must determine the acceptability of data contained in large, heterogeneous data repositories. Considerations and requirements for acceptance of these data are described below.

Location

STORET and USGS data are geo-referenced with latitude, longitude, and Reach File 3 (RF3) codes. Geo-reference data can be used to select specific locations, or specific USGS Hydrologic Units. In addition, STORET often contains a site description. Knowledge of the rationale and methods of site selection from the original investigators may supply valuable information. Metadata of this type, when known, is frequently stored within large long-term databases.

Variables and Analytical Methods

Thousands of variables are recorded in database records. Each separate analytical method yields a unique variable. For example, five ways of measuring TP results in five unique variables. We do not recommend mixing analytical methods in data analyses because methods differ in accuracy, precision, and detection limits. Data analyses should concentrate on a single analytical method for each parameter of interest. Selection of a particular "best" method may result in too few observations, in which case it may be more fruitful to select the most frequently used analytical method in the database. Data may have been recorded using analytical methods under separate synonymous names, or analytical methods incorrectly entered when data were first added to the database. Review of recorded data and analytical methods recorded by knowledgeable personnel is necessary to correct these problems.

Laboratory Quality Control (QC)

Laboratory QC data (blanks, spikes, replicates, known standards, etc.) are infrequently reported in larger data repositories. Records of general laboratory quality control protocols and specific quality control procedures associated with specific datasets are valuable in evaluating data quality. However, premature elimination of lower quality data can be counterproductive, because the increase in variance caused by analytical laboratory error may be negligible compared to natural variability or sampling error, especially for nutrients and related water quality parameters. However, data of uncertain and undocumented quality should not be accepted.

Data Collecting Agencies

Selecting data from particular agencies with known, consistent sampling and analytical methods and known quality will reduce variability due to unknown quality problems. Requesting data review for quality assurance from the collecting agency will reduce uncertainty about data quality.

Time Period

Long-term records are critically important for establishing trends. Determining if trends exist in the time series database is also important for characterizing reference conditions for nutrient criteria. Length of time series data needed for analyzing nutrient data trends is discussed in Chapter 4.

Index Period

An index period for estimating average concentrations can be established if nutrient and water quality variables were measured through seasonal cycles. The index period may be the entire year or the summer growing season. The best index period is determined by considering stream characteristics for the region, the quality and quantity of data available, and estimates of temporal variability (if available). Additional information and considerations for establishing an index period are discussed in Chapter 4.

Representativeness

Data may have been collected for specific purposes. Data collected for toxicity analyses, effluent limit determinations, or other pollution problems may not be useful for developing nutrient criteria. Further, data collected for specific purposes may not be representative of the region or stream classes of interest. The investigator must determine if stream systems or a subset of the stream systems in the database are representative of the population of stream systems to be characterized. If a sufficient sample of representative systems cannot be found, then a new survey will be necessary (see Chapter 4).

Gather New Data

New data should be gathered following the sampling design protocols discussed in Chapter 4. New data collection activities for developing nutrient criteria should focus on filling in gaps the database and collecting regional monitoring data. Data gathered under new monitoring programs should be imported into databases or spreadsheets and merged with existing data for criteria development.

Data Reduction

Data reduction requires a clear idea of the analysis that will be performed and a clear definition of the sample unit for the analysis. For example, a sample unit might be defined as "a watershed during July-August". For each variable measured, a mean value would then be estimated for each watershed in each July-August index period on record. Analyses are then done with the observations (estimated means) for each sample unit, not with the raw data. Steps in reducing the data include:

- Selecting the long-term time period for analysis;
- Selecting an index period;
- Selecting relevant chemical species;
- Identifying the quality of analytical methods;
- Identifying the quality of the data recorded; and
- Estimating values for analysis (mean, median, minimum, maximum) based on the reduction selected.

QUALITY ASSURANCE/QUALITY CONTROL

The validity and usefulness of data depend on the care with which they were collected, analyzed and documented. The EPA provides guidance on data quality assurance (QA) and quality control (USEPA 1998b) to assure the quality of data. Factors that should be addressed in a QA/QC plan are elaborated below. The QA/QC plan should state specific goals for each factor and should describe the methods and protocols used to achieve the goals. The five factors discussed below are: representativeness, completeness, comparability, accuracy and precision.

Representativeness

Sampling program design (when, where, how you sample) should produce samples that are *representative* or typical of the environment being described. Sampling designs for developing nutrient criteria are addressed in Chapter 4.

Completeness

Data sets are often incomplete because of spilled samples, faulty equipment, and/or lost field notebooks. A QA/QC plan should describe how complete the data set must be in order to answer the questions posed (with a statistical test of given power and confidence) and the precautions being taken to ensure that completeness. Data collection procedures should document the extent to which these conditions have been met. Incomplete data sets may not invalidate the collected data, but may reduce the rigor of statistical analyses. Therefore, precautions should be taken to ensure data completeness. Precautions to ensure completeness may include collecting extra samples, having back-up equipment in the field, installing alarms on freezers, copying field notebooks after each trip, and/or maintaining duplicate sets of data in two locations.

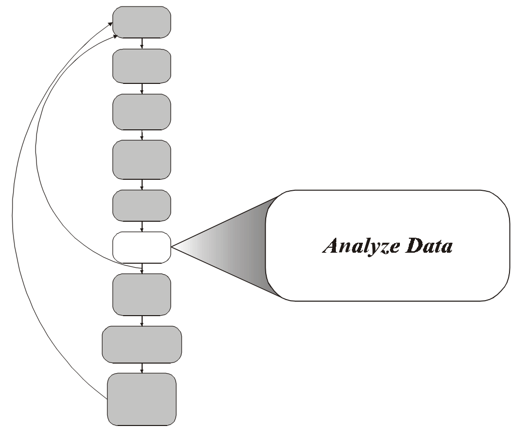
Comparability
In order to compare data collected under different sampling programs or by different agencies, sampling protocols and analytical methods must demonstrate comparable data. The most efficient way to produce comparable data is to use sampling designs and analytical methods that are widely used and accepted.

Accuracy
To assess the accuracy of field instruments and analytical equipment, a standard (a sample with a known value) must be analyzed and the measurement error or bias determined. Internal standards should periodically be checked with external standards provided by acknowledged sources. At Federal, State, Tribal, and local government levels, the National Institute of Standards and Technology (NIST) provides advisory and research services to all agencies by developing, producing, and distributing standard reference materials. For calibration sevices and standards see: http://ts.nist.gov/ts/htdocs/230/233/calibration/home.html.

Standards and methods of calibration are typically included with turbidity meters, pH meters DO meters, and DO testing kits. The USGS, the EPA and some private companies provide reference standards or QC samples for nutrients. Reference standards for chlorophyll are also available from the EPA and some private companies, although chlorophyll standards are time and temperature sensitive because they degrade over time.

Variability
Natural variability rather than imprecision in the method used, is usually the greatest source of error in the constituent measured. The variability in field measurements and analytical methods should be demonstrated and documented to identify the source of variability when possible. EPA QA/QC guidance provides an explanation and protocols for measuring sampling variability (USEPA 1998b). Methods for creating a chlorophyll standard to determine if the spectrophotometer is measuring chlorophyll consistently from one year to the next or from the beginning to the end of an analytical run are described in Wetzel and Likens (1991). In addition, a large number of replicates for each sample time and site must be collected because the largest source of variation is likely to be natural (i.e., in the samples).

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 $\mu g/L$. 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.

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

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

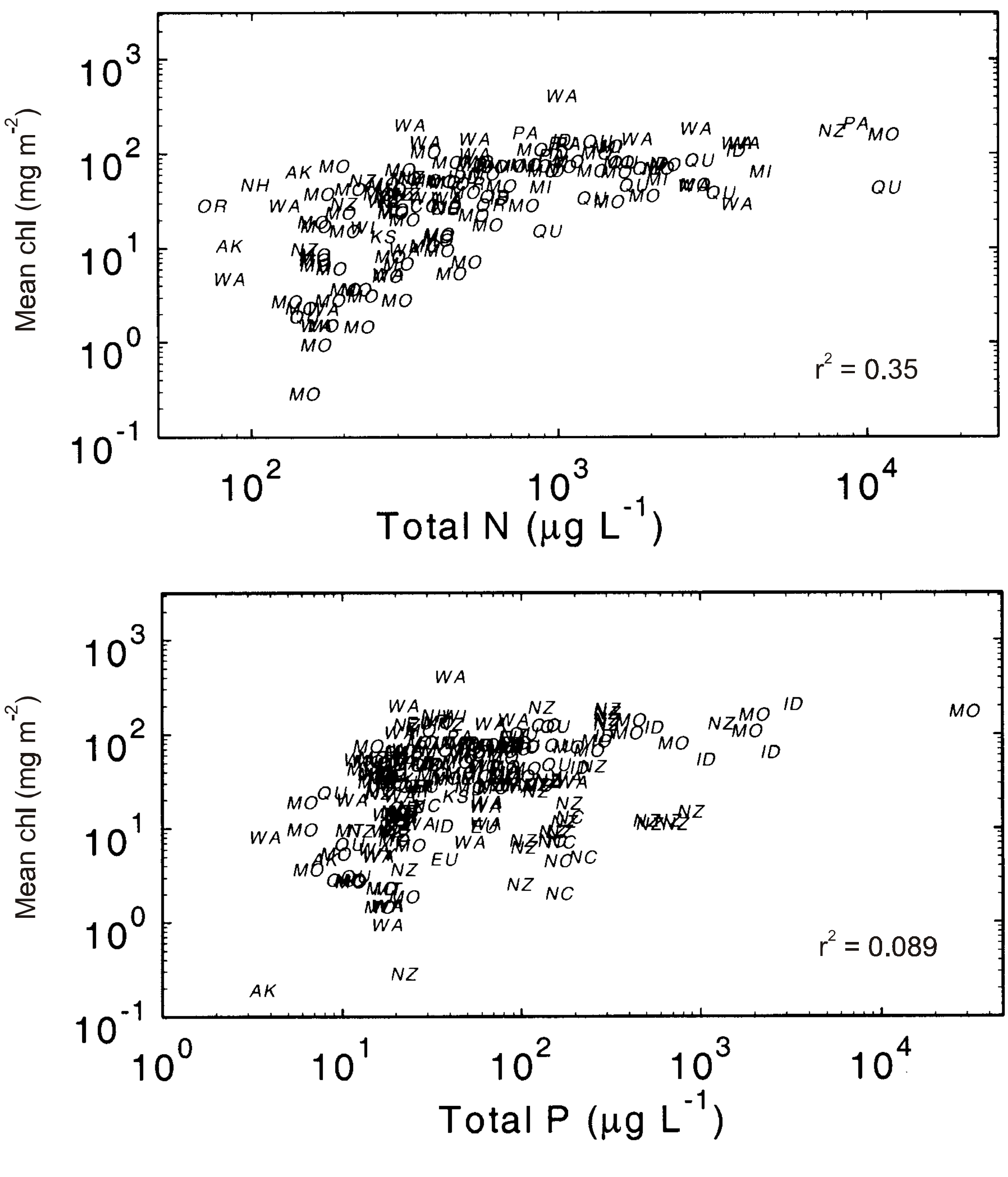


Figure 7. Relationships of log-transformed mean chlorophyll *a* as a function of TN and TP.

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

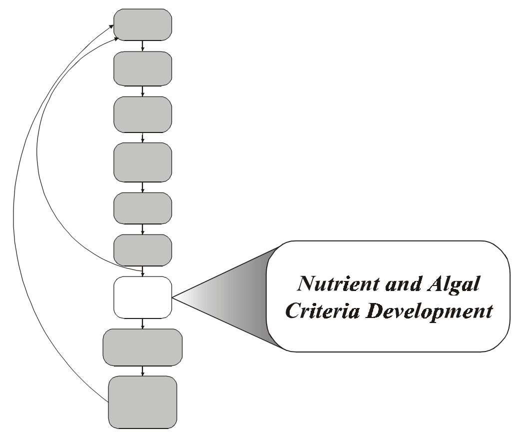
Various statistical tests are used to assess the hypotheses being tested. Statistical tests of significance differ in their applicability to the dataset of interest, and the power of the test (the ability of the test to detect a false null hypothesis). A parametric test of significance assumes a normal 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

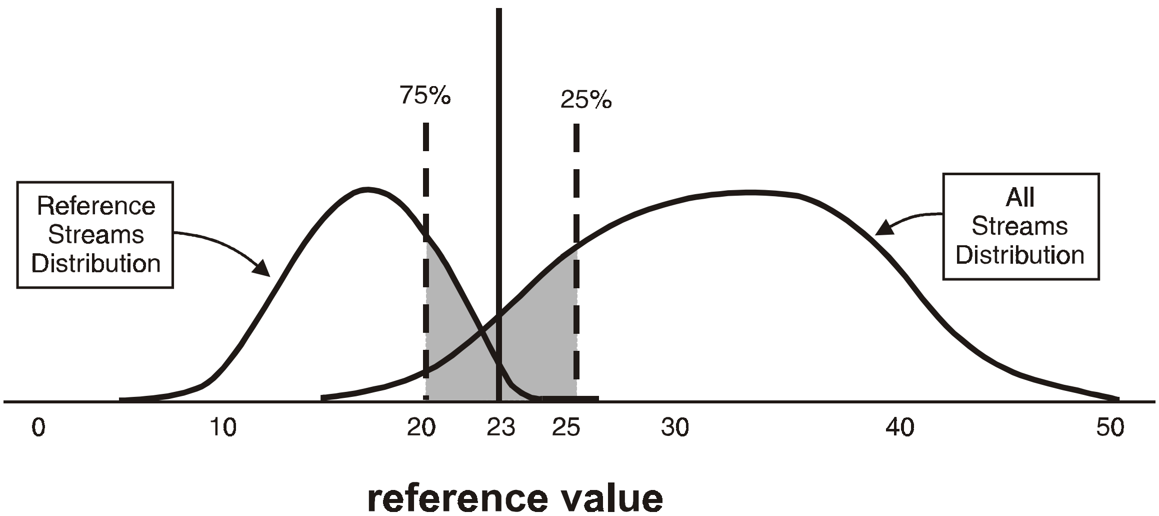


Figure 8. Selecting reference values for total phosphorus concentration (μg/L) using percentiles from reference streams and total stream populations.

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

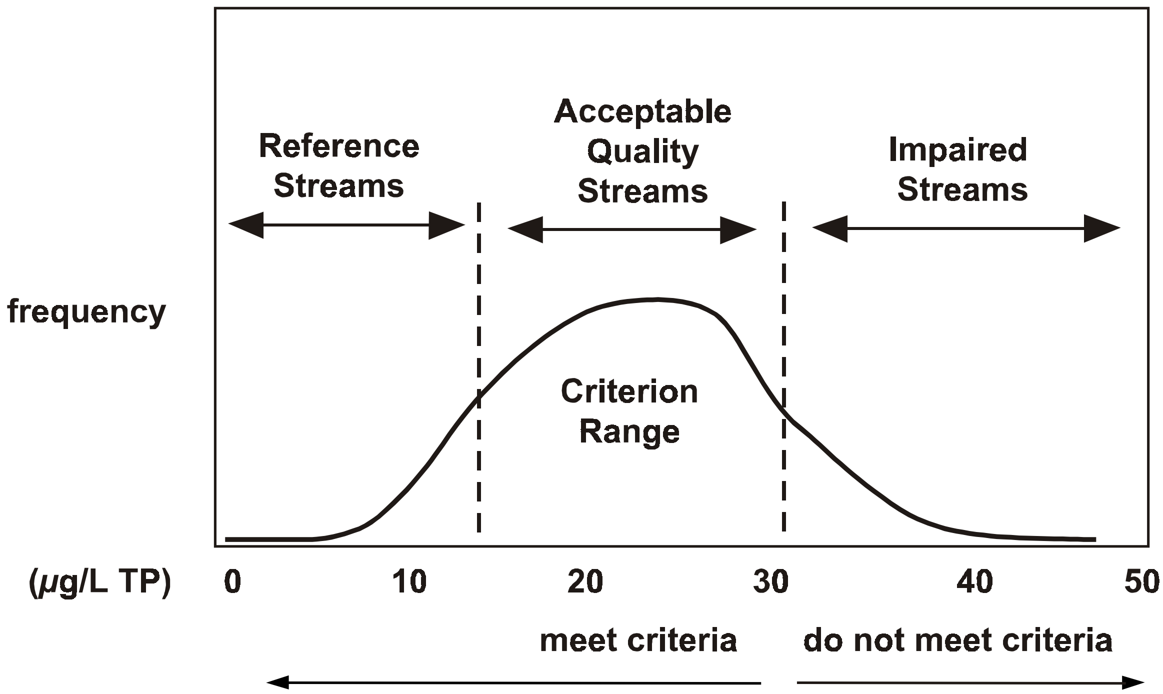


Figure 9. Frequency distribution divided into three segments that represent (from left to right) high-quality reference streams, acceptable quality streams, and impaired streams.

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/m^2						
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.

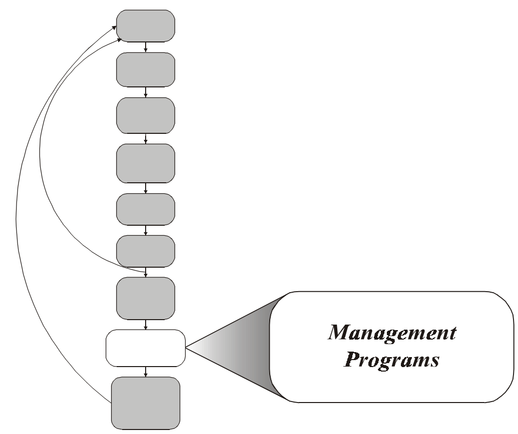
Designated Use Approaches
Once a regional criterion has been established, it is subject to periodic review and calibration. Any State or Tribe in the region may elect to use the criterion as the basis for developing its own criteria to protect designated uses for specific stream classes. This is entirely 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 8. Management Programs



8.1 INTRODUCTION

This chapter provides information on regulatory and non-regulatory programs that may utilize or be affected by nutrient criteria, as well as management solutions for problems associated with varying streamflow conditions. This chapter is intended to inform resource managers and foster potential links among regulatory and non-regulatory programs to best manage watersheds. Information about other agency programs that may assist in implementing criteria and maintaining water quality is also included.

The information provided by nutrient surveys of stream systems in a region will permit the resource manager to rank stream systems by trophic state; i.e., the manager should be able to classify systems according to the degree of nutrient enrichment. Stream systems can be selected for priority attention for management action. Documented stream nutrient and algal conditions and an understanding of regional public preferences regarding limits of productivity can be used to establish three categories of streams:

1. Systems with algal and/or nutrient problems. The most severely degraded waterbodies requiring extensive, expensive restoration.

2. Systems with a strong potential for developing algal problems (factors other than nutrients are unlikely to be limiting). The intermediate streams in need of remedial management to improve conditions requiring various levels of expense and manpower depending on the characteristics and problems identified in each case.

3. Systems with a low potential for developing algal problems that do not contribute to degraded nutrient conditions in downstream waterbodies. The systems in excellent condition requiring no restoration and for which management is essentially the protection of this resource through careful watershed land use planning and diligent observation of conditions. This is usually a relatively low cost option allowing for the protection of many such waterbodies with little expenditure of budget or personnel.

Systems with high nutrient loading but low potential for developing algal problems due to other limiting factors should be prioritized based on the potential for degradation of downstream receiving waters. The management strategies required for nutrient reduction within streams and those for lakes and estuaries are not different, so these processes should be linked when management plans are being formulated.

The next logical action is the design of management plans to enhance collective water body resources. The initial categorization helps set priorities for the best use of limited personnel and funds by selecting some optimal combination of many low cost but effective projects combined with some important restoration projects, and perhaps long range planning to begin to address major restoration of one or two important stream systems on an incremental basis.

This chapter is separated into discussions of point source and nonpoint source programs. Each program is discussed and a list of source information or contacts is provided. This chapter is intended to aid the resource manager in identifying programs that may assist in implementation of nutrient criteria. These programs include regulatory and non-regulatory programs that address both point and nonpoint sources of nutrients. Consultation with these programs is recommended for watershed and development planning activities. Linking with other programs may allow maximization of resources for addressing water quality concerns.

8.2 MANAGING STREAMFLOW CONDITIONS

LOW FLOWS

Maintaining flow is often essential to habitat protection. In many regions of the United States, stream segments periodically lose water due to irrigation, industrial and municipal withdrawals; and/or diversion for hydroelectric power; evaporation; and groundwater infiltration. Additionally, during low-flow conditions, impacts from point source discharges of chemical stressors are typically greatest, because effluent constitutes a larger percentage of (or sometimes all) stream water at low flow, with increased pollutant concentration. National Pollutant Discharge Elimination System (NPDES) permits based on low flow conditions (e.g., 7Q10) often cannot antic-ipate various combinations of climatic conditions and water demand that lead to exceedingly low flows.

Impacts attributable to low flows caused by human actions can be mitigated by several in-stream restoration techniques, including:

- Reducing channelization,
- Restoring wetlands for conservation and storage purposes thereby restoring natural hydrologic regimes,
- Controlling evaporation through restoration of the riparian canopy,
- Replacing exotic riparian plant species that have high evapotranspiration rates with native species that have lower transpiration rates,
- Constructing drop structures to create pools that provide protection for aquatic life during low-flow periods,

- Increasing channel depth and undercut banks to provide protective areas for fish and other species during periods of low flow, and
- Increasing groundwater recharge to streams through increased infiltration (e.g., reduced imperviousness in recharge areas).

Minimum flows can also be addressed by applying techniques in the surrounding watershed, such as managing watershed land use to prevent excessive dewatering. Restoration practices to mitigate low velocity/low-flow conditions often require close collaboration with other resource management agencies (e.g., USDA Forest Service), zoning authorities (e.g., county governments), and agricultural extension agencies. Several agricultural activities contribute to low velocity/low flow conditions. Agricultural extension agencies have developed specific techniques to modify the practices that result in low-flow impact to streams. For example, irrigation plans can be optimized to reduce the demand for water that is diverted directly from the stream. Changing crop rotations and using less water-intensive crop alternatives are other tools that have been used effectively to address low velocity/low-flow situations. Source: [http://www.epa.gov/owowwtr1/NPS/Ecology/chap3.html]

HIGH FLOWS

High-energy flows can erode substrate and bank materials, destabilize the physical structure of aquatic habitats, eradicate resident aquatic organisms, and destroy eggs located in the benthic environment. Seasonal cycles of high-energy flow events (e.g., spring floods) are typical in most aquatic systems. Habitat alteration and degradation, however, may exacerbate impacts of high-energy flows and contribute to impairment of designated uses. For instance, in a channelized stream with minimal riparian vegetation, flow velocity and volume will likely be much greater than would be expected in a "natural stream," thereby increasing its erosive potential.

Two aspects of flooding are considered here. It has recently been recognized that water retention structures remove the natural flooding that is part of a normal stream ecosystem (the flood pulse concept). Such floods are known to reduce levels of algae and macrophytes and may be beneficial to stream communities otherwise. The floods appear destructive on the short term, but most stream organisms are adapted to some level of flooding.

Alternatively, channel alteration and watershed modification can lead to abnormally high water velocities through the stream channel and amplify the effects of floods. For example, channelization can reduce the amount of refugia used by stream organisms to escape floods. Removal of riparian vegetation, urbanization, and deforestation of watersheds can lead to much greater peak flows during floods for a given amount of rain. Watershed disturbance can also lead to increases in sedimentation, which will scour away excessive algal biomass and, if deposited, make it difficult for periphyton to become established. However, such sediment will compromise the ecological integrity by harming fish and invertebrates in the stream channels.

In-stream and riparian techniques that can mitigate high flow impacts include:

- Restoring natural stream meander and channel complexity;
- Increasing substrate roughness;
- Promoting growth of riparian vegetation, which serves as a drag on flows;

- Modifying land use along buffers and other source areas; and
- Creating plunge pools and flow baffles to decrease the high energy of discharged waters.

These in-stream practices may need to be accompanied by techniques applied in the surrounding watershed, such as upland revegetation or the establishment of nonpoint source best management practices (BMPs).

Resource management agencies, for example, can encourage or allow beavers to colonize stream segments; beaver dams create wetlands and retain water that supplements low flow during dry periods. Restored wetlands can have the same effect as a beaver dam. In areas below dams where flow is very stable and excessive growths of macrophytes and periphyton are common, water releases to mimic natural floods may be considered. Local zoning authorities have also begun to encourage impervious area reduction in watersheds through land-use ordinances. Increased infiltration and reduced peak flows from rapid runoff contributes to a more sustained base flow to the stream from groundwater discharge. Source: [http://www.epa.gov/owowwtr1/watershed/wacademy/acad2000/river/]

8.3 MANAGING POINT SOURCE POLLUTION

The term "point source" means any discernible, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged. This term does not include agricultural storm water discharges and return flows from irrigated agriculture. This section describes some of the regulatory programs that permit point source discharges into rivers and streams. The regulatory programs discussed here apply to federal requirements of the Clean Water Act (Section 303). State, Tribal, and local governments frequently have regulatory programs that operate on agency specific requirements. These agencies should be considered in management planning activities.

WATER QUALITY STANDARDS

Anti-degradation

Water quality standards include an anti-degradation policy and methods through which the State or Tribe implements the anti-degradation policy. Anti-degradation is a policy required in State water quality standards to protect waters from degradation. At a minimum, States must maintain and protect the quality of waters to support existing uses. Anti-degradation was originally based on the spirit, intent, and goals of the Clean Water Act, especially the clause “...restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (USEPA 1994). The water quality standards regulation sets out a three-tiered anti-degradation approach for the protection of water quality.

Tier 1

Maintains and protects existing uses and the water quality necessary to protect these uses (40 CFR 131.12[a][1]). An existing use can be established by demonstrating that fishing, swimming, or other uses have actually occurred since November 28, 1975, or that the water quality is suitable to allow such uses to occur, whether or not such uses are designated uses for the water body in question.

Tier 2
Protects the water quality in waters whose quality is better than that necessary to protect "fishable/swimmable" uses of the water body (40 CFR 131.12[a][2]). The water quality standards regulation requires that certain procedures be followed and certain showings be made (an "anti-degradation review") before lowering water quality in high quality waters. In no case may water quality for a tier 2 water body be lowered to a level at which existing uses are impaired.

Tier 3
Preserves outstanding national resource waters (ONRWs), which are provided the highest level of protection under the anti-degradation policy (40 CFR 131.12[a][3]). ONRWs generally include the highest quality waters of the United States. However, the ONRW anti-degradation classification also offers special protection for waters of "exceptional ecological significance," i.e., those water bodies which are important, unique, or sensitive ecologically, but whose water quality, as measured by the traditional parameters such as dissolved oxygen or pH, may not be particularly high. Waters of exceptional ecological significance also include waters whose characteristics cannot adequately be described by traditional parameters (such as wetlands and estuaries).

Anti-degradation implementation procedures address the measures used by States and Tribes to ensure that permits and control programs meet water quality standards and anti-degradation requirements.

General Policies
The water quality standards regulation allows States and Tribes to include implementation in their standards policies and provisions, such as mixing zones, variances, and low-flow exemptions. Such policies are subject to EPA review and approval. These policies and provisions should be specified in the State or Tribe's water quality standards document. The rationale and supporting documentation should be submitted to EPA for review during the water quality standards review and approval process.

Mixing Zones
States and Tribes may, at their discretion, allow mixing zones for dischargers. The water quality standards should describe the methodology for determining the location, size, shape, outfall design, and in-zone quality of mixing zones. Careful consideration must be given to the appropriateness of a mixing zone where a substance discharged is bioaccumulative, persistent, carcinogenic, mutagenic, or teratogenic.

Low-Flow Provisions
State and Tribal water quality standards should protect water quality for the designated and existing uses in critical low-flow situations. States and Tribes may, however, designate a critical low-flow below which numerical water quality criteria do not apply. When reviewing standards, States and Tribes should review their low-flow provisions for conformance with EPA guidance.

Water Quality Standards Variances
As an alternative to removing a designated use, a State or Tribe may wish to include a variance as part of a water quality standard, rather than changing the entire standard, especially if the State or Tribe believes that it can ultimately be attained. By maintaining the standard rather than changing it, the State or Tribe will assure that further progress is made in improving water quality and attaining the standard. Variances are temporary, subject to review every three years, and may be extended upon expiration. If a

variance specifies an interim criterion applicable for the duration of the variance for a particular pollutant, a long-term underlying goal criterion is also specified that is adequate to protect the designated use. EPA has approved variances in the past and will continue to do so if:

- The variance is included as part of the water quality standard;
- The variance is subjected to the same public review as other changes in water quality standards;
- The variance is granted based on a demonstration that meeting the standard is not feasible due to the presence of any of the same conditions as if a designated use were being removed (these conditions are listed in section 131.10(g) of the water quality standards regulation); and
- Existing uses will be fully protected.

For additional information, see http://www.epa.gov:80/ostwater/econ/chaptr5.pdf.

NPDES PERMITS

The Clean Water Act requires wastewater dischargers to have a permit establishing pollution limits, and specifying monitoring and reporting requirements. More than 200,000 sources are regulated by the NPDES permits nationwide. These permits regulate household and industrial wastes that are collected in sewers and treated at municipal wastewater treatment plants. Permits also regulate industrial point sources and concentrated animal feeding operations that discharge into other wastewater collection systems or that have the potential to discharge directly into receiving waters. Permits regulate discharges with the goals of 1) protecting public health and aquatic life, and 2) assuring that every facility treats wastewater. Typical pollutants regulated by NPDES are "conventional pollutants" such as fecal coliforms or oil and grease from the sanitary wastes of households, businesses, and industries and "toxic pollutants" including pesticides, solvents, polychlorinated biphenyls (PCBs), dioxins, and heavy metals that are particularly harmful to animal or plant life. "Non-conventional pollutants" are any additional substances that are not conventional or toxic that may require regulation, including nutrients such as N and P. [Source: http://www.epa.gov/owm/gen2.htm].

Discharge monitoring data for pollutants limited and/or monitored pursuant to NPDES permits issued by States, Tribes, or EPA are required to be stored in the central EPA Permit Compliance System (PCS). The assessment of point source loadings is not a simple process of assessing PCS data, even though PCS is an important data source. The PCS database does not provide complete information for important N sources. Most PCS N data is generated by water quality-based permit limitations on ammonia, often applied in discharges to smaller streams. Few data exist in PCS on other forms of N, or TN; and data for TP is not frequently found in PCS. This situation exists largely because most permits do not include limits and/or monitoring requirements for N or P. The lack of nutrient limits and/or monitoring requirements in permits is due to a general lack of State water quality standards for these parameters. [Source: http://www.epa.gov/msbasin/protocol.html]

The NPDES Storm Water Permitting Program

Storm water runoff is one of the remaining causes of contaminated lakes, streams, rivers, and estuaries throughout the country. Pollution in storm water runoff is responsible for closing beaches and shellfish harvesting areas, contaminating fish, and reducing populations of water plants and other aquatic life. High flows of storm water runoff cause flooding, property damage, erosion and heavy siltation. The

Clean Water Act requires EPA and States/Tribes to implement a national storm water control program to correct these problems. In the first phase of the program, discharges of storm water from municipal separate storm sewers serving populations of over 100,000 and from industrial facilities are illegal unless controlled by an NPDES storm water permit. Phase II of the program required that EPA, in consultation with the States, conduct a study identifying additional sources of storm water contamination and establish procedures and methods to control these discharges.
Source: [http://www.epa.gov/owmitnet/pipes/wetlib/disc_pap.txt]

Construction Permits

The 1987 Congressional Amendments to the Clean Water Act required EPA to control pollution from storm water discharges. Phase I storm water regulations were finalized by EPA in 1990, and NPDES permit coverage was required for construction sites disturbing five or more acres beginning in 1992. General permits provide EPA with an effective mechanism to regulate these discharge from tens of thousands of construction sites, thus protecting and improving surface water quality across the nation.

EPA Regions 1, 2, 3, 7, 8, and 9 have reissued the general permit which authorizes the discharge of storm water associated with construction activity disturbing five or more acres (Phase I sources) and smaller Phase II sources that are designated by the Agency on a case-by-case basis. This multi-regional permit is know as the "Construction General Permit" (CGP). As used in the permit, the term "storm water associated with construction activity" refers to category (x) of the definition of "discharge of storm water associated with industrial activity" which includes construction sites and common plans of development or sale that disturb five or more acres (See 40 CFR 122.26 [b][14]). This permit replaces the Baseline Construction General Permit issued by EPA in September 1992. Issuance of the new CGP will not affect areas where the State is the NPDES permitting authority.

Region 4 has issued a separate construction general permit for the State of Florida and Indian Country lands in Florida, Mississippi, Alabama, and North Carolina. Region 6 is also issuing its own construction general permit for the States of Texas and New Mexico; Indian Country lands in Texas, New Mexico, Oklahoma and Louisiana; and construction activity at oil, gas, and pipeline facilities in Oklahoma in the near future. [Source: http://www.epa.gov/owmitnet/cgp.htm]

COMBINED SEWER OVERFLOWS (CSOS)

Combined sewer overflows, or CSOs, are a significant water pollution and public health threat. EPA's 1994 CSO Control Policy addresses CSOs in a flexible, cost-effective manner that provides for local decision-making and negotiation to achieve compliance with the Clean Water Act. CSOs contain not only storm water but also untreated human and industrial waste, toxic materials, and debris. This is a major water pollution concern for cities with combined sewer systems. CSOs are among the major sources responsible for beach closings, shellfishing restrictions, and other water body impairments. During dry weather, these "combined sewer systems" transport wastewater directly to sewage treatment plants. In periods of rainfall or snowmelt, however, the wastewater volume in a combined sewer system can exceed the capacity of the sewer system or treatment plant. For this reason, combined sewer systems are designed to overflow occasionally and discharge excess wastewater directly to nearby streams, rivers, lakes, or estuaries.

EPA's CSO Control Policy published April 19, 1994, is a national framework for control of CSOs through the NPDES permitting program. The Policy resulted from negotiations among municipal organizations, environmental groups, and State agencies. It provides guidance to municipalities and State and Federal permitting authorities on meeting pollution control goals of the Clean Water Act in a flexible, cost-effective manner. Information on EPA's CSO Control Policy can be found on the following Website. Source: [http://www.epa.gov/OWM/cso.htm]

STORMWATER PLANNING

The Watershed Management Institute, Inc. recently published a new manual entitled *Operation, Maintenance, and Management of Stormwater Management Systems* (1998). This manual presents a comprehensive review of the technical, educational, and institutional elements needed to assure that stormwater management systems are designed, built, maintained and operated properly during and after their construction. The manual was developed in cooperation with the U.S. EPA Office of Water to assist individuals responsible for designing, building, maintaining, or operating stormwater management systems. It will also be helpful to individuals responsible for implementing urban stormwater management programs.

The book includes fact sheets on 13 common stormwater treatment best management practices (BMPs). These summarize operation, maintenance, and management needs and obligations, along with construction recommendations. Other chapters review planning and design considerations, programmatic and regulatory aspects, considerations for facility owners, construction inspection, inspection and maintenance after construction, costs and financing, and disposal of stormwater sediments. Forms for inspecting BMPs during construction and determining maintenance needs afterwards are included in the book and in a separate supplement.
Source: [http://www.epa.gov/owowwtr1/NPS/wmi/index.html]
Additional information: [http://www.epa.gov/owowwtr1/NPS/ordinance/osm6.htm] and [http://www.epa.gov/owowwtr1/info/NewsNotes/issue05/nps05sto.html]

TOTAL MAXIMUM DAILY LOAD

States, territories, and authorized Tribes establish section 303(d) lists of impaired waters based on information contained in their 305(b) reports as well as other relevant and available water quality data. The section 303(d) list is a prioritized list of waters not meeting water quality standards. The USEPA has 30 days in which to approve the lists or add waters to the State's lists, if the Agency determines the list is not complete. Once a waterbody is placed on the 303(d) list, a TMDL must be prepared for the system.

A TMDL is a written, quantitative plan and analysis for attaining and maintaining water quality standards in all seasons for a specific waterbody and pollutant. Specifically, a TMDL is the sum of the allowable loads of a pollutant from all contributing point, nonpoint, and background sources. Total maximum daily loads may be established on a coordinated basis for a group of waterbodies in a watershed. Total maximum daily loads must be established for waterbodies on the list of impaired waterbodies and must include the following 11 elements:

1. The name and geographic location of the impaired waterbody;
2. Identification of the pollutant and the applicable water quality standard;
3. Quantification of the pollutant load that may be present in the waterbody and still ensure attainment and maintenance of water quality standards;
4. Quantification of the amount or degree by which the current pollutant load in the waterbody, including the pollutant load from upstream sources that is being accounted for as background loading, deviated from the pollutant load needed to attain and maintain water quality standards;
5. Identification of source categories, source subcategories or individual sources of pollutant;
6. Wasteload allocations;
7. Load allocations;
8. A margin of safety;
9. Consideration of seasonal variations;
10. Allowance for reasonably foreseeable increases in pollutant loads including future growth; and
11. An implementation plan.

Both the 1996 and 1998 section 303(d) lists, as well as more recent 305(b) reports reflect similar patterns: sediments, nutrients, and pathogens are the top three causes of waterbody impairment.
Source: [http://www.epa.gov/owowwtr1/tmdl/faq.html]

Waste Load Allocation
A waste load allocation (WLA) is the proportion of a receiving water's total maximum daily load that is allocated to point sources of pollution. Water quality models are often utilized by regulatory agencies in conducting an assessment to determine a WLA. Models establish a quantitative relationship between a waste load and its impact on water quality. WLAs are used by permit writers to establish Water Quality Based Effluent Limits (WQBELs).
Source: [http://www.epa.gov:80/owmitnet/permits/pwcourse/chapt_06.pdf]

Continuing Planning Process (CPP)
Each State is required to establish and maintain a continuing planning process (CPP) as described in section 303(e) of the Clean Water Act. A State's CPP contains, among other items, a description of the process that the State uses to identify waters needing water quality-based controls, a priority ranking of these waters, the process for developing TMDLs, and a description of the process used to receive public review of each TMDL. Descriptions may be as detailed as the Regional office and the State determine is necessary to describe each step of the TMDL development process. This process may be included as part of the EPA/State Agreement for TMDL development.
[Source: http://www.epa.gov/owowwtr1/tmdl/decisions/dec4.html]

LOOK TO THE FUTURE ... POLLUTANT TRADING

Point and nonpoint source pollutant trading involves financing reductions in nonpoint source pollution in lieu of undertaking more expensive point source pollution reduction efforts. A trading program is intended to produce cost savings for point source dischargers while improving water quality. Implementing a trading program requires a waterbody identifiable as a watershed or segment, as well as a measurable combination of point sources and controllable nonpoint sources. There must be significant load reductions for which the cost per pound reduced for nonpoint source controls is lower than the cost for upgrading point source controls. Lastly, point source dischargers must face requirements to either upgrade facility treatment capabilities or trade for nonpoint source reductions in order to meet water quality goals.

Such a program allows the private sector to allocate its resources to reduce pollutants in the most cost-effective manner, and it encourages the development of a watershed-wide or basin-wide approach to water quality protection. A pollutant trading program also requires cooperation between agencies, and requires a system to arrive at trading ratios between point and nonpoint source controls.

For example, in a North Carolina watershed, the Tar-Pamlico Basin Association (a coalition of point source dischargers) and State and regional environmental groups have proposed a two-phased nutrient management strategy that incorporates point and nonpoint source pollutant trading. The plan requires association members to finance nonpoint source reduction activities in the basin if their nutrient discharges exceed a base allowance.
Source: [http://www.epa.gov/OWOW/NPS/MMGI/funding.html#9]

8.4 MANAGING NONPOINT SOURCE POLLUTION

During the first 15 years of the national program to abate and control water pollution, EPA and the States have focused most of their water pollution control activities on traditional "point sources," such as discharges through pipes from sewage treatment plants and industrial facilities. These point sources have been regulated by EPA and the States through the NPDES permit program established by section 402 of the Clean Water Act. Discharges of dredged and fill materials into wetlands have also been regulated by the U.S. Army Corps of Engineers and EPA under section 404 of the Clean Water Act.

The Nation has greatly reduced pollutant loads from point source discharges and has made considerable progress in restoring and maintaining water quality as a result of the above activities. However, the gains in controlling point sources have not solved all of the Nation's water quality problems. Recent studies and surveys by EPA and by State/Tribal water quality agencies indicate that the majority of the remaining water quality impairments in our nation's rivers, streams, lakes, estuaries, coastal waters, and wetlands result from nonpoint source pollution and other nontraditional sources, such as urban storm water discharges and combined sewer overflows.

Nonpoint source pollution generally results from land runoff, precipitation, atmospheric deposition, drainage, seepage, or hydrologic modification. Technically, the term "nonpoint source" is defined to mean any source of water pollution that does not meet the legal definition of "point source" in section 502(14) of the Clean Water Act, defined in the preceding section. Although diffuse runoff is generally treated as nonpoint source pollution, runoff that enters and is discharged from conveyances such as those described above is treated as a point source discharge and hence is subject to the permit requirements of the Clean Water Act. In contrast, nonpoint sources are not subject to Federal permit requirements.

The pollution of waters by nonpoint sources is caused by rainfall or snowmelt moving over and through the ground. As the runoff moves, it picks up and carries away natural pollutants and pollutants resulting from human activity, finally depositing them into lakes, rivers, wetlands, coastal waters, and ground waters. Nonpoint source pollution can also be caused by atmospheric deposition of pollutants onto waterbodies. Furthermore, hydrologic modification is a form of nonpoint source pollution that often adversely affects the biological and physical integrity of surface waters. A more detailed discussion of the range of nonpoint sources and their effects on water quality and riparian habitats is provided in subsequent chapters of this guidance. A summary of State laws related to nonpoint source pollution can be found in the *Almanac of Enforceable State Laws to Control Nonpoint Source Water Pollution* (ELI 1988). This report can be accessed on the internet at http://www.eli.org/bookstore/research.htm.

NONPOINT SOURCES OF NUTRIENTS

Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters (USEPA 1993a) was developed by EPA for the planning and implementation of Coastal Nonpoint Pollution Programs. The guidance focuses on controlling five major categories of nonpoint sources that impair or threaten waters nationally. Management measures are specified for (1) agricultural runoff; (2) urban runoff (including developing and developed areas); (3) silvicultural (forestry) runoff; (4) marinas and recreational boating; and (5) hydromodification (e.g., channelization and channel modification, dams, and streambank and shoreline erosion). EPA guidance also includes management measures for wetlands, riparian areas, and vegetated treatment systems that apply generally to various categories of sources of nonpoint pollution. Management measures are defined in the Coastal Zone Act Reauthorization Amendments of 1990 as economically achievable measures to control the addition of pollutants to waters, which reflect the greatest degree of pollutant reduction achievable through the application of the best available nonpoint pollution control practices, technologies, processes, siting criteria, operating methods, or other alternatives.

The following section outlines some of the management measures specified in the CZARA guidance for the various types of nonpoint sources. These measures should be considered when implementing programs targeting nutrient releases into waters of the U.S.

Agricultural Runoff

- erosion and sediment control
- control of facility wastewater and runoff from confined animal facilities
- nutrient management planning on cropland
- grazing management systems
- irrigation water management

Urban Runoff

- control of runoff and erosion from existing and developing areas
- construction site runoff and erosion control
- construction site chemical control (includes fertilizers)
- proper design, location, installation, operation, and maintenance of on-site disposal systems
- pollution prevention education (e.g., household chemicals, lawn and garden activities, golf courses, pet waste, on-site disposal systems, etc.)
- planning, siting, and developing roads, highways, and bridges (including runoff management)

Silvicultural Runoff

- streamside management
- road construction and management
- forest chemical management (includes fertilizers)

- revegetation
- preharvest planning, harvesting management

Marinas and Recreational Boating
- siting and design
- operation and maintenance
- storm water runoff management
- sewage facility management
- fish waste management
- pollution prevention education (e.g., proper boat cleaning, fish waste disposal, and sewage pump out procedures)

Hydromodification (i.e., channelization, channel modification, dams)
- minimize changes in sediment supply and pollutant delivery rates through careful planning and design
- erosion and sediment control
- chemical and pollutant control (includes nutrients)
- stabilization and protection of eroding streambanks or shorelines

Wetlands, Riparian Areas, Vegetated Treatment Systems
- protect the NPS abatement and other functions of wetlands and riparian areas through vegetative composition and cover, hydrology of surface and ground water, geochemistry of the substrate, and species composition
- promote restoration of preexisting function of damaged and destroyed wetlands and riparian systems
- promote the use of engineered vegetated treatment systems if they can serve a NPS pollution abatement function

EFFORTS TO CONTROL NONPOINT SOURCE POLLUTION

Efforts to control nonpoint source pollution include nonpoint source management programs, the National Estuary Program, atmospheric deposition, coastal nonpoint pollution control programs, and Farm Bill conservation provisions. These efforts are described below.

Nonpoint Source Management Programs
In 1987, in view of the progress achieved in controlling point sources and the growing national awareness of the increasingly dominant influence of nonpoint source pollution on water quality, Congress amended the Clean Water Act to focus greater national efforts on nonpoint sources. In the Water Quality Act of 1987, Congress amended section 101, "Declaration of Goals and Policy," to add the following fundamental principle:

> It is the national policy that programs for the control of nonpoint sources of pollution be developed and implemented in an expeditious manner so as to enable the goals of this Act to be met through the control of both point and nonpoint sources of pollution.

More importantly, Congress enacted section 319 of the Clean Water Act, which established a national program to control nonpoint sources of water pollution. Under section 319, States address nonpoint pollution by assessing nonpoint source pollution problems and causes within the State, adopting management programs to control the nonpoint source pollution, and implementing the management programs. While not required, many States have incorporated the management measures specified in the 1993 CZARA guidance into their State Nonpoint Source Management Programs.

Section 319 also authorizes EPA to issue grants to States to assist them in implementing those management programs or portions of management programs which have been approved by EPA. As of FY 2000, over $1 billion in grants have been given to States, Territories, and Tribes for the implementation of nonpoint source pollution control programs.

For additional information on the Nonpoint Source Management Program and distribution of Section 319 grants in your State, contact your State's designated nonpoint source agency. For many states, the nonpoint source agency is the State Water Quality Agency. However, in several instances, other agencies or departments are given nonpoint source responsibility (see Table 5).

National Estuary Program

EPA also administers the National Estuary Program under section 320 of the Clean Water Act. This program focuses on point and nonpoint pollution in geographically targeted, high-priority estuarine waters. Under this program, EPA assists State, regional, and local governments in developing comprehensive conservation and management plans that recommend priority corrective actions to restore estuarine water quality, fish populations, and other designated uses of the waters. For additional information, contact your local estuary program. The following estuaries are currently enrolled in the program:

- Albemarle-Pamlico Sounds, NC
- Barataria-Terrebonne Estuarine Complex, LA
- Barnegat Bay, NJ
- Buzzards Bay, MA
- Casco Bay, ME
- Charlotte Harbor, FL
- (Lower) Columbia River Estuary, OR and WA
- Corpus Christi Bay, TX
- Delaware Estuary, DE, NJ, and PA
- Delaware Inland Bays, DE
- Galveston Bay, TX
- Indian River Lagoon, FL
- Long Island Sound, NY and CT
- Maryland Coastal Bays, MD

Table 5. States for which the nonpoint source agency is not the water quality agency.

State	State Nonpoint Source Agency
Arkansas	State Department of Soil and Water Conservation
Delaware	State Department of Soil and Water Conservation
Oklahoma	State Department of Soil and Water Conservation
Tennessee	State Department of Agriculture
Texas	Department of Soil and Water Conservation (for agriculture) Texas Water Quality Board (all other nonpoint sources)
Vermont	State Department of Agriculture
Virginia	State Department of Soil and Water Conservation

- Massachusetts Bays, MA
- Mobile Bay, AL
- Morro Bay, CA
- Narragansett Bay, RI
- New Hampshire Estuaries, NH
- New York-New Jersey Harbor, NY and NJ
- Peconic Bay, NY
- Puget Sound, WA
- San Francisco Estuary, CA
- San Juan Bay, PR
- Santa Monica Bay, CA
- Sarasota Bay, FL
- Tampa Bay, FL
- Tillamook Bay, OR

Atmospheric Deposition

While runoff from agricultural and urban areas may be the largest sources of nonpoint pollution, growing evidence suggests that atmospheric deposition may have a significant influence on nutrient enrichment, particularly from nitrogen (Jaworski et al. 1997). Gases released through fossil fuel combustion and agricultural practices are two major sources of atmospheric N that may be deposited in waterbodies (Carpenter et al. 1998). Nitrogen and nitrogen compounds formed in the atmosphere return to the earth as acid rain or snow, gas, or dry particles (http://www.epa.gov/acidrain/effects/envben.html). EPA has several programs that address the issue of atmospheric deposition, including the National Ambient Air Quality Standards, the Atmospheric Deposition Initiative, and the Great Waters Program.

National Ambient Air Quality Standards
The Clean Air Act provides the principal framework for national, State, and local efforts to protect air quality. Under the Clean Air Act, national ambient air quality standards (NAAQS) for pollutants which are considered harmful to people and the environment are established.

The Clean Air Act established two types of national air quality standards. Primary standards set limits to protect public health, including the health of "sensitive" populations such as asthmatics, children, and the elderly. Secondary standards set limits to protect public welfare, including protection against decreased visibility, damage to animals, crops, vegetation, and buildings (http://www.epa.gov/airs/criteria.html).

Atmospheric Deposition Initiative
In 1995, EPA's Office of Water established an "Air Deposition Initiative" to work with the EPA Office of Air and Radiation to identify and characterize air deposition problems with greater certainty and examine solutions to address them. The Air and Water Programs are cooperating to assess the atmospheric deposition problem, conduct scientific research, provide innovative solutions to link Clean Air Act and Clean Water Act tools to reduce the of these pollutants, and communicate the findings to the public. To date, most efforts have focused on better understanding of the links between nitrogen and mercury emissions and harmful effects on water quality and the environment. Significant work has also been done towards quantifying the benefits to water quality of reducing air emissions and developing sensible, cost effective approaches to reducing the emissions and their ecosystem and health effects (http://www.epa.gov/owowwtr1/oceans/airdep/index.html).

Great Waters Program
On November 15, 1990, in response to mounting evidence that air pollution contributes to water pollution, Congress amended the Clean Air Act and included provisions that established research and reporting requirements related to the deposition of hazardous air pollutants to the "Great Waters." The waterbodies designated by these provisions are the Great Lakes, Lake Champlain, and Chesapeake Bay. As part of the Great Waters Program, Congress requires EPA, in cooperation with the National Oceanic and Atmospheric Administration, to monitor hazardous pollutants by establishing sampling networks, investigate the deposition of these pollutants, improve monitoring methods, monitor for hazardous pollutants in fish and wildlife, determine the contribution of air pollution to total pollution in the Great Waters, evaluate any adverse effects to public health and the environment, determine sources of pollution, and provide a report to Congress every 2 years. These reports provide an information base that can be used to establish whether air pollution is a significant contributor to water quality problems of the Great Waters, determine whether there are significant adverse effects to humans or the environment, evaluate the effectiveness of existing regulatory programs in addressing these problems, and assess whether additional regulatory actions are needed to reduce atmospheric deposition to the Great Waters. For more detail, the Great Waters biennial Reports to Congress discuss current scientific understanding of atmospheric deposition (http://www.epa.gov/airprogm/oar/oaqps/gr8water/xbrochure/program.html).

Coastal Nonpoint Pollution Control Programs
In November 1990, Congress enacted the Coastal Zone Act Reauthorization Amendments of 1990. These Amendments were intended to address several concerns, a major one of which is the impact of nonpoint source pollution on coastal waters.

To address more specifically the impacts of nonpoint source pollution on coastal water quality, Congress enacted section 6217, "Protecting Coastal Waters," which was codified as 16 U.S.C. -1455b. This section provides that each State with an approved coastal zone management program must develop and submit a Coastal Nonpoint Pollution Control Program for EPA and the National Oceanic and Atmospheric Administration (NOAA) approval. The purpose of the program "shall be to develop and implement management measures for nonpoint source pollution to restore and protect coastal waters, working in close conjunction with other State and local authorities."

States with Coast Nonpoint Pollution Control Programs are required to include measures in their programs that are “in conformity” with the 1993 CZARA guidance, as discussed previously. A listing of States with Coastal Nonpoint Pollution Control Programs is presented in Table 6. For additional information on the programs in these States, contact the State water quality agency.

Farm Bill Conservation Provisions

Technical and financial assistance for landowners seeking to preserve soil and other natural resources is authorized by the Federal Government under provisions of the Food Security Act (Farm Bill). Provisions of the 1996 Farm Bill relating directly to installation and maintenance of BMPs are summarized in the following sections. Contact your Natural Resources Conservation Service (NRCS) State Conservationist’s office for State-specific information.

<u>Environmental Conservation Acreage Reserve Program (ECARP)</u>

ECARP is an umbrella program established by the 1996 Farm Bill which contains the conservation Reserve Program (CRP), Wetlands Reserve Program (WRP), and Environmental Quality Incentives Program (EQIP). It authorizes the Secretary of Agriculture to designate watersheds, multi-state areas, or regions of special environmental sensitivity as conservation priority areas which are eligible for enhanced Federal assistance. Assistance in priority areas is to be used to help agricultural producers comply with NPS pollution requirements of the Clean Water Act and other State or Federal environmental laws. The ECARP is authorized through 2002.

<u>Conservation Reserve Program (CRP)</u>

First authorized by the Food Security Act of 1985 (Farm Bill), this voluntary program offers annual rental payments, incentive payments, and cost-share assistance for establishing long-term, resource-conserving cover crops on highly erodible land. CRP contracts are issued for a duration of 10 to 15 years for up to 36.4 million acres of cropland and marginal pasture. Land can be accepted into the CRP through a competitive bidding process through which all offers are ranked using an environmental benefits index, or through continuous sign-up for eligible lands where certain special conservation practices will be implemented.

The Conservation Reserve Enhancement Program (CREP) is a new initiative of CRP authorized under the 1996 Federal Agricultural Improvement and Reform Act. CREP is a joint, State-federal program designed to meet specific conservation objectives. CREP targets State and Federal funds to achieve shared environmental goals of national and state significance. The program uses financial incentives to encourage farmers and ranchers to voluntarily protect soil, water, and wildlife resources.

Table 6. An alphabetical list of States and Territories with Coastal Nonpoint Pollution Control Programs.

States and Territories with Coastal Nonpoint Pollution Control Programs		
Alabama	Maine	Oregon
Alaska	Maryland	Pennsylvania
American Samoa	Massachusetts	Puerto Rico
California	Michigan	Rhode Island
Connecticut	Mississippi	South Carolina
Delaware	New Hampshire	Virgin Islands
Florida	New Jersey	Virginia
Guam	New York	Washington
Hawaii	North Carolina	Wisconsin
Louisiana	Northern Mariana Islands	

Wetlands Reserve Program (WRP)
The WRP is a voluntary program to restore and protect wetlands and associated lands. Participants may sell a permanent or 30-year conservation easement or enter into a 10-year cost-share agreement with USDA to restore and protect wetlands. The landowner voluntarily limits future use of the land, yet retains private ownership. The NRCS provides technical assistance in developing a plan for restoration and maintenance of the land. The landowner retains the right to control access to the land and may lease the land for hunting, fishing, and other undeveloped recreational activities.

Environmental Quality Incentives Program
The EQIP was established by the 1996 Farm Bill to provide a voluntary conservation program for farmers and ranchers who face serious threats to soil, water, and related natural resources. EQIP offers financial, technical, and educational help to install or implement structural, vegetative, and management practices designed to conserve soil and other natural resources. Current priorities for these funds dictate that one half of the available monies be directed to livestock-related concerns. Cost-sharing may pay up to 75% of the costs for certain conservation practices. Incentive payments may be made to encourage producers to perform land management practices such as nutrient management, manure management, integrated pest management, irrigation water management, and wildlife habitat management.

Wildlife Habitat Incentives Program (WHIP)
This program is designed for parties interested in developing and improving wildlife habitat on private lands. Plans are developed in consultation with NRCS and the local Conservation District. USDA will provide technical assistance and cost-share up to 75% of the cost of implementing the wildlife

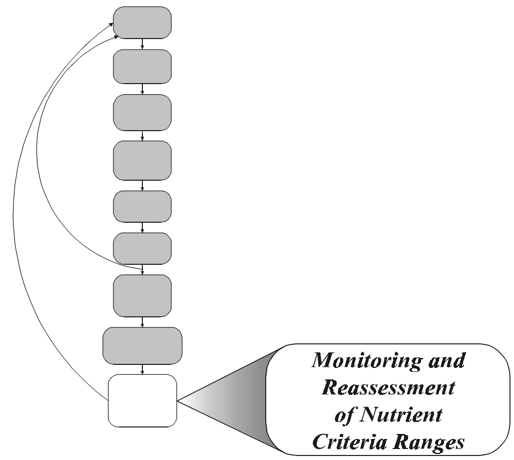
conservation practices. Participants generally must sign a 5- to 10-year contract with USDA which requires that they maintain the improvement practices.

<u>Forestry Incentives Program (FIP)</u>
Originally authorized in 1978, the FIP allows cost sharing of up to 65% (up to a maximum of $10,000 per person per year) for tree planting, timber stand improvement, and related practices on nonindustrial private forest land. The FIP is administered by NRCS and the U.S. Forest Service. Cost share funds are restricted to individuals who own no more than 1,000 acres of eligible forest land.

<u>Conservation of Private Grazing Land</u>
This program was authorized by the 1996 Farm Bill for the purpose of providing technical and educational assistance to owners of private grazing lands. It offers opportunities for better land management, erosion reduction, water conservation, wildlife habitat, and improving soil structure.

<u>Cooperative Extension</u>
State land grant universities and Cooperative Extension play an important role in management implementation. They have the expertise to research, transfer, and implement agriculture management systems that will be needed to meet nutrient criteria. In addition, they have developed models and other predictive management tools that will aid in selecting the most appropriate management activities. Contact your local Cooperative Extension Agent, or the Agriculture Department at a State land grant university for more information on the services they can provide.

Chapter 9. Monitoring and Reassessment of Nutrient Criteria Ranges



9.1 INTRODUCTION

After criteria are set, compliance determinations made, and management plans implemented, resource managers should continue to monitor river and stream systems while reassessing goals and nutrient criteria. This step should (1) evaluate the appropriateness of established 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.

Those streams selected for management may be approached using a rational course of action beginning with a statement of major problems or symptoms and progressing logically to a final assessment to determine the relative success of the effort. Throughout this process, the water quality manager must re-examine (1) the initial goals identified for the stream system(s) prior to criteria development and (2) subsequent management actions taken to evaluate the effectiveness of criteria and management plans. The manager should assess the efficacy of management actions and potentially re-evaluate the appropriateness of established criteria if monitoring data indicate that goals are not being met.

9.2 ASSESSMENT OF PROCESS THROUGH MONITORING AND PERIODIC REVIEW

The management plan should always include "before," "during," and "after" water resource quality monitoring to demonstrate the relative response of the system to management efforts, thus the recommendation that initial survey stations should generally be maintained and expanded. Availability of continuous, year-to-year monitoring data is critical and can be used as a bench mark for evaluating progress. If monitoring data indicate that water quality is improved, monitoring should continue to validate the progress made. Should water quality decline, the criteria development process should be re-visited and potentially revised. At a minimum, monitoring data should be reevaluated every five years to gauge progress. The reevaluation of monitoring data should include seasonality and periodic data assessment intervals for management review to provide the opportunity for responses to changing circumstances, modifications of methods, schedules, and changes of emphasis as needed. Control of point source nutrients may result in fairly quick system recovery from cultural eutrophication (Edmonson

1994), although nutrient cycling mechanisms and changes in food web dynamics may result in a persistent eutrophic state in many systems (Carpenter et al. 1999). Therefore, continued monitoring and reevaluation of nutrient control strategies is of particular importance.

9.3 COMPLETION AND EVALUATION

Management projects are frequently planned, initiated, and concluded with new initiatives undertaken to meet pressing schedules without sufficiently evaluating what was or was not initially accomplished. Review of progress, original objectives or goals, and monitoring data will reveal whether the river or stream trophic state was successfully protected or improved. Just as important, this evaluation will provide the documentation necessary to determine if methods and techniques attempted in this instance can be applied, perhaps with modification, elsewhere. Alternatively, it will also reveal if mistakes were made which should be noted and avoided in future projects and if perhaps a sequel to the current project is required to fully accomplish that which was intended.

9.4 CONTINUED MONITORING OF THE SYSTEM

Monitoring programs initiated and expanded in the course of the project can now be reduced to the periodic measuring of key variables at critical times and locations. At this stage, the purpose of monitoring is to keep sufficiently informed of the status of the river or stream to ensure that the protection or remediation achieved is maintained. Intervention should be possible at an early point to minimize the costs of remediation if periodic maintenance monitoring indicates a return of trophic decline. The evaluation and periodic monitoring steps of this process essentially close the loop. If new issues arise, the manager returns to step one with a new problem statement.

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APPENDIX A. NUTRIENT CRITERIA CASE STUDIES

The following five case studies are meant to capture some of the variability of stream systems located throughout the country. Although these case studies exhibit varying levels of complexity, they are meant to provide the reader with real-world examples of how criteria can be developed on a practical level and several region-specific issues that may be encountered as one works through the criteria development process. The ecoregional nutrient criteria process discussed in the Tennessee case study involves refinement of the Level III ecoregions found within the State; identification and monitoring of reference stream systems; and correlational analyses of nutrient levels, conventional water chemistry parameters, and biological indices to derive criteria. In contrast, the Clark Fork, Montana, case study delineates a process for setting target nutrient and algal levels based on a combination of modified established criteria, literature values, and observed thresholds for nuisance algal growth. The Upper Midwest river systems case study describes the results of a cooperative effort among three USGS NAWQA projects in the upper Midwest Corn Belt region that evaluated algal and macroinvertebrate response to nonpoint agricultural sources relative to naturally-occurring factors (e.g., riparian vegetation, hydrology). The Bow River, Canada, case study details the reduction of nuisance biomass (both periphyton and macrophytes) over a 16-year period through decreases in nitrogen (~50%) and phosphorus (80%) from domestic wastewater effluent. Finally, the desert stream case study discusses several of the determinants of nutrient regimes in desert streams that should be considered when developing nutrient criteria for these, as well as other, complex, highly variable stream systems.

TENNESSEE ECOREGIONAL NUTRIENT CRITERIA

In 1992, the Tennessee DWPC (Division of Water Pollution Control) faced an important decision on how water quality assessment would be done in the future. When program status was assessed, there were problems that were likely to be amplified in the future. For example:

- The “one-size-fits-all” statewide numeric criteria approach provided stability, but lacked regional flexibility. Statewide criteria were clearly overprotective in parts of the state, but arguably underprotective in other areas.

- Narrative criteria were based on a verbal description of water quality, rather than a number. Thus, they provided flexibility, but lacked an objective means of interpretation. As an example, the narrative criterion for biological integrity states “*waters shall not be modified to the extent that the diversity and/or productivity of aquatic biota within the receiving waters is substantially reduced*”. However, an interpretation of the word “substantially” was not provided.

- Unlike biological integrity, nutrients did not have specific narrative criteria. Nutrients were assessed under the more generic “free from” statements found in toxicity sections of the fish and aquatic life criteria and under “aesthetic” sections of the recreational criteria. Thus, before any stream could be assessed as impacted by nutrients, the existence of a “problem” had to be established.

- Tennessee was encouraged by EPA to convert to a watershed approach for issuance of water quality permits. Without a sense of regional variability in water quality, there was a distinct disadvantage in goal setting for these watersheds. Additionally, the rigors of 303(d) listing and TMDL development required accurate interpretation of Tennessee’s narrative water quality criteria. The specter of lawsuits by citizens and members of the regulated community required that assessments be defensible.

A method was needed for comparing the existing conditions found in a stream to unimpacted conditions. This reference condition varied across the state. The reference condition established should be within a similar area, to avoid “apples and oranges” comparisons. It was determined that *ecoregions* were the best geographic basis upon which to make this assessment.

> *An ecoregion is a relatively homogeneous area defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, and other ecologically relevant variables.*

The “Ecoregions of the United States” map (Level III) developed in 1986 by James Omernik of EPA's Corvallis Laboratory delineated eight ecoregions in Tennessee. The DWPC arranged for Omernik and Glenn Griffith to sub-regionalize and update state ecoregions.

The Tennessee Ecoregion Project began in 1993 and was envisioned to occur in three phases:

PHASE I: DELINEATE SUB-ECOREGION BOUNDARIES

Phase I of the project involved geographic data gathering, development of a draft sub-regionalization scheme, and ground-truthing of the draft into a final product. This product included new maps and digitized coverages for use in the DWPC GIS system. This part of the project began in 1993 and was completed in 1995. This refinement resulted in a total of 14 ecoregions for the state (Figure A-1).

PHASE II: REFERENCE STREAM SELECTION

EPA and DWPC staff identified potential reference streams. Reference streams selected were located in relatively unimpacted watersheds typical for that ecoregion (Figure A-2). When possible, watersheds within state or federally protected areas were selected.

A reference stream is a least impacted waterbody within an ecoregion that can be monitored to establish a baseline to which other waters can be compared. Reference streams are not necessarily pristine or undisturbed by humans.

Division staff visited each candidate stream. Chemical and benthic macroinvertebrate samples were used to cull the list of streams down to a final list. Three reference streams per sub-ecoregion were considered the minimum requirement.

PHASE III: INTENSIVE MONITORING OF REFERENCE STREAMS

Since August 1996, final selected reference sites have been monitored quarterly. During the first year of the project, water chemistry was monitored using grab samples collected on three consecutive days (if possible). Chemical sampling procedures followed modified clean technique methodology as outlined in the Division's Chemical Standard Operating Procedure: Modified Clean Technique Sampling Protocol (TNDEC 1996).

Chemical sampling at reference sites generally included all the parameters historically included by the Division in its long-term ambient monitoring network. As a concession to resource constraints, certain parameters, such as mercury, were dropped after they were never detected the first year of sampling. Additional parameters such as chlorophyll *a* were considered to have value, but were not sampled due to the need make the best use of program funding. Division staff were recently trained in algal assessment techniques and will likely incorporate rapid biological assessment protocols in future sampling efforts.

Macroinvertebrate samples were collected at ecoregional reference sites beginning in August 1996. Habitat and flow were also assessed. Outside expertise was sought to analyze the monitoring data to determine how sub-ecoregions aggregate by aquatic habitat and biological community to form ecosystems or bioregions. This step was essential for assessing benthic communities accurately and consistently.

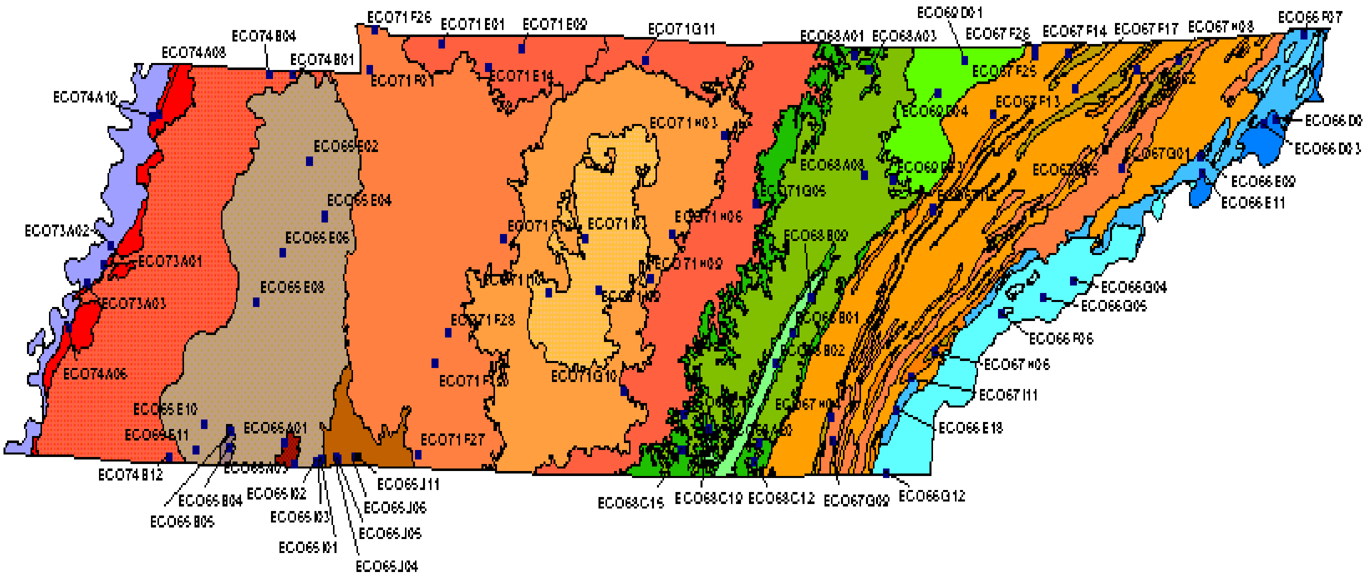

Figure A-1. Tennessee Level IV ecoregions and locations of reference streams.

Figure A-2. The Little River within the Great Smoky Mountains National Park was selected as a reference stream for sub-ecoregion 66g.

How Are Reference Stream Data Being Used?

For the first time, the DWPC has regionally-based chemical, physical, and biological data representing least impacted conditions in Tennessee. These data are important to our program and have multiple applications.

For some time, it was known that an ecoregion-specific approach to certain water quality standards would provide greater accuracy. This ecoregion project has provided the data necessary to initiate nutrient criteria discussions.

Figures A-3 and A-4 illustrate the levels of total phosphorus (TP) and nitrate-nitrite (NO_3-NO_2), respectively, documented at reference streams within each ecoregion. The box and whisker plot shows median measured concentrations and ranges. Based on the data collected, TP at less impacted streams is generally higher in West Tennessee than Middle and East Tennessee.

Finalizing the Ecoregion Reference Stream Nutrient Database

Additional steps are needed to finalize the ecoregion nutrient database:

- Incorporate data from other States. If reference streams in neighboring States are located within shared ecoregions and are selected and sampled in a similar manner to those in Tennessee, the nutrient data can be added into our database.

- Review the database for quality assurance. Data will be checked for outliers that may represent data entry errors. Outliers that indicate degrading conditions in reference streams will be identified. The Division considered eliminating outliers based on a consistent rationale, such as values more than two standard deviations from the mean, but decided against such an approach.

Development of Regional Interpretations of Narrative Nutrient Criteria

Division staff will propose ecoregion-specific interpretations of the narrative nutrient criteria for TP and nitrate-nitrite for the year 2000 triennial water quality standards review. These numeric goals will be used primarily for water quality assessment purposes.

The specific goals will likely be based on the establishment of the nutrient concentration for each ecoregion or subecoregions database at the 90th percentile of the reference stream data. (However, the Division has not ruled out the possibility of setting the criteria at the 75th percentile.) As an important part of the process, Division staff will statistically analyze nutrient levels and their ranges at each sub-ecoregion. Where significant differences exist between sub-ecoregions, the nutrient criteria will be established at the sub-ecoregion level. Where no significant difference is found between sub-ecoregions, the data will be aggregated back to the ecoregion level.

These numeric goals will provide the means to assess nutrient levels at similar streams within the same ecoregion. Streams with nutrient levels less than the 90th (or 75th) percentile of the reference stream database will be considered to meet the narrative criteria. Streams with nutrient levels higher than the reference stream database range will be considered in violation of the narrative criteria. These streams

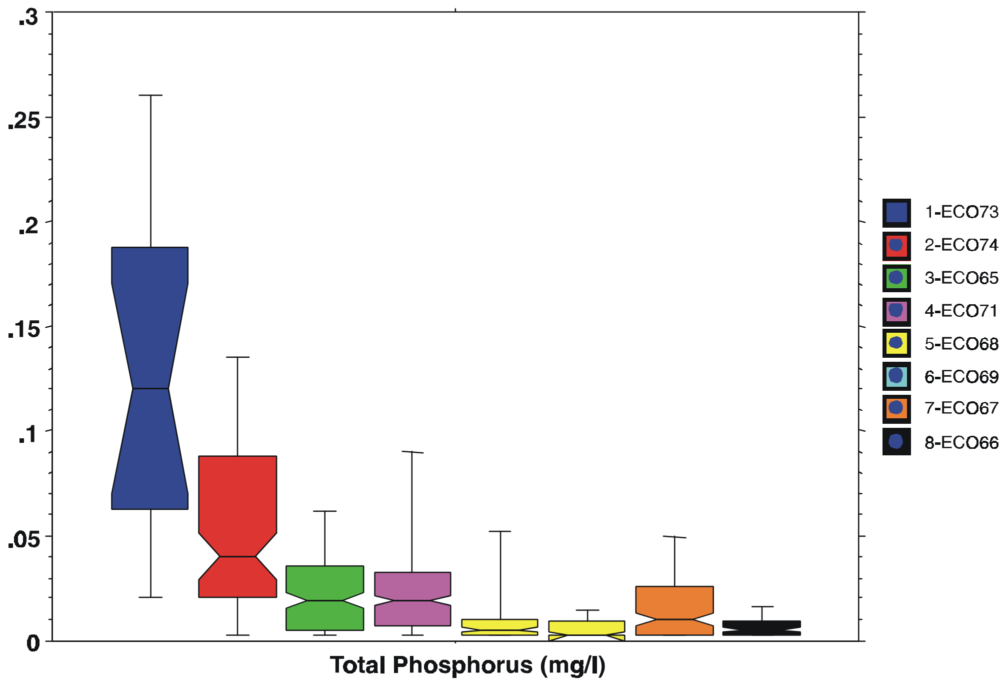

Figure A-3. Total phosphorus concentrations (μg/L) for reference streams within each ecoregion.

Key: 1 = Mississippi Alluvial Plain, 2 = Mississippi Valley Loess Plains, 3 = Southeastern Plains, 4 = Interior Plateau, 5 = Southeastern Appalachians, 6 = Central Appalachians, 7 = Ridge and Valley, 8 = Blue Ridge Mountains.

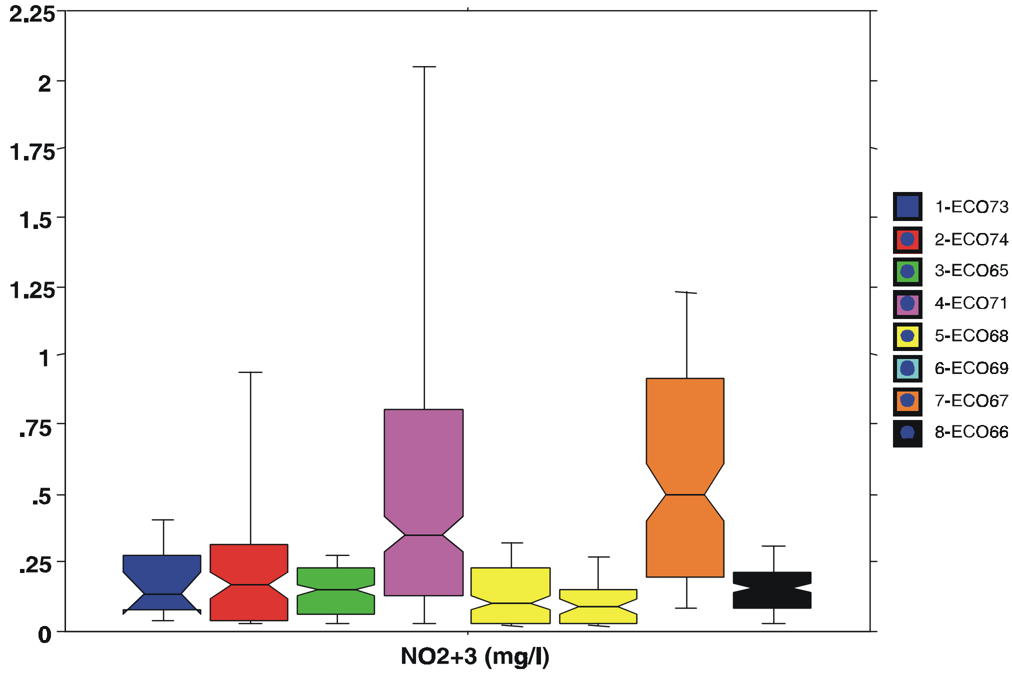

Figure A-4. Total nitrate-nitrite concentrations (mg/L) for reference streams within each ecoregion.

Key: 1 = Mississippi Alluvial Plain, 2 = Mississippi Valley Loess Plains, 3 = Southeastern Plains, 4 = Interior Plateau, 5 = Southeastern Appalachians, 6 = Central Appalachians, 7 = Ridge and Valley, 8 = Blue Ridge Mountains.

will be added to the 303(d) list for future TMDL generation. Additionally, the regional interpretation of the narrative criteria will provide the goal for TMDL control strategies.

Data Relationships

Division staff have taken a preliminary look at the reference stream data in an attempt to investigate relationships between sampled parameters. Examination of these relationships has three facets: (1) consideration of possible nutrient data surrogates, (2) exploring relationships between nutrient levels and biological indices, and (3) comparison of reference stream data to EPA's regional nutrient database.

1. The initial investigation was whether there was a relationship between nutrient levels and other chemical constituents in the water column. If a strong correlational relationship could be established, these other values could be used as data surrogates if nutrient data were unavailable or as a less costly substitute for nutrient sampling.

   Relationships were investigated primarily for turbidity, total organic carbon (TOC), and suspended solids. We found numerous positive correlations, but the large number of data points at the detection level caused relationships to be suspect. For example, Figures A-5 and A-6 illustrate the relationship between total phosphorus and turbidity (r^2 value = 0.282) as well as total phosphorus and TOC (r^2 value = 0.163) in ecoregion 67g.

   We intend to do the same type analysis with regional data from EPA's national nutrient database. At least in theory, this database would contain fewer observations below detection level.

2. If the correlation between either TP or nitrate+nitrite levels and the quality of biological communities can be established, a stronger rationale for ecoregion-specific numerical nutrient criteria can be provided. However, it should be noted that even where correlation is strong, identifying a numeric nutrient criteria is dependent on knowing the biological integrity score above which, the community is considered impaired. Fortunately, as in the case of nutrients, this biological integrity goal can be established from the reference stream data.

   In sub-ecoregion 71h (Outer Nashville Basin), a preliminary comparison was done. Nitrate-nitrite levels were compared to two biological indices frequently used by the Division, the North Carolina Biotic Index (NCBI) and the Hilsenhoff Biotic Index (EPA Rapid Bioassessment Protocols, 1999). While there was some scatter in the dataset, a relationship was suggested which was slightly stronger for the Hilsenhoff index (Figure A-7) than the NCBI. (Figure A-8).

   An additional test was done with the appearance of a relationship between nitrate-nitrite and NCBI scores. According to the reference stream database for sub-ecoregion 71h, the 75th percentile of the NCBI data is a score of approximately 5.0. Presuming that an NCBI score of 5.0 is the biological goal for sub-ecoregion 71h, then according to the above chart, nitrate-nitrite levels should not exceed approximately 1.2 mg/L. Following the same approach with the Hilsenhoff scores also produced a similar nitrate-nitrite level, approximately 1.2 mg/L. It is interesting to note that the 90th percentile of the reference stream nitrate-nitrite data for 71h is approximately 1.0 mg/L.

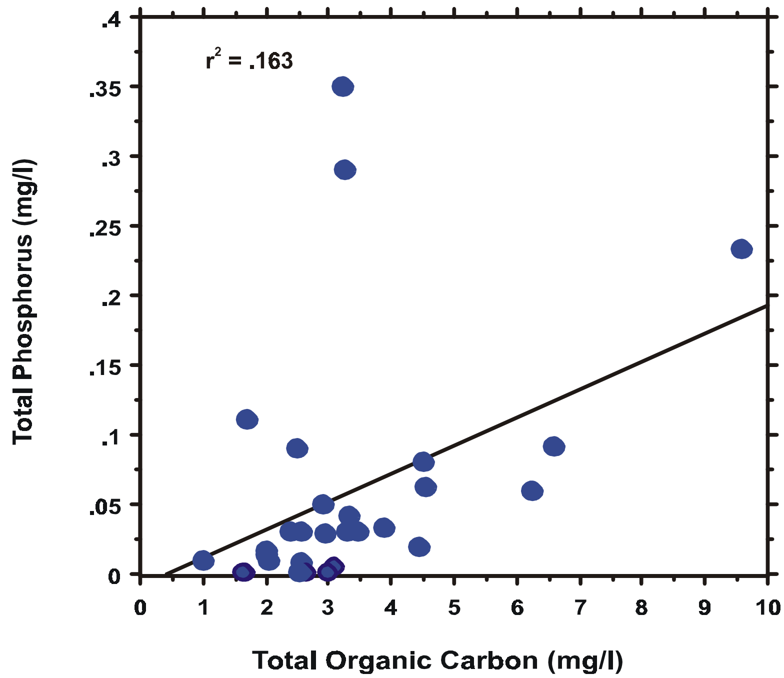

Figure A-5. Relationship between total phosphorus and TOC (r^2 value = 0.163) in ecoregion 67g.

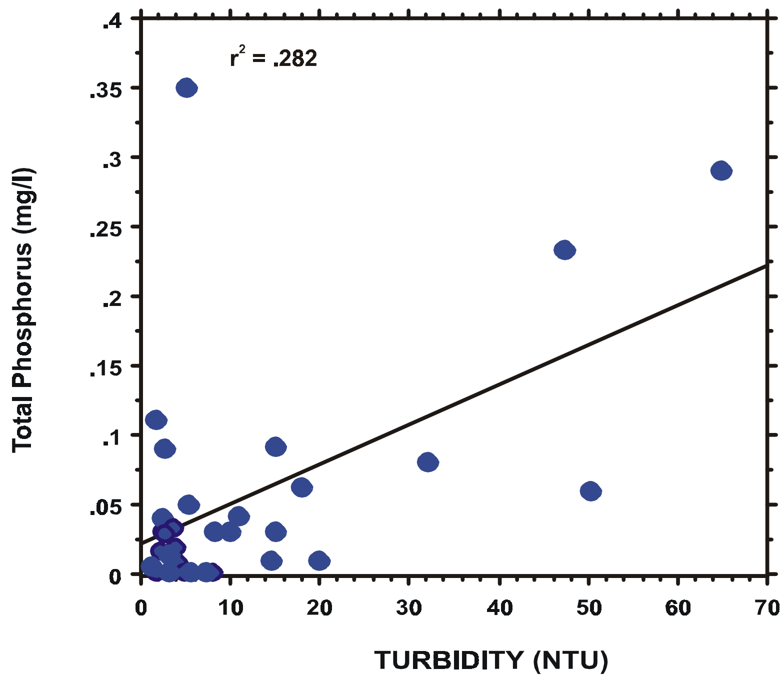

Figure A-6. Relationship between total phosphorus and turbidity (r^2 value = 0.282) in ecoregion 67g.

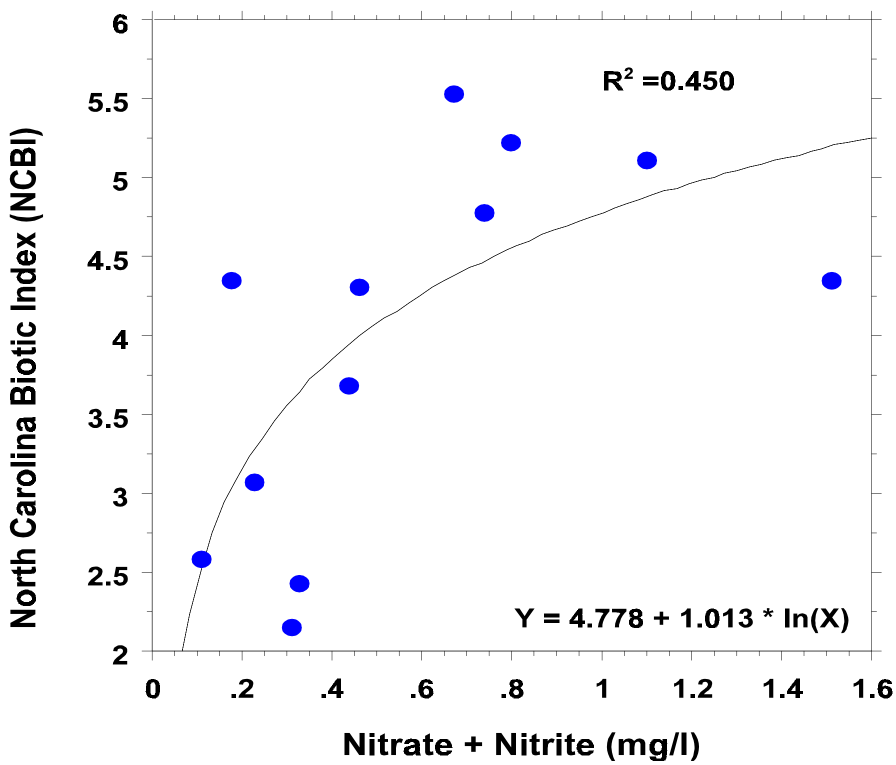

Figure A-7. Relationship between nitrate-nitrite levels and the Hilsenhoff Biotic Index.

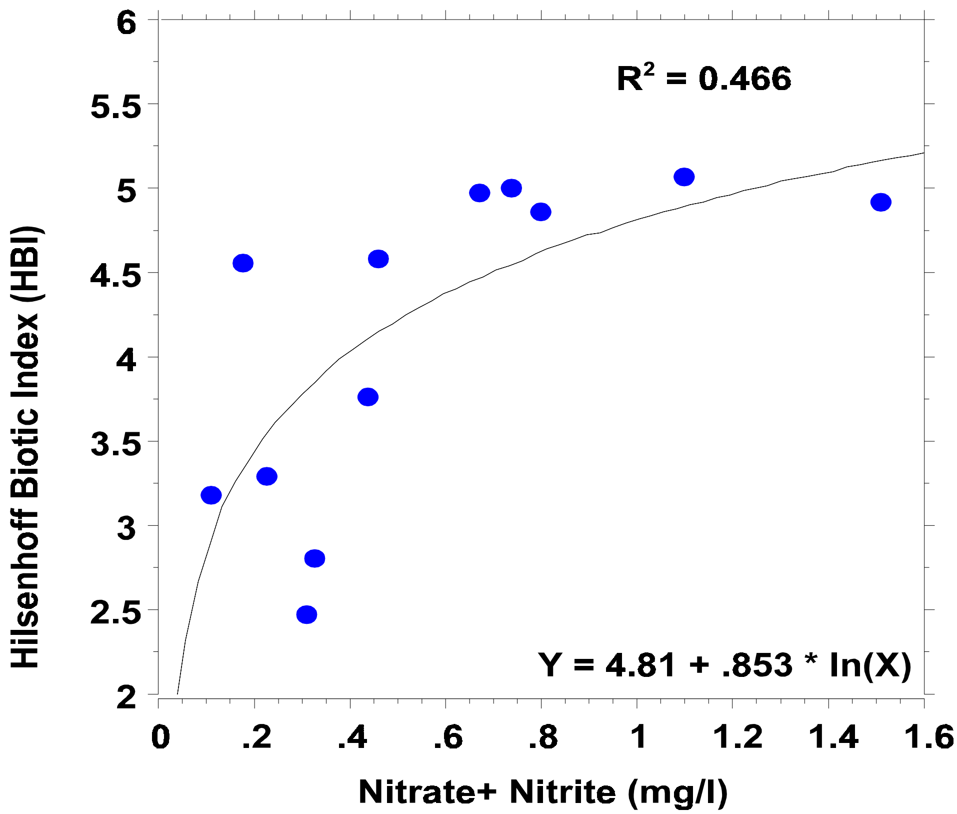

Figure A-8. Relationship between nitrate-nitrite levels and the North Carolina Biotic Index (NCBI).

While the two values, 1.2 and 1.0 mg/L, are not exactly the same, clearly these two methods of criteria development can be used to strengthen the rationale for a final criteria recommendation or to justify a "margin of safety". It also demonstrates that should the Division set the nitrate+nitrite goal for 71h at 1.0 mg/L, that level should generally be protective of biological integrity.

3. Another potential methodology for nutrient criteria development was examined. According to EPA draft guidance, the reference conditions may be compared to all other nutrient data to potentially provide a range for criteria selection. EPA suggests that the range is established by comparing the reference stream data at the 75^{th} percentile with the 25^{th} percentile of all other data. We were curious to see if this approach would work and if so, would it provide values similar to those we had already identified.

   To assist in this effort, EPA provided us with the nutrient databases from STORET for the three large nutrient regions in Tennessee. (For purposes of this initial test, only Tennessee STORET data were included.) Nutrient Region XI in east Tennessee is a combination of Level III ecoregions 66, 67, 68, and 69. Nutrient Ecoregion IX in middle Tennessee is composed of Ecoregions 71, 65, and 74. Ecoregion 73 in west Tennessee is Nutrient Ecoregion X.

   The EPA nutrient database was primarily data collected by the Division of Water Pollution Control, the Tennessee Valley Authority (TVA), and the U.S. Geological Survey (USGS). As we were familiar with TVA's monitoring program, we were concerned that some percentage of their data was from lakes or embayments. Since we were developing stream nutrient criteria, rather than lake or embayment criteria, we did not consider it appropriate to include non-stream data. Lacking the time to identify and cull only the embayment or lakes data from the database, we decided to exclude all TVA data.

   Figure A-9 illustrates a comparison of the National nutrient database for Nutrient Ecoregion Region XI and the reference stream database for the same geographic area. The 75^{th} percentile of the reference stream data and the 25^{th} percentile of the National Nutrient database lined up well for some ecoregions (68, 69, & 66), but not for the Central Appalachian Ridge and Valley Region (67).

   We also looked at EPA draft Nutrient Aggregate Ecoregion IX in West Tennessee (Figure A-10). Data for total phosphorus were elevated nearly an order of magnitude higher than the reference stream data. We discovered that a few stations provided a sizable number of data points within the database. It is possible that some of these data represent "storm chasing" sampling events designed to quantify worst case nutrient loadings. Another possibility is that sampling in the phosphorus-rich soils of ecoregion 71 biased the database. If we can identify these sites and determine that these data are not representative of the ambient water quality in the ecoregion, these data could be excluded and the database re-formed.

SUMMARY

With the assistance of EPA, the Tennessee Division of Water Pollution Control subdelineated ecoregions from Level III to Level IV. Reference streams were identified in each sub-ecoregion to establish a

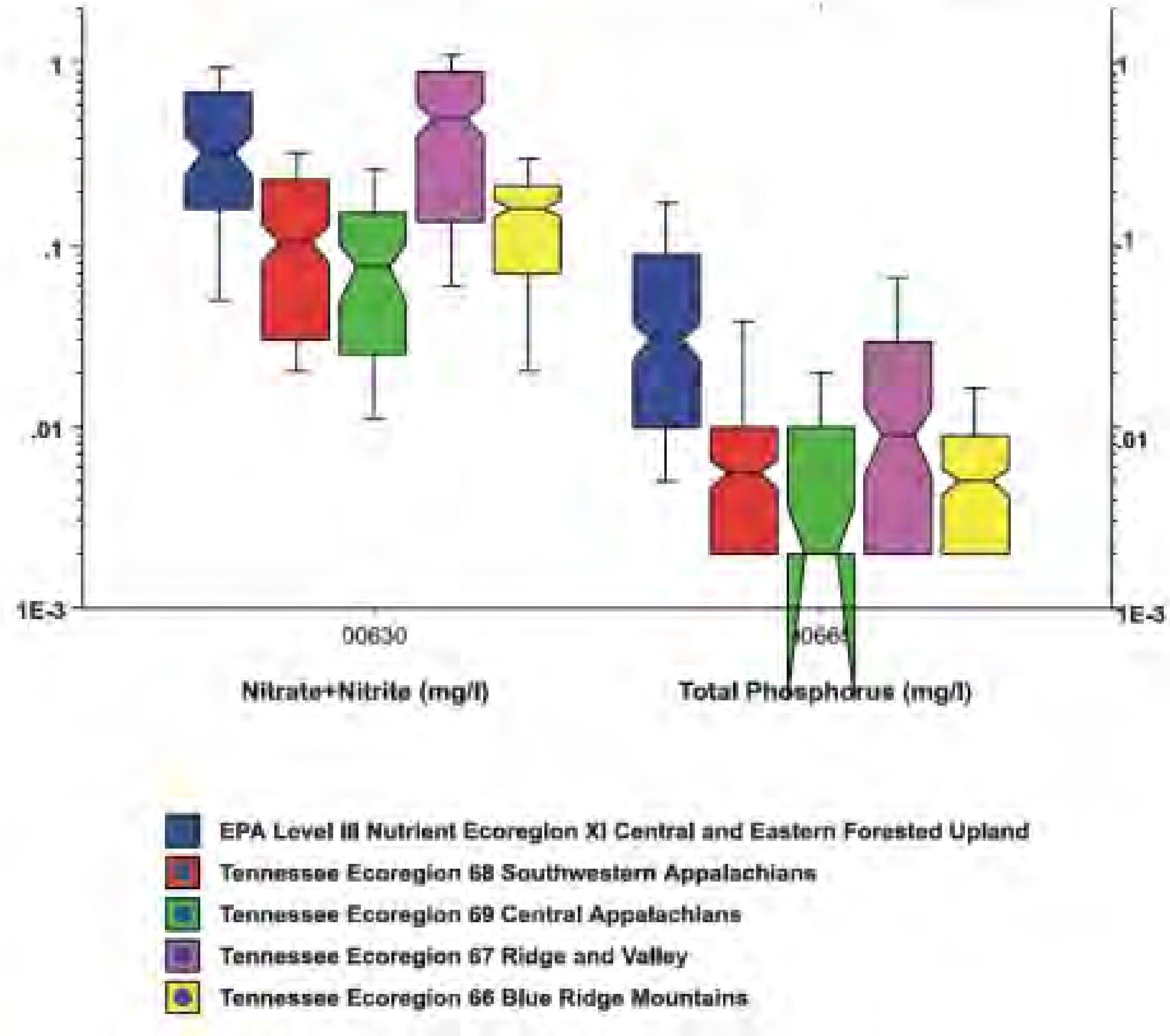

Figure A-9. Comparison of EPA Nutrient Ecoregion Region XI data to the Tennessee reference stream database for the same geographic area.

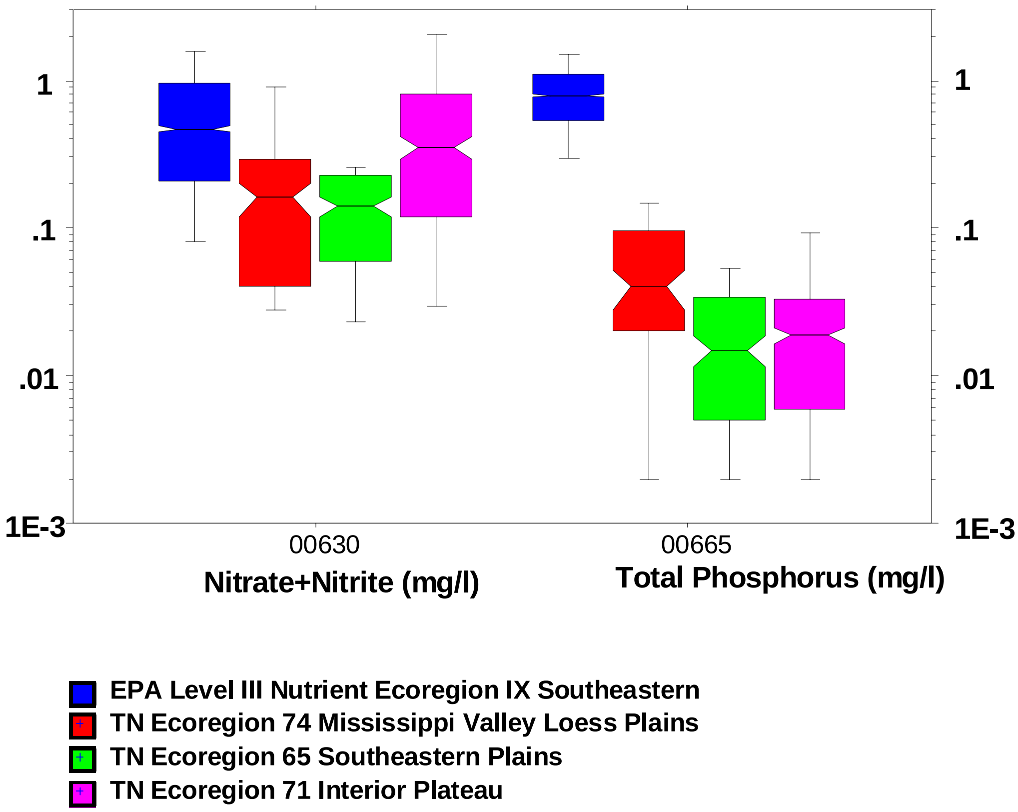

Figure A-10. Comparison of EPA Nutrient Ecoregion Region IX data to the Tennessee reference stream database for the same geographic area.

database of least-impacted conditions. These databases will be used to develop nutrient criteria based on either the 75^{th} or 90^{th} percentile of the data.

Attempts to identify a relationship between nutrient levels and other parameters such as turbidity, TOC, and suspended solids were confounded by the amount of data below the detection level. While data relationships were indicated, they were not strong. Further investigations might include similar comparisons using the national nutrient database values.

Relationships between nutrient data and biological indices were explored to see if positive correlations could be established. Such correlations could be used to strengthen a criteria justification and to insure that potential criteria values will be protective of biological integrity. The preliminary results are promising.

Tennessee's reference stream data were also compared to values from the national nutrient database. In several ecoregions, the 75^{th} percentile of the reference data corresponded well with the 25^{th} percentile of the national database. However, certain ecoregions did not correspond well, possibly suggesting that there are distinct differences within the EPA nutrient ecoregions. States would be well advised to consider these differences in setting nutrient goals.

Additionally, states should examine the national nutrient database carefully and use local knowledge to identify stormwater or embayment stations. Data from specific event sampling and reservoir or embayment stations may not be representative of the ambient water quality in the region. Such data could inappropriately bias results.

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Contact: Jim Harrison, Region 4 Nutrient Coordinator
United States Environmental Protection Agency
61 Forsyth Street, SW ◆Atlanta, GA 30303-3104
harrison.jim@epa.gov

CLARK FORK RIVER—SCIENTIFIC BASIS OF A NUTRIENT TMDL FOR A RIVER OF THE NORTHERN ROCKIES[1]

V.J. Watson,[2] Gary Ingman,[3] and Bruce Anderson[4]

ABSTRACT: In recent decades, river bottom algal levels have interfered with beneficial uses of western Montana's Clark Fork of the Columbia. The total maximum daily load analysis (TMDL) required by the Clean Water Act was addressed through a voluntary nutrient reduction plan developed by a stakeholder group with the aid of scientists. Targets for acceptable nutrient and algal levels were set using modifications of established criteria, literature values, and levels observed in the Clark Fork where algae problems did and did not occur. These targets were considered starting points that would be refined as more long-term data on the Clark Fork become available. Nutrient load reductions needed to meet instream targets were estimated using a model that diluted loads in the 30 day 10 year low flows. It appeared possible to achieve instream targets in most of the river with reductions that the main dischargers considered reasonably achievable, if other small dischargers and nonpoint sources were also controlled. Hence 4 local governments and one large industry signed the VNRP, and a VNRP coordinator was hired to obtain the participation of other sources.

KEY TERMS: nutrients, TMDL, benthic algae, benthic chlorophyll

INTRODUCTION

River bottom algal levels were first recognized as a water quality problem in the Clark Fork River of western Montana in the 1970's when it was found to lower dissolved oxygen levels below state standards on warm summer nights (Braico 1973). Massive algae growths and low oxygen levels were noted through the low flow summers of the 1980's (Watson 1989a; Watson and Gestring 1996) and identified as a critical problem by the Montana governor's office (Johnson and Schmidt 1988). In 1987, the reauthorization of the Clean Water Act called for a study and action plan to address nutrients and associated nuisance growths in the Clark Fork basin from Montana to Washington. The act also established the Tristate Implementation Council to carry out the study and plan. The resulting study (USEPA 1993b) documented that nuisance levels of algae were interfering with beneficial uses in 250 miles of river in Montana. The Council convened a group of stakeholders (dischargers, local governments, and conservation groups) which spent 4 years developing a voluntary nutrient reduction plan or VNRP to restore the river's integrity. The plan was signed in August, 1998, and EPA accepted the VNRP as a TMDL because it had a rational, scientific basis and provided a margin of safety. The VNRP will continue to serve as a TMDL as long as reasonable progress is shown toward its goals.

Unlike a TMDL, the VNRP did not require that effluent limits be written into permits, rather permits simply reference the VNRP which states the instream targets for algae and nutrient levels and timetables

[1] This paper was published previously and is used with permission of the publisher. Olsen, D. S. and J. P. Potyondy (Eds.). 1999. *Wildland Hydrology.* American Water Resources Association. Herndon, VA. TPS-99-3. 536 pp.
[2] Professor, Environmental Studies, University of Montana, Missoula, MT
[3] Chief, Data Mgt. & Monitoring, MT Dept. Env. Quality, Helena, MT
[4] Hydrologist, Land & Water Consulting, Missoula, MT

to achieve these, suggests likely loading reductions needed to achieve these, and lists some methods signatories agree to pursue to achieve reductions. By signing the VNRP, stakeholders agree to implement certain efforts to achieve loading reductions, to monitor and evaluate results and to pursue additional efforts if needed to reach targets. The VNRP also recognizes that the targets and reductions pursued are based on the best information currently available and are subject to renegotiation as more information becomes available. The VNRP references a long-term monitoring plan aimed at gathering more information and evaluating the effect of load reduction efforts. The VNRP also explains the scientific basis for the targets, load reductions, monitoring plan, margin of safety and areas of uncertainty. This paper discusses the scientific basis of the VNRP by addressing a series of questions.

QUESTIONS ADDRESSED BY THE VNRP

What Are Current and Desired Algae Levels in the River?

Summer algal levels in the Clark Fork vary dramatically in time and space, from highs of over 500 mg of chlorophyll a/sq. m. in the upper river in the 1980’s to lows of 3 mg/sq. m. at some sites in recent years (Watson and Gestring 1996; Watson unpublished data).

Currently, EPA is developing guidance to assist states in developing nutrient and algal criteria. It is likely that this guidance will direct the states to develop criteria based on little-impacted reference water bodies in each ecoregion. The Clark Fork VNRP committee found little guidance in the literature on what algal levels were natural to this region or what levels were associated with water quality problems.

The British Columbia Ministry of the Environment considers that recreation and aesthetics are protected when algal levels are below 50 mg chlorophyll a/sq. m. and that undesirable changes in aquatic life will be avoided at levels below 100 mg/sq. m. (Nordin 1985). Although these criteria were developed for small, shallow streams, Nordin agrees that it is reasonable to apply them to the shallow parts of large rivers (Nordin pers. comm.). Welch et al. (1988) demonstrated that filamentous algae tend to dominate stream communities when chlorophyll levels exceed 100 mg/sq. m. and proposed that nuisance levels exist above 100-150 mg/sq. m. The VNRP committee decided to adopt these target algal levels: less than 100 mg chlorophyll a/sq. m. when averaged over the growing season and 150 mg/sq. m. as the maximum acceptable peak.

The committee agreed that these algal targets might be revised in time as more information becomes available concerning what levels appear associated with water quality problems. In the mid 1980’s, river algal levels contributed to violations of the state dissolved oxygen standard. However, that standard has since been raised and is no longer exceeded, changing this view of what constitutes nuisance levels. But it was recently discovered that river algae lower dissolved oxygen and pH sufficiently on summer nights to release toxic heavy metals from old mine wastes in the river bed, violating water quality standards. Further studies are needed to determine what algal levels would avoid this and other water quality problems.

What Actions Seem Most Likely to Reduce Algal Levels?

Many factors affect river algal levels, including scouring, shading, grazing, toxic chemicals and available nutrients. The VNRP committee agreed that the factor that can best be managed to reduce algal levels in the Clark Fork is available nutrients.

How Much Must Nutrients Be Reduced to Achieve Algal Targets?

This question raised many others. What form of nutrients should be assessed, total or soluble? Which nutrient is most limiting, nitrogen or phosphorus? At what levels do nutrients become limiting? Should we focus on nutrient levels or loads?

Based on N:P ratios in the river, Watson (1989b) found that both N and P appear to be limiting at some times in some river reaches. Hence, the committee concluded that both nutrients should be reduced if possible. Artificial stream studies by Bothwell (1989) and Watson (1989) indicated what levels of soluble nutrients are low enough to reduce algal levels in artificial streams. However, using a 200 river database, Dodds et al. (1997) pointed out that total nutrient levels are better correlated with algal levels than are soluble nutrient levels. So the VNRP committee opted to focus on total nutrients (while monitoring soluble nutrients to insure they did not rise). A variety of approaches suggested targets ranging from 250-350 total N and 20-45 total P, so the committee adopted 300 ppb total N and 39 ppb total P in the middle river and 20 ppb total P in the upper river (where a higher N:P ratio was desired to discouraged the filamentous alga *Cladophora*).

What Are Major Nutrient Sources and How Much Reduction Is Needed?

The basin wide study called for in the 1987 Clean Water Act bill found that both point and nonpoint sources accounted for significant portions of nutrient loading, hence both must be reduced (USEPA 1993b). However, the largest sources were found to be three municipal discharges (Butte, Deer Lodge and Missoula), a pulp mill and a county (Missoula) with large areas of unsewered development. Hence these 4 local governments and one private industry were initial signatories to the VNRP. Ultimately, the VNRP committee hopes to convince smaller point sources and nonpoint sources (other developing counties and large landowners) to agree to certain efforts to control nutrients and to sign the VNRP.

To estimate the amount of load reductions needed, the Montana Department of Environmental Quality (DEQ) modified a model provided by EPA that estimates instream concentrations from loads, flows and historic percent losses within each river reach. This model allowed DEQ to estimate how much loads would need to be reduced from various sources to meet instream targets. Once again, the committee recognized that this simple model did not include all the gains and losses and so provided only a rough estimation of likely concentrations resulting from given loads. The model predicted that reductions the committee felt were reasonably possible would achieve instream targets in almost all the impaired reaches. The model suggested reaching targets in the few remaining miles of river would require reductions of questionable feasibility. The committee agreed to use the model only as a general guide and not to set required reductions. It was pointed out that algal uptake might reduce nutrient levels lower than the model predicted.

How Was a Margin of Safety Incorporated in the VNRP?

A margin of safety is provided by using instream nitrogen targets that are more protective than those recommended by Dodds et al. (1997). In addition, needed load reductions were estimated using the river's dilution capacity at very low flows—the 30 day 10 year low flow (the lowest 30 day average flow likely to be observed in one of 10 summers). Hence, targets will likely be met in almost all the river, in all but one month out of 10 years.

What Actions Are Expected to Achieve Needed Load Reductions?

All the municipalities in the area have adopted a phosphate detergent ban which has reduced P loads. The city of Deer Lodge agreed to land apply its wastewater. The city of Butte agreed to augment stream flows and pursue various land application options. The city of Missoula has reduced nutrient loading by operating its activated sludge plant like a biological nutrient removal plant. It also plans to construct a biological nutrient removal plant or use a combination of wetland treatment and land application in the future. The pulp mill will reduce summer discharge, store its water so as to reduce seepage, and increase use of a color removal process that also reduces nutrients. Missoula County will reduce and control loading from septic systems through land use planning and controls.

The VNRP committee has hired a VNRP coordinator to work with small discharges, local governments and land owners to identify ways these can reduce or at least control nutrient loads. These efforts are needed to avoid losing ground given the rapid population growth occurring in the area.

How Will Progress Towards the Targets Be Determined?

The TriState Implementation Council contracted with Land & Water, Inc., to develop and carry out a long term monitoring plan (Land & Water 1996) that will provide reliable information on nutrient related water quality status and trends in the basin. The monitoring plan uses a statistically rigorous sampling scheme designed to be able to detect trends in algal and nutrient levels in the Clark Fork and to assess compliance with instream targets. Using a seasonal Kendall with Sen slope estimate, the monitoring plan is intended to be able to detect a 50% change in nutrient levels over a 10 year period with 95% confidence and 90% power. In addition it can detect a 35% change in algal levels over a 10 year period with 90% confidence and 80% power. Compliance with instream targets will be evaluated annually using excursion analysis.

Monitoring consists of sampling 32 stations on the mainstem and major tributaries for total and soluble nutrients monthly (with biweekly sampling in summer). Algal levels are sampled at 7 mainstem stations twice a summer. Because of the high spatial variability in algal distributions, 10-20 replicates are collected. Details of the algal sampling scheme appear in Watson and Gestring (1996).

Timelines in the VNRP focus on timing of actions. However, the goal of the VNRP is to reduce algal level to the point that beneficial uses are fully supported by the end of the 10 year plan. Hence, the plan should be regarded as successful if a significant downward trend in nutrient and algal levels is detected 5 years into the plan, and if targets are no longer exceeded by the end of the 10 year plan. Of course, it will be necessary to evaluate changes in these parameters in light of the flows observed over this 10 year period.

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Contact: Debbi Hart (4304), Headquarters Nutrient Program
United States Environmental Protection Agency
Ariel Rios Building ◆1200 Pennsylvania Avenue, NW
Washington, DC 20460
hart.debra@epa.gov

UPPER MIDWEST RIVER SYSTEMS—ALGAL AND NUTRIENT CONDITIONS IN STREAMS AND RIVERS IN THE UPPER MIDWEST REGION DURING SEASONAL LOW-FLOW CONDITIONS

Stephen D. Porter, U.S. Geological Survey, Water Resources Division, Denver, CO 80225

INTRODUCTION

Extensive agricultural practices in the Midwestern Corn Belt region over the past 100 years have contributed to nonpoint source degradation of water quality and biological integrity in many streams and rivers. For example, intensive row-crop production and confined animal-feeding operations in Iowa, Illinois, and southern Minnesota have resulted in accelerated nutrient and organic enrichment in tributary streams, as well as in the Mississippi River (Goolsby et al. 1991; Coupe et al. 1995) and the Gulf of Mexico (Turner and Rabalais 1994; Rabalais et al. 1996). When ambient light and other algal-growth factors are favorable, nutrient enrichment can promote excessive productivity and respiration in streams and rivers, resulting in aesthetic and recreational impairments, departures from water quality criteria, and adverse effects to aquatic life. The U.S. Environmental Protection Agency (USEPA) has been charged with developing guidance for establishing regional water quality criteria to protect streams and rivers from accelerated eutrophication processes (http://www.cleanwater.gov). Results from State water quality (305[b]) reports to Congress indicate that over 40% of streams and rivers in the U.S. are contaminated by nutrient runoff and resultant indicators of excessive algal productivity.

Despite the prevalence of eutrophication, no implicit standards or criteria have been proposed to protect beneficial water uses (e.g., no significant ecological changes) in streams and rivers, apart from drinking-water standards for nitrate and chronic aquatic life criteria for elemental phosphorus in estuarine/marine waters. Although predictive algal-nutrient relations have been established for classifying the trophic status of lakes and reservoirs (Carlson 1977; Reckhow and Chapra 1983), there is no generally accepted system for classifying streams and rivers (Dodds et al. 1998; Dodds and Welch 2000). Recent approaches for classifying algal-nutrient relations in lotic systems have focused on constructing frequency distributions of total nutrients and periphyton (Biggs 1996; Dodds et al. 1998) or seston (suspended algae or phytoplankton) (Van Niewenhuyse and Jones 1996), and establishing boundaries between oligotrophic–mesotrophic and mesotrophic-eutrophic conditions, similar to trophic criteria established for lakes. Results from these investigations have suggested criteria for total nitrogen (TN>1500 μg/L), total phosphorus (TP>75 μg/L), seston chlorophyll *a* (chl *a*>30 μg/L), and periphyton (chl *a*>100-200 mg/m^2) to avoid adverse effects of stream eutrophication. Periphyton results from these and other such studies (Welch et al. 1988; Biggs and Close 1989; Lohman et al. 1992; Watson and Gestring 1996; Dodds et al. 1997) are representative of streams with gravel or rock substrates that were characterized by nuisance growths of filamentous green algae. Relatively little is known about nutrient and algal-productivity relations in low-gradient streams with unstable, sand, or silt bottoms. Even less is known about natural and human factors that contribute to the predominance of seston or periphyton in streams, relations with landscape factors such as agricultural intensity and riparian zones, and how differences in algal-nutrient relations influence stream metabolism and biological integrity.

To provide better understanding of eutrophication conditions and processes in streams and rivers in the upper Midwest Corn Belt region, the USGS National Water-Quality Assessment (NAWQA) Program conducted a large water quality study in the Minnesota, Wapsipinicon, Cedar, Iowa, Skunk, and Illinois River basins during seasonal low-flow conditions in August 1997. The study was a cooperative effort among three NAWQA projects: the Upper Mississippi River basin, Eastern Iowa basins, and Lower Illinois River basin study units. The objective of the study was to evaluate algal and macroinvertebrate responses to nutrient, herbicide, and organic enrichment from nonpoint agricultural sources relative to natural factors such as riparian vegetation, soil-drainage characteristics, and hydrology. This paper summarizes the status of algal and nutrient conditions in portions of the Central and Western Corn Belt Plains ecoregions (Omernik 1986), which could serve as a starting point for USEPA and State/Tribal agencies to establish regional nutrient criteria in rivers and streams in relation to low-flow conditions.

METHODS

Water chemistry and biological samples were collected from 70 streams and rivers in southern Minnesota, eastern Iowa, and western Illinois during seasonal low-flow conditions in August 1997. The study area is one of the most intensive and productive agricultural regions in the world; average row-crop production of corn and soybeans in stream watersheds accounts for over 90 percent of land cover (Sorenson et al. 1999). The density of riparian vegetation was quantified at two spatial scales: stream reach and segment. The length of a stream reach was approximately 20 times the mean wetted channel width (Fitzpatrick et al. 1998). The length of a stream segment was defined as the $\log_{10}$ of the basin area upstream from each sampling location, ranging from approximately 3 km to 4.9 km. Basin soil-drainage characteristics were quantified using information from the U.S. Soil Conservation Service STATSGO database. Water chemistry samples were collected for total and dissolved nutrients, dissolved herbicides and metabolites, and suspended and dissolved organic carbon (Shelton 1994). Stream productivity and respiration were estimated from continuous measurements of dissolved oxygen (DO) concentrations and pH over a 48-hour period. Phytoplankton (algal seston) samples were collected in conjunction with water-chemistry sampling, and quantitative samples of periphyton (benthic algae) and macroinvertebrates were collected from submerged woody debris. Water clarity was quantified using a light meter and submersible quantum sensor; the depth of the euphotic zone was measured or estimated by comparing subsurface photosynthetically-active radiation (PAR) with PAR measurements at the bottom of the deepest pool in the stream reach. Stream flow and velocity were measured using standard USGS procedures. Land-use and cover information was determined for each basin using ARC-INFO GIS procedures with the most-recent (1996-97) agricultural data that were available. A summary of the study design and methods, and data discussed in this report is presented by Sorenson et al. (1999; http://wwwrcolka.cr.usgs.gov/nawqa).

NUTRIENT INDICATORS OF TROPHIC CONDITION

Nutrient concentrations in many streams in the upper Midwest region are relatively higher than in other areas of the country, exceeding criteria proposed generally for temperate streams and rivers. For example, median concentrations of total nitrogen (TN; NH_4+NO_2+NO_3+organic N) and total phosphorus (TP; dissolved orthophosphate + particulate phosphorus) (Table A-1) exceeded the mesotrophic-eutrophic boundaries of 1500 μg/L (TN) and 75 μg/L (TP) proposed for temperate streams (Dodds et al. 1998). Average stream concentrations of dissolved nitrite+nitrate nitrogen (NO_2+NO_3-N) and total organic nitrogen (TON) were significantly ($p<0.05$) higher in the Minnesota River basin; nitrate concentrations exceeded 8 mg/L in nearly one third of these streams. Although concentrations and annual loads of TN increase with the intensity of nitrogen sources (e.g., fertilizer application and other land-use practices) nationally (Fuhrer et al. 1999), concentrations in Midwestern streams during seasonal, low-flow conditions were not related to rates of fertilizer application or the number of livestock in agricultural watersheds. Instead, NO_2+NO_3-N concentrations increased significantly with stream flow, corresponding with differences in rainfall and runoff in the region during the months prior to the study, and TON concentrations were correlated with the abundance of phytoplankton (seston), as indicated by chl *a* concentrations. Concentrations of NO_2+NO_3-N decreased significantly with increases in seston chl *a* concentrations. Particulate phosphorus (total phosphorus as P; Table A-1) concentrations did not differ significantly relative to human or natural factors; however, concentrations of dissolved orthophosphate (DoP) varied in relation to the importance of ground-water discharge and the abundance of benthic algae (periphyton) in Midwestern streams and rivers. Dissolved orthophosphate (available directly for algal growth) accounted for about 28 percent of the concentration of TP.

NATURAL FACTORS THAT INFLUENCE NUTRIENT INDICATORS OF TROPHIC CONDITION

Soil drainage and landform characteristics in the upper Midwest region were influenced profoundly by patterns of glacial advance and retreat during the late Pleistocene era. For example, soils on the Wisconsin glacial lobe in north-central Iowa and southern Minnesota are characterized by fine-grained materials through which water drains very poorly, whereas soils in eastern Iowa and western Illinois contain relatively larger proportions of sand and coarser grained materials that constitute moderately-well drained soils. The proportion of stream water that is derived from ground-water inflow is substantially less in streams on the Wisconsin lobe than in streams located to the southeast of the Wisconsinan glacial advance (Winter et al. 1998). Land-surface runoff, via tile drains, is probably an important contributor to nutrient fluxes in streams that drain low-gradient, prairie-pothole landscapes. In contrast, ground-water inflow contributes appreciably to stream flow, particularly during low-flow periods, in areas with moderately-well drained soils such as the Wapsipinicon, Cedar, and Illinois River

Table A-1. Distribution of nutrient concentrations (in µg/L) in Midwestern agricultural streams and rivers.

Water quality constituent	10^{th} percentile	25^{th} percentile	50^{th} percentile (median)	75^{th} percentile	90^{th} percentile	Maximum value
Total Nitrogen[1]	948	1364	2627	4205	8550	13679
Total Phosphorus[2]	61	114	175	235	523	1092
Dissolved NH_4-N	<15	<15	17	46	105	1308
Dissolved NO_2+NO_3-N	103	278	1320	3415	8145	12730
Total Organic N	356	570	934	1258	1560	2899
Dissolved ortho-P	<10	19	41	72	161	409
Particulate Phosphorus[3]	39	68	139	185	378	778

[1] Sum of dissolved NH_4-N + dissolved NO_2+NO_3-N + total organic N

[2] Sum of dissolved ortho-PO_4 + particulate phosphorus

[3] Total phosphorus as P (USGS WATSTORE code 00665)

basins (Walton 1965; Heintz 1970; O'Hearn and Gibb 1980; Squillace et al. 1996). Figure A-11 shows soil-drainage relations among stream and river basins in the study (U.S. Soil Conservation Service STATSGO data normalized to watershed area; Sorenson et al. 1999) and the correspondence with the Wisconsinan glacial advance.

Concentrations of TN and TP varied in relation to soil-drainage and riparian-zone conditions in the upper Midwest region (Figure A-12). Average TN concentrations were significantly higher in stream basins with very-poorly drained soils, such as those in the Minnesota River basin. In basins with moderately-well drained soils, concentrations of TN were significantly lower in streams with well-developed riparian zones, suggesting that the presence of riparian trees may beneficially influence water quality conditions in streams with appreciable ground-water discharge. Average TP concentrations were relatively lower in streams with moderately- or poorly-drained basins and well-developed riparian zones (Figure A-12), but concentrations of TP did not differ significantly in relation to riparian conditions. Average TP concentrations were significantly less in streams with a low percentage of riparian trees and very-poorly drained basins; however, average TP concentrations were generally (but not significantly) lower in streams with well-developed riparian zones (Figure A-13).

Dissolved nutrient concentrations differed in relation to basin soil-drainage properties and riparian-zone conditions, but nutrient conditions were more related to algal abundance and productivity in streams and rivers than physical factors. Average concentrations of dissolved ammonia-nitrogen (NH_4-N) were significantly higher in streams that drain basins with moderately well-drained soils, whereas average dissolved NO_2+NO_3-N concentrations were significantly higher in streams with very-poorly drained soils, such as those on the Wisconsin lobe. Similarly, average DoP concentrations were relatively higher in highly-shaded streams with moderately well-drained basins, whereas concentrations of DoP were significantly lower in poorly-shaded, poorly-drained stream systems on the Wisconsin lobe. The combination of very-poorly drained soils, high rainfall and land-surface runoff relations, and extensive tile drainage in the Minnesota River basin may account for the higher-than-expected concentrations of TN and dissolved NO_2+NO_3-N concentrations in these streams. Relatively larger concentrations of NH_4-N and DoP in streams with moderately-well drained basins may indicate ground-water fluxes of these constituents that reflect both present and past agricultural intensity. The time of constituent transport along local and regional ground-water flow paths can range from months to years. Integration of these results could indicate an interaction among land-use practices, stream hydrology, riparian shading, and algal-nutrient relations.

ALGAL INDICATORS OF TROPHIC CONDITION

Algal indicators of eutrophication in streams and rivers of the upper Midwest region are related to agricultural intensity (fertilizer application and livestock in stream basins), soil-drainage conditions, hydrology, and riparian-zone conditions along stream segments. Median and inter-quartile seston chl *a* concentrations (Table A-2) are similar to those reported from mesotrophic-to-eutrophic lakes and reservoirs (e.g., Carlson 1977), and seston (but not periphyton) chl *a* concentrations were significantly higher in poorly-shaded than well-shaded streams (Figures A-13 and A-14). These results likely indicate that ambient light conditions influence the development of large phytoplankton populations in Midwestern streams and rivers. Seston chl *a* values indicative of eutrophic conditions (greater than 30 μg/L) were found in streams that drain basins with poor soil drainage, high rates of fertilizer application,

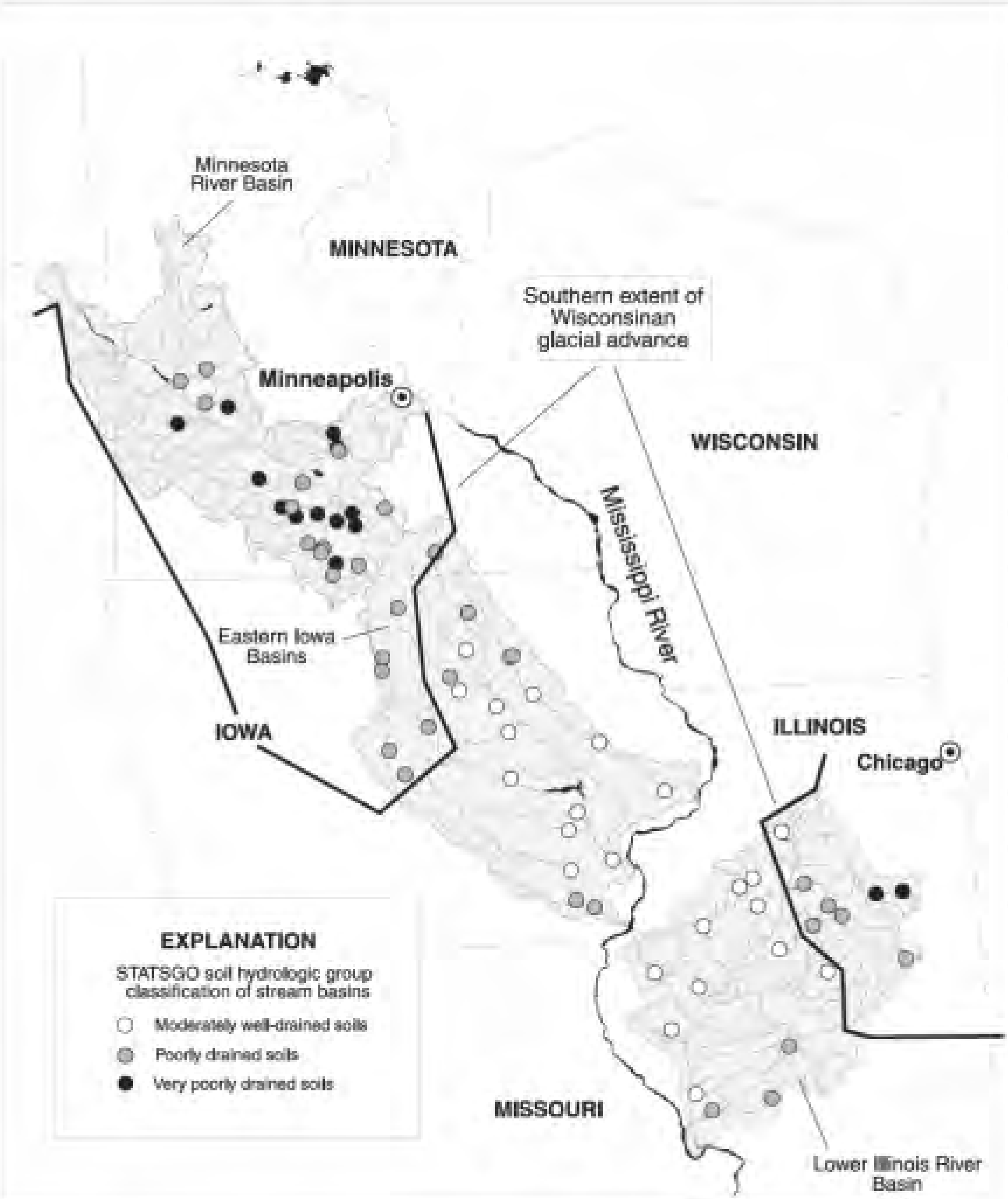

Figure A-11. Classification of Midwestern streams and rivers relative to basin soil-drainage characteristics and relation with southern extent of Wisconsin glacial advance.

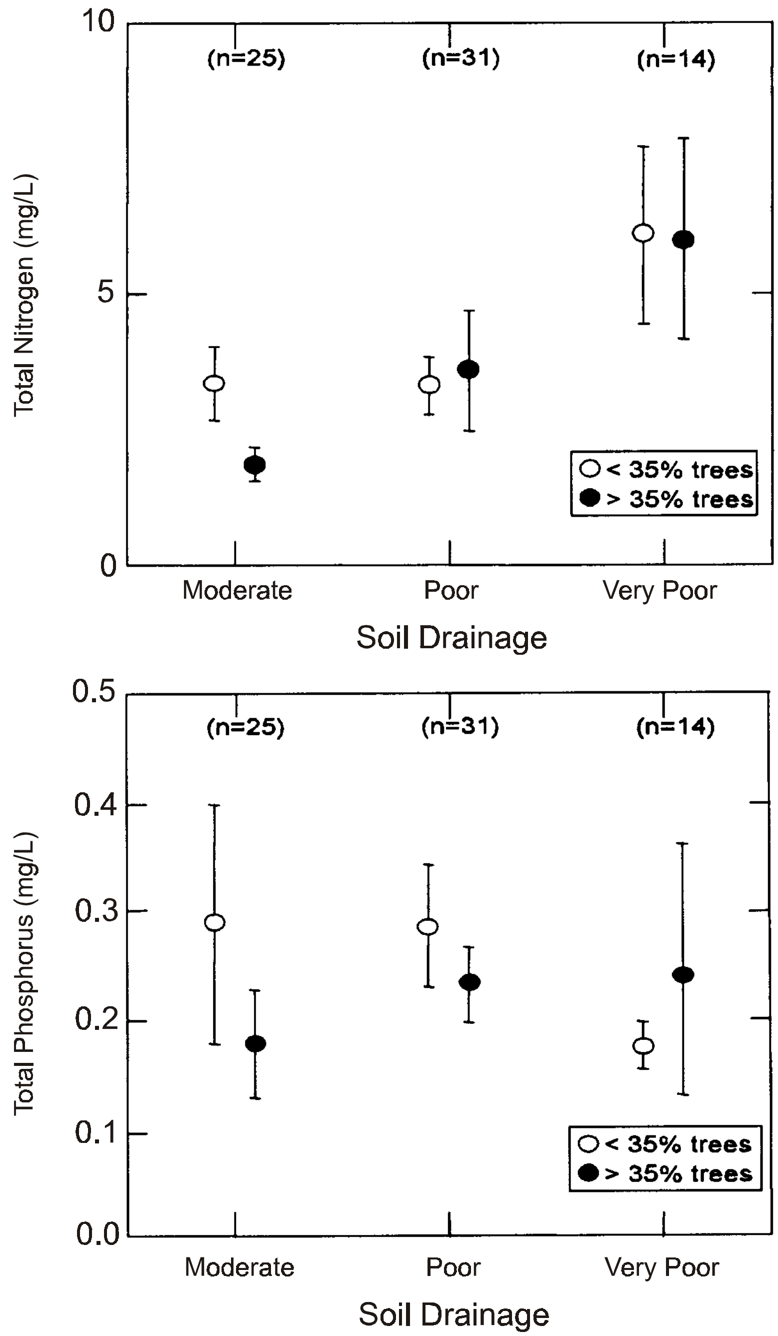

Figure A-12. Total nitrogen and phosphorus concentrations relative to soil drainage and riparian conditions in Midwestern streams and rivers.

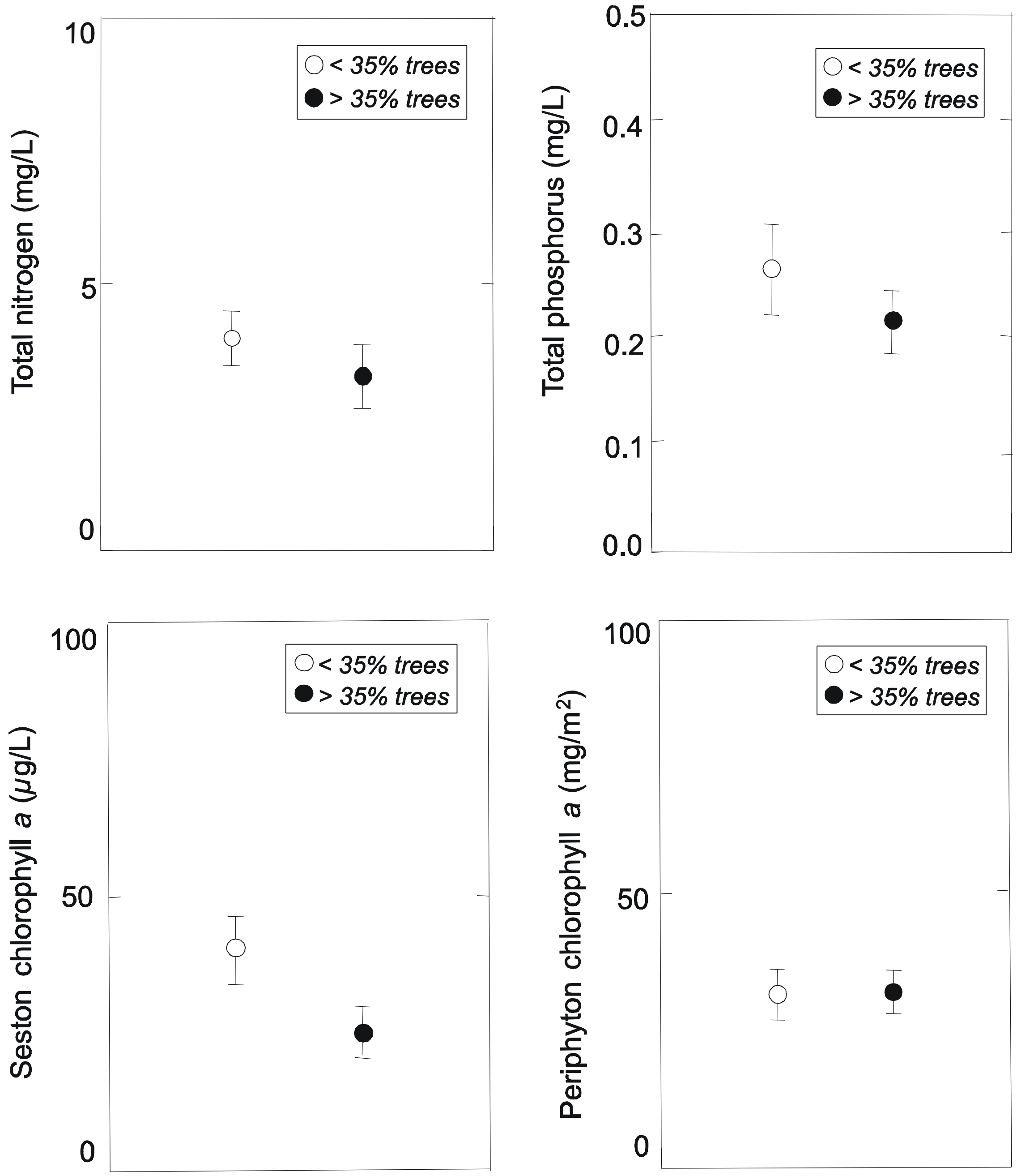

Figure A-13. Average concentrations of total nitrogen, total phosphorus, seston chlorophyll *a*, and periphyton chlorophyll *a* in relation to riparian-zone conditions.

Table A-2. Distribution of algal seston, periphyton, ash-free dry mass, stream productivity and respiration, suspended and dissolved carbon, total suspended solids, and water clarity (euphotic zone depth) in Midwestern agricultural streams and rivers.

Water quality constituent	10th percentile	25th percentile	50th percentile (median)	75th percentile	90th percentile	Maximum value
Seston chlorophyll *a* (mg/L)	6.40	11.0	18.4	38.7	71.7	175
Periphyton chlorophyll *a* ($\mu g/m^2$)	3.67	13.1	25.1	42.2	72.6	102
Periphyton ash-free dry mass (g/m^2)	15.7	19.3	25.4	31.5	39.5	57.8
Stream productivity ($g\ O_2/m^3/hr$)	0.113	0.242	0.398	0.697	0.998	1.46
Stream respiration ($g\ O_2/m^3/hr$)	0	0	-0.044	-0.159	-0.226	-0.804
Suspended organic carbon (mg/L)	0.5	0.7	1.3	2.5	3.6	5.0
Dissolved organic carbon (mg/L)	2.4	3.6	4.3	5.8	7.2	11
Total Suspended Solids (mg/L)	19	41	72	128	158	330
Estimated euphotic zone depth (m)	0.32	0.42	0.56	0.68	0.84	1.4

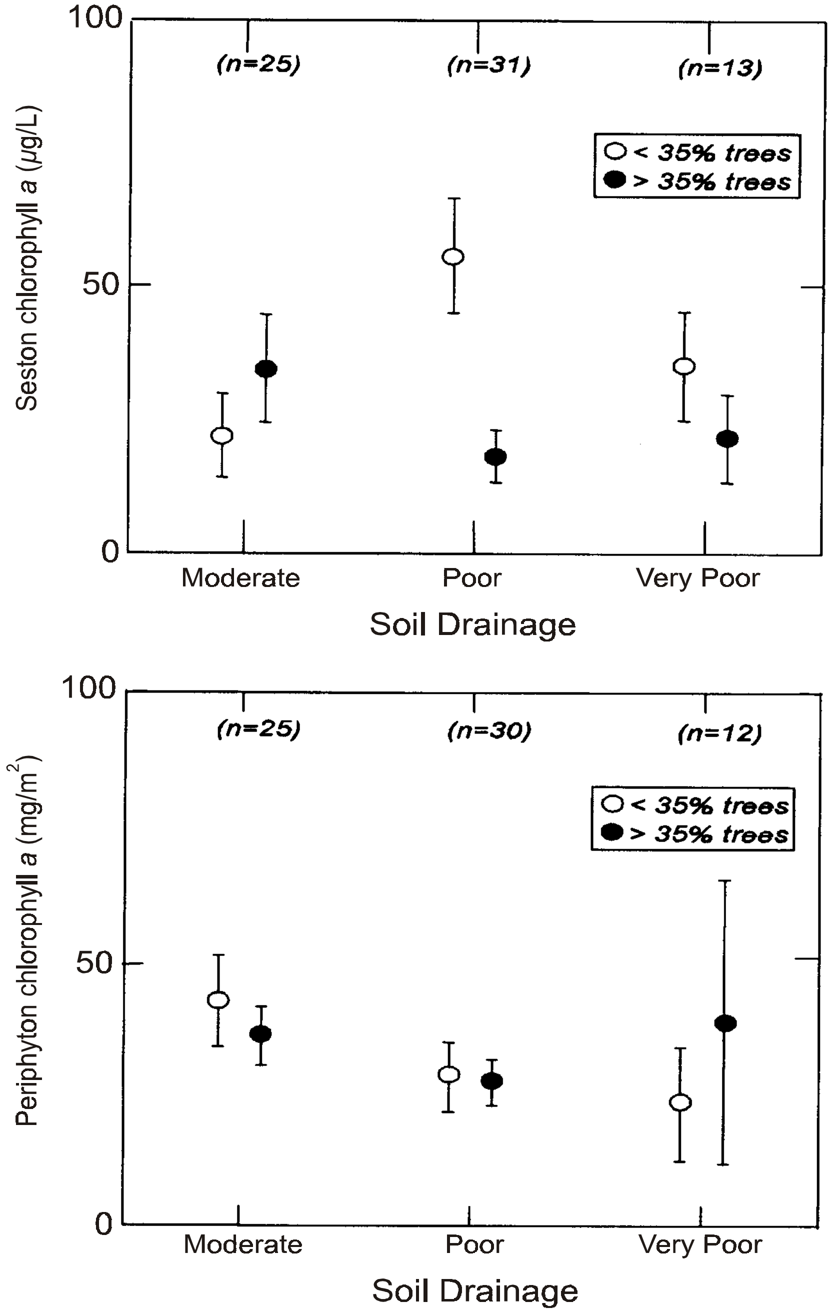

Figure A-14. Seston and periphyton chlorophyll a values relative to soil drainage and riparian conditions in Midwestern streams and rivers.

and relatively large populations of hogs and other livestock. Stream productivity (P_{max}) and respiration (R_{max}) values increased significantly with seston chl *a* concentrations. Concentrations of NO_2+NO_3-N decreased significantly with increases in stream productivity, which is probably correlated with algal uptake of dissolved nutrients. Seston chl *a* concentrations were positively correlated with concentrations of suspended organic carbon (SOC), TON, particulate phosphorus, and total suspended sediment (TSS), which suggests that total nutrient and organic enrichment in Midwestern streams is reflected by large populations of algal seston. Seston chl *a* concentrations were negatively correlated with euphotic zone depth, indicating that water clarity decreases with increases in the abundance of suspended algae (phytoplankton).

Periphyton chl *a* values were significantly larger in streams with high water clarity and riparian shading, and above-average stream velocity. Concentrations of total and dissolved nutrients and seston chl *a* in periphyton-dominated streams were generally moderate to low; however, productivity (P_{max}) was about average for Midwestern streams, suggesting that periphyton (rather than seston) influences the productivity of streams with high riparian shading and appreciable ground-water discharge. Large populations of diatoms and blue-green algae were observed growing on sand (near the hyporheic zone) in these streams. Although concentrations of NH_4-N and DoP were larger in streams that drain basins with moderately-well drained soils, regionally, periphyton uptake of dissolved nutrients from ground-water discharges might account for the lower-than-expected concentrations of these constituents in the Wapsipinicon and Cedar River basins of eastern Iowa. While P_{max} rates in periphyton-dominated streams were near average regionally, rates of stream respiration (R_{max}) were generally low, and early-morning concentrations of DO appeared to be favorable for aquatic life. In contrast, rates of R_{max} were relatively high in seston-dominated streams; DO concentrations during early-morning hours were low and benthic macroinvertebrate community structure was poor (Harris and Porter in review).

Periphyton chl *a* and ash-free dry mass (AFDM) values were positively correlated; however, chl *a* and AFDM relations (refer to Table A-2) differed with respect to precedent stream-flow conditions, water clarity, and non-algal sources of carbon. Ratios of chl *a* to AFDM were relatively low (less than one) in over half the streams in the Minnesota River basin, where the organic content of soils is relatively high and soil drainage is very poor. In addition, above-average stream flow and water turbidity, as well as a higher frequency of hydrologic disturbances associated with summer storms during the months prior to the study (Figure A-15) probably limited the growth of algal periphyton in the Minnesota River basin. In contrast, chl *a*/AFDM ratios were larger (greater than one) in streams with relatively stable stream flow and good water clarity.

Periphyton samples were analyzed for species composition and abundance (cells/cm^2), and the biovolume of each algal taxon was determined by measuring cell dimensions and calculating the average volume of the cell (μm^3) in relation to the nearest geometric shape (e.g., sphere, cylinder, etc.). Biovolume ($\mu m^3/cm^2$) for each species was calculated by multiplying the volume of one cell by the abundance of the species in the sampling reach. Total algal biovolume (cm^3/m^2) was estimated by summing biovolumes for all species present in the sample. Total algal biovolume (TAB) is positively correlated with periphyton chl *a* and AFDM, and periphyton chl *a* can be estimated from TAB using the following regression relation:

$$\text{chl } a = (4.229 + 2.733 * \log_{10}(\text{TAB}))^2 \qquad \text{adjusted } R^2=0.570;\ p<0.001;\ n=67$$

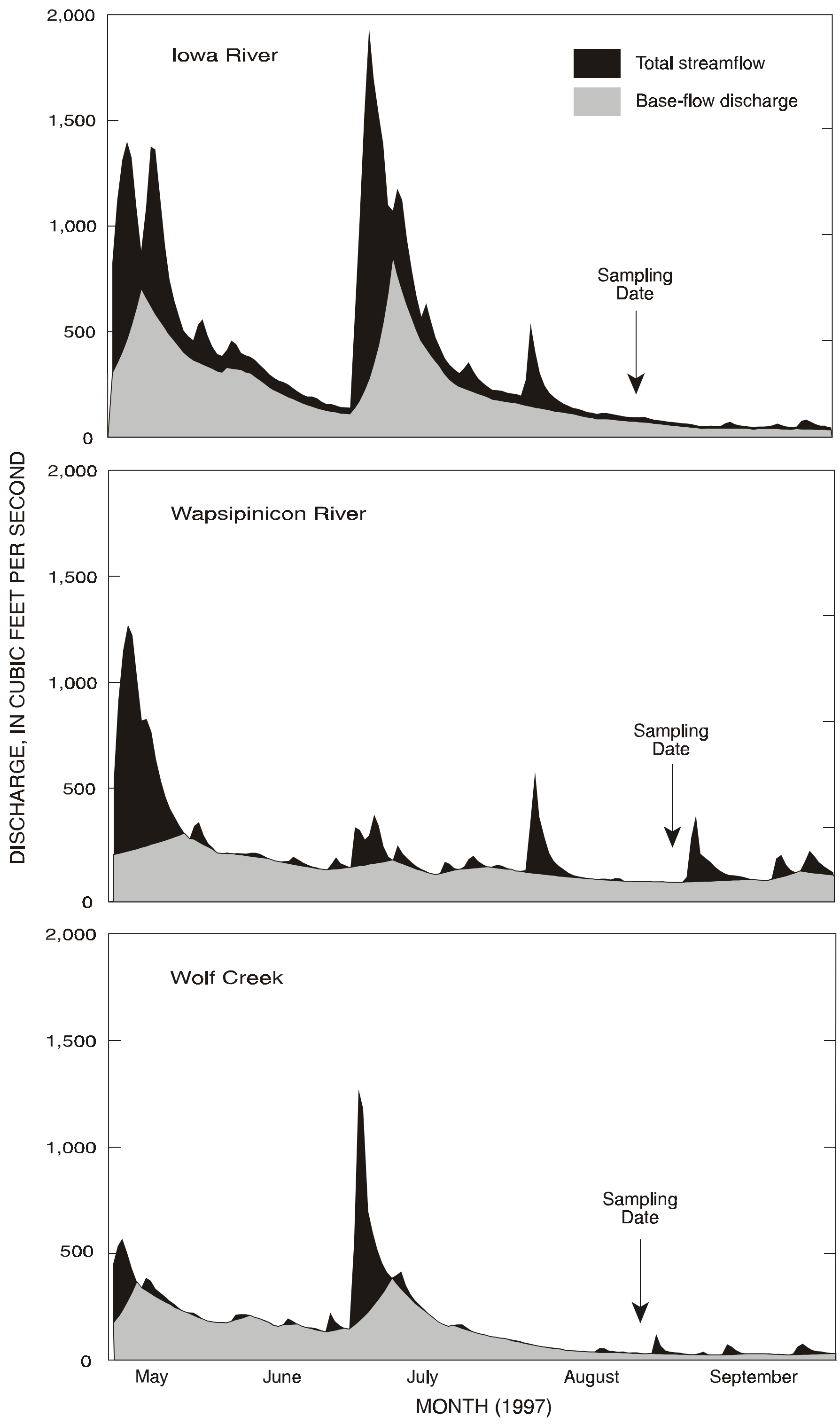

Figure A-15. Total and base flow discharge for selected Midwestern streams and rivers in relation to collection date of water quality samples.

where: chl *a* = periphyton chlorophyll *a* (mg/m^2)
TAB = total algal biovolume (cm^3/m^2)

Unexplained variance associated with the regression is probably attributable to differences in chl *a* content among algal species, differences in riparian shading or other factors that influence ambient light conditions (e.g., Darley 1982; Rosen and Lowe 1984), and challenges with distinguishing live and dead cells during taxonomic enumeration.

Periphyton communities were dominated by eutrophic microalgae (diatoms, blue-green algae, and green algae). Filamentous red algae (*Audouinella hermanii*) were abundant in streams with above average water clarity, velocity, and riparian-tree density (i.e., periphyton-dominated streams). Filamentous green algae were relatively uncommon on submerged woody debris; however, sparse to moderate growths of *Cladophora glomerata* were observed in flowing streams with bedrock or boulders, and moderate growths of *Spirogyra* spp. were present in pools and slow-flowing sections of streams with sand or silt bottoms. The predominance of fine streambed materials (sand and silt) in many Midwestern streams probably precluded the establishment and growth of nuisance filamentous algal species. These factors probably account for the relatively lower periphyton chl *a* values observed in this study when compared with those associated with eutrophic streams in the western U.S., that are dominated by filamentous green algae (Welch et al. 1988; Watson and Gestring 1996; Dodds et al. 1997).

FACTORS ASSOCIATED WITH SESTON OR PERIPHYTON DOMINANCE IN MIDWESTERN STREAMS AND RIVERS

Streams were classified relative to the abundance of algae (seston, periphyton, or both) as indicated by chl *a* values, and analysis of variance (ANOVA) and Tukey multiple range tests were used to determine whether water quality conditions differed significantly among stream classifications. Approximately 30 percent of the streams contained above-average (relative to this data set) concentrations of seston chl *a* and below-average values for periphyton chl *a* (seston-dominated streams). Concentrations of TON, dissolved organic carbon (DOC), and SOC were significantly higher in seston-dominated streams, suggesting organic enrichment from autotrophic (in-stream) processes. About 28 percent of streams contained below-average concentrations of seston chl *a* and above-average values for periphyton chl *a* (periphyton-dominated streams). Water clarity in periphyton-dominated streams was good, as indicated by significantly lower TSS concentrations and significantly larger euphotic-zone depths.

Stream productivity (P_{max}) was moderate to high in all streams with above-average amounts of algae; however, high rates of stream respiration (R_{max}) were associated primarily with large populations of seston. High R_{max} conditions were associated with low DO concentrations during early morning hours, at levels that can adversely affect aquatic fauna. Stream respiration was significantly higher in streams with above-average chl *a* values for both seston and periphyton (algal-eutrophic streams; 22 percent of streams in the study). Concentrations of dissolved (but not total) nutrients were relatively lower in algal-eutrophic streams than in streams with below-average seston and periphyton chl *a* (nutrient-eutrophic streams; 20 percent of streams in the study). Stream productivity (P_{max}) was significantly higher in algal-eutrophic and seston-dominated streams than in nutrient-eutrophic and periphyton-dominated streams. However, periphyton chl *a* decreased significantly with modest increases in the relative abundance of macroinvertebrate scraper organisms (Harris and Porter, in review); therefore, monitoring of algal-nutrient relations in Midwestern streams should probably consider the abundance of grazer organisms that consume benthic algae.

The abundance of algal tychoplankton (species that are loosely associated with, but not attached to, submerged benthic surfaces) in the periphyton community was a primary factor in identifying differences in community structure among Midwestern streams. These species, including *Microcystis*, *Anabaena*, other blue-green algae, and centric diatoms, are found commonly in eutrophic lakes, reservoirs, and other warm, slow-flowing water bodies such as large impounded rivers. The abundance of blue-green algae increased with the concentration of triazine herbicides (atrazine, cyanazine, and degradation products). The predominance of tychoplankton in periphyton communities in algal-eutrophic and seston-dominated streams was associated with large populations of these species in the seston, probably indicating that they had settled from the water column. Indicators of organic enrichment (SOC, DOC, TON) and stream metabolism (P_{max} and R_{max}) are consistent with the large abundance of algae in these streams, whereas concentrations of dissolved nutrients were relatively low. The highest rates of stream respiration were found in algal-eutrophic streams; benthic macroinvertebrate indicators of biological integrity (e.g., EPT richness) indicated poor water quality conditions in algal-eutrophic and seston-dominated streams (Harris and Porter in review).

In contrast, algal communities in periphyton-dominated and nutrient-eutrophic streams were dominated by diatoms, blue-green algae, and red algae that grow attached to benthic surfaces. These species are found commonly in cool, flowing streams and rivers. A secondary factor in classifying differences in algal community structure in the region relates to the age of the periphyton community as inferred by the presence or dominance of certain algal species. For example, periphyton communities in streams of the Minnesota and upper Iowa River basins were characterized by diatoms (e.g., *Fragilaria vaucheriae* and *Achnanthidium minutissimum*) that are typically found in abundance on bare or recently-scoured substrates. Algae that are associated with soils (e.g., *Luticola mutica*, *Chlorococcum* sp., and *Protococcus* sp.) were also common in these streams. Periphyton community structure in these streams is consistent with recent hydrologic disturbance as indicated by relatively high rainfall, surface-water runoff, and elevated streamflow in the region (Figure A-15). Water quality in these streams is influenced by relatively low rates of stream metabolism and high concentrations of nutrients (notably TN, NO_2+NO_3-N and TP). In contrast, periphyton communities in streams of the Wapsipinicon and upper Cedar River basins consisted of species found commonly in diverse, mature algal communities (e.g., *Audouinella hermanii*, *Navicula* spp. and *Gyrosigma* spp.), which is consistent with relatively stable hydrologic conditions, ground-water discharge, and seasonally-typical streamflow (Figure A-15).

SUMMARY AND IMPLICATIONS FOR ESTABLISHING AND MONITORING ALGAL-NUTRIENT CRITERIA

Nutrient concentrations and the abundance of algae during low-flow conditions were not related directly to rates of fertilizer application or the number of livestock in Midwestern stream basins; however, rates of stream metabolism (P_{max} and R_{max}) increased significantly with indicators of agricultural intensity. Algal-nutrient relations during August 1997 were more a function of landscape characteristics (riparian zones and soil properties), hydrology (ground-water and surface-water relations), and rainfall-runoff characteristics than agricultural land use, which is relatively homogeneous throughout the region. For example, average nutrient concentrations were significantly higher in the Minnesota River basin despite relatively lower agricultural intensity. Above-average rainfall and runoff from poorly drained soils, discharged through tile drains, probably explains the higher-than-expected nutrient concentrations in these streams. Average rates of stream metabolism were relatively lower in streams in the Minnesota River basin, which is consistent with relatively higher concentrations of suspended solids and lower water clarity. Over half of these streams contained above-average seston chl *a* concentrations, which corresponds with relatively less riparian shading in Minnesota than in Illinois or Iowa. However, seston and periphyton communities were dominated by species associated with soils or those with high rates of colonization and reproduction. Benthic invertebrate and periphyton communities contained relatively fewer species; however, reduced species richness was more indicative of hydrologic disturbance (high, flashy stream flow and velocity) than organic enrichment.

In contrast, average dissolved nitrate concentrations in the Illinois River basin were significantly lower, even though agricultural intensity in those stream basins was among the highest in the region. Below-average rainfall (near drought conditions), resulting in significantly lower (surface water) nutrient yields from stream watersheds, lower stream velocities, and high rates of stream metabolism, probably explain the lower-than-expected dissolved nutrient and DO concentrations. However, concentrations of dissolved NH_4-N were relatively higher, probably attributable (in part) to ground-water fluxes in basins with moderately well-drained soils. Water quality conditions in Illinois streams during August 1997 were relatively degraded, as revealed by relatively high concentrations of SOC, DOC, and TON (indicators of organic enrichment), low minimum dissolved-oxygen concentrations, high rates of stream respiration, and poor macroinvertebrate communities (low taxa and EPT richness).

Water quality in Iowa streams differed in relation to basin soil properties and riparian shading. Overall water quality was best in streams that drain basins with moderately well-drained soils and a high percentage of riparian trees (Wapsipinicon and upper Cedar River basins). These periphyton-dominated streams were characterized by low to moderate concentrations of nutrients and average stream productivity. Seston chl *a* values and rates of respiration were relatively low, and macroinvertebrate communities (e.g., EPT richness) indicated good water quality and habitat conditions.

Although phytoplankton chl *a* criteria are available to classify the trophic status of lakes and reservoirs (e.g., Carlson 1977), comparable criteria have not been established for seston or periphyton in lotic water bodies. Average chlorophyll values in the upper Midwest region are considerably lower than criteria proposed by Dodds et al. (1998) for temperate streams and rivers, whereas proposed criteria for total nutrients (TN>1500 μg/L; TP>75 μg/L) are exceeded in 74 percent (TN) to 89 percent (TP) of the streams in this study. Periphyton chl *a* values exceeded 70 mg/m^2 (proposed <u>minimum</u> eutrophic criterion) in only 13 percent of the streams, and seston chl *a* values exceeded 30 μg/L (proposed eutrophic criterion) in about one-third of the streams in this study. The higher recommended criteria for periphyton chl *a* (100 mg/m^2 to 200 mg/m^2) (Welch et al. 1988; Watson and Gestring 1996; Dodds et al. 1998) was intended to protect streams and rivers from nuisance growths of filamentous algae such as *Cladophora glomerata*, other macroalgae, or other aquatic plants. These taxa require stable benthic surfaces (e.g., submerged rocks or bedrock) on which to colonize and grow to nuisance proportion. Sand and silt bottom streams of the Midwest, and submerged woody debris in these streams, do not generally provide suitable habitat to sustain nuisance filamentous algal growths. However, dense growths of microalgae (primarily diatoms and blue-green algae) on sand or woody snags in Midwestern streams could provide visible evidence of stream eutrophication during low-flow periods; the proposed minimum eutrophic criterion may be appropriate for indicating that condition.

Results from this study suggest that the abundance and composition of algal seston (phytoplankton) may be one of the better indicators of trophic conditions in streams and rivers of the upper Midwest region. Because of the highly significant correspondence between the standing crop (e.g., chl *a*) of algal seston and concentrations of total nutrients and carbon, criteria established for seston (evaluated during stable, low-flow conditions) is likely to represent total nutrient concentrations in the water and the extent to which organic enrichment is a problem for maintaining biological integrity in streams and rivers. Seston criteria would also provide an index for evaluating the clarity of streams and rivers, an important consideration relative to the public perception of trophic conditions, and water quality in general. However, criteria for total nutrients (and perhaps total suspended solids) cannot be abandoned for streams where algal growth is limited by inorganic turbidity or dense riparian-canopy shading. For example, if best-management practices (BMPs) are applied in watersheds to reduce adverse effects of sedimentation without consideration given to commensurate reductions in nitrogen or phosphorus loads to streams, excessive algal growths could ensue when productivity is no longer limited by the availability of light (e.g., in nutrient eutrophic streams and rivers). A consideration of water quality variables for establishing and monitoring the trophic condition of temperate streams and rivers is presented in Table A-3.

Table A-3. Summary of water quality variables for establishing criteria and monitoring the trophic condition of temperate streams and rivers.

Variable	**Media (and frequency)**	**Relevance**	**Risk**
Total nutrients	Water chemistry (monthly & in relation to hydrology)	chemical indicator	eutrophy
Dissolved nutrients	Water chemistry (monthly & in relation to hydrology)	chemical indicator algal-nutrient relations	eutrophy
Seston	Water samples (growing season & in relation to hydrology)	biological indicator organic enrichment food-web relations	eutrophy aquatic life biocriteria
Periphyton	Natural substrates (growing season & in relation to hydrology and aquatic herbivores)	biological indicator organic enrichment food-web relations	eutrophy; aquatic life; biocriteria
Stream metabolism	Estimates of system productivity & respiration (low flow conditions with chemical & biological measures)	biological indicator understanding	direct measure of process; aquatic life; biocriteria
Water clarity	Euphotic zone depth; water transparency; secchi depth (seasonal; with chemical & biological measures)	physical indicator understanding	aesthetic properties light availability for algal growth
Aquatic fauna	Natural substrates (low-flow conditions)	biological indicator understanding response to organic enrichment	receptor biocriteria TMDL process

Improved understanding of natural factors and algal-nutrient relations that contribute to chemical and biological indicators of eutrophication in lotic systems could enhance the development of water quality criteria within and among ecoregions in the U.S. (e.g., Level III; Omernik 1986). For example, results from this study indicate larger variance within the Western Corn Belt Plains ecoregion than between the Central and Western Corn Belt Plains ecoregions. Differences in soil drainage, ground-water/surface-water relations, and precedent rainfall-runoff conditions account for part of this variance. Improved understanding of dissolved nutrient relations with the abundance of seston and periphyton, rates of stream metabolism, and organic enrichment processes in streams could assist water managers with decisions concerning BMPs, total maximum daily load (TMDL) allocations, and the establishment of appropriate biocriteria relative to natural and human factors that contribute to the quality of streams and rivers.

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Contact: Dave Pfeifer, Region 5 Nutrient Coordinator
United States Environmental Protection Agency
77 West Jackson Boulevard ◆Chicago, IL 60604-3507
pfeifer.david@epa.gov

BOW RIVER, ALBERTA

The Bow River is a documented case of recovery from point source nutrient loading rather than one of setting criteria. In contrast to lakes, cases in which the recovery of streams or rivers from nutrient reduction was thoroughly evaluated are scarce. The Bow River, Alberta, is an exception; it has been monitored for over 16 years to evaluate the effect of a reduction in first phosphorus (80%) and later nitrogen (~ 50%) from two domestic wastewater plants in Calgary (Sosiak pers. comm.). Algae and macrophytes had caused problems in the river by clogging irrigation water intakes, interfering with boating and angling, and causing low DO at night. Nitrogen removal was for the purpose of minimizing risk of ammonia toxicity rather than control of algae or macrophytes. Both periphyton and macrophytes decreased downstream in response to nutrient reduction, but the distribution and timing of the decreases were to some extent unexpected. The river's response to nutrient reduction offers pertinent implications and guidance for setting nutrient criteria in large fast-flowing, gravel-bed rivers. Median April to October flow in the Bow River over the sampling period ranged from approximately 75 to 130 m^3/s.

Prior to P reduction, periphyton biomass consisted mostly of diatoms, although filamentous green algae (including *Cladophora*) were also present (Charlton et al. 1986). Biomass reached summer maximums downstream averaging approximately 300-400 mg chlorophyll a/m^2, but occasionally up to 600 mg chlorophyll a/m^2. Such maxima have persisted within 10 km of the effluent input since P reduction in 1983, but decreased markedly farther downstream over an approximately 90-330 km reach (Table A-4; note that data from two stations between km 304 and 533 are not shown). The decrease in periphyton occurred rather gradually over 13 years following P reduction as total dissolved P (TDP) declined to very low levels (median 10 µg/L) downstream (Sosiak pers. comm). Within 10 km downstream of the effluent input, however, TDP declined initially from a mean summer value of 111 µg/L to 19-24 µg/L and periphyton biomass exhibited no change from the high pre-treatment levels of 300-400 mg chlorophyll a/m^2. The data upstream and downstream demonstrated that if summer TDP consistently averaged < 10 µg/L, maximum periphyton biomass typically averaged less than 100 mg chorophyll a/m^2. Maximum summer biomass averaged approximately 1.4 times mean values.

TDP and periphyton biomass decreased gradually over the 13-year period following treatment with the largest decline occurring after 1989, although this is not apparent from the data summary in Table A-4. The delayed decrease in TDP may have been due to declining recycling from sediments, the TP content of which declined downstream, but not upstream of Calgary (Sosiak pers. comm.).

This extensive data base also indicates that TDP was linked much closer to periphytic biomass than TP, which decreased markedly following treatment upstream (Stier's Ranch). The change was only slight downstream, in contrast to the 50% decrease in TDP (Table A-4). Note that average maximum biomass varied from 77 to 428 over a range of summer mean TP of only 40 to 59 µg/L. Periphytic biomass was also correlated with TDP (r values of -.61 to .70), but not with TP (Sosiak pers. comm.). Sosiak concluded that TDP was a much better indicator of periphytic biomass throughout the river than TP.

An interesting contrast for this case study in comparison with the Clark Fork River involves the reduced frequency of filamentous green algae and lower maximum biomass levels in the Bow River. *Cladophora* was the dominant taxa that extensively covered the bottom substrata and created the nuisance condition interfering with recreational use in the Clark Fork River. In the Bow River, the periphyton was dominated by diatoms, which can be highly visible if biomass is high. Although *Cladophora* was present downstream from Calgary prior to nutrient reduction (Charlton et al. 1986), there was apparently not the high percent cover of filamentous greens that interferes with recreation. Part of the difference in nuisance conditions between the two rivers may be related to the higher summer flows in the Bow. Nevertheless, *Cladophora* and some other filamentous greens did largely disappear after nutrient reduction (Sosiak pers. comm.).

These data indicate that: 1) periphyton biomass in streams and rivers does respond to nutrient reduction, 2) biomass levels below nuisance levels (~150 mg chlorophyll a/m^2) can be attained if P can be sufficiently reduced, 3) sufficient reductions are defined by levels approaching ~10-15 µg/L TDP, and 4) response to nutrient reductions may not occur quickly even in rivers where water exchange is immediate. The gradual reduction in river TDP suggests that there is a long-term, slow release of P stored in (or adsorbed to) bottom sediments, even in rubble-bottom rivers.

In addition, macrophytes (mostly pond weeds) reached biomass levels of > 2000 g/m^2 within 30 km downstream of discharges prior to effluent treatment in 1987, but declined soon after N reduction, reaching levels in 1995-1996 of < ~200 mg/m^2. The cause for macrophyte decline in response to N reduction is not clear, but was hypothesized to be due to increased N limitation at plant roots (Sosiak pers. comm., citing Barko et al. 1991). Nitrogen in the water was never considered limiting to macrophytes or algae, because DIN:TDP ratios were always well above 20:1 by weight even after N removal.

The downstream change in periphyton biomass was simulated with a model that predicts spatial and temporal biomass and nutrient (in this case SRP) concentrations in cobble/gravel-bed rivers during summer low-flow conditions. SRP was not determined in the Bow River, but TDP was converted to SRP using 0.65 X TDP. Of five years suitable for model calibration, a 4-week period in October 1997 was selected (Elswick et al. 2000). Sloughing loss was assumed negligible during this period, as has been observed in laboratory channel experiments during which periphyton is actively growing to a maximum biomass (Horner et al. 1990; Anderson et al. 1999). The model was not verified, because there was insufficient data for some processes, such as grazing, for which a constant value was used (10% of existing biomass/day).

Model simulation compared favorably with actual data (Figure A-16). The largest discrepancy was at Carsland (56 km) where biomass was overestimated by 100%. Biomass was also overestimated at all other sites, but by an average of only 25%. Part of this difference may have been related to P retention in run-of-the-river impoundments located upstream from Carsland and not included in the model. Also, grazing may have been greater than the assumed rate. Grazing rates per unit grazer biomass are available in the literature and could be used in this model if grazer biomass were available. Nevertheless, the model demonstrates the phenomenon of biomass reduction downstream following nutrient reduction at a point source. That is, while P concentrations were still too high to reduce periphyton biomass below the nuisance level (~150 mg chl/m^2), P concentrations and biomass declined downstream and the extent of the decline can be estimated with this model.

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Table A-4. Summary of Phosphorus and Periphytic Algal Data from the Bow River, Canada[1]							
Station	Distance from Headwaters, km	Data type	Data years	Average summer periphytic biomass, mg/m^2	Average maximum summer periphytic biomass, mg/m^2	Mean summer total phosphorus, µg/L	Mean summer total dissolved phosphorus, µg/L
85th St. bridge	254.53	post-treatment	1984-87	27	49	8.5	5.0
		post-treatment	1988-96	71	97	15.1	4.7
STP	279.14						
STP	294.03						
Stier's Ranch	304.81	pre-treatment	1981-82	248	294	147.7	110.6
		post-treatment	1984-87	189	369	52.4	18.9
		post-treatment	1988-96	225	428	59.2	24.2
Bow City	533.78	pre-treatment	1980-82	83	196	54.0	21.6
		post-treatment	1984-87	51	77	40.0	10.5
		post-treatment	1988-96	57	92	37.3	7.0
Ronalane Bridge	625.61	pre-treatment	1980-82	94	148	40.0	15.3
		post-treatment	1984-86	74	111	39.1	10.8
		post-treatment	1996-98	59	111	16.8	7.0

[1]Data from Sosiak, Alberta Environment Protection, personal communication

Contact: Debbi Hart (4304), Headquarters Nutrient Program
United States Environmental Protection Agency
Ariel Rios Building◆1200 Pennsylvania Avenue, NW ◆Washington, DC 20460
hart.debra@epa.gov

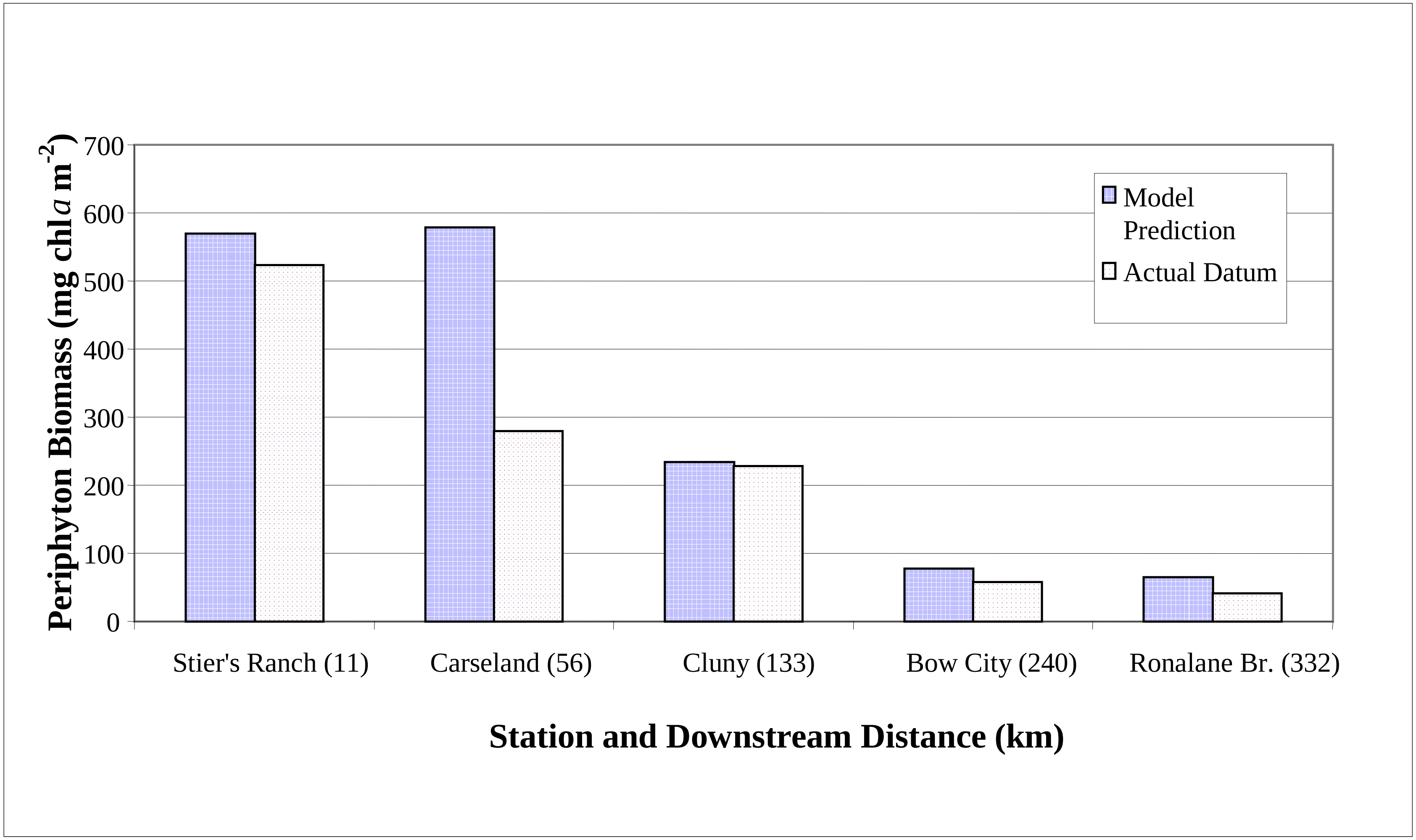

Figure A-16. Model simulation of periphyton biomass during low-flow conditions in the Bow River downstream from the Fish Creek STP, Calgary, Alberta, compared with actual 1997 data.

NUTRIENT CRITERIA DEVELOPMENT FOR DESERT STREAMS—DETERMINANTS OF NUTRIENT DYNAMICS IN SOUTHWESTERN HOT DESERT STREAM ECOSYSTEMS

As a class of pollutants, nutrients are unique from toxicants such as mercury or DDT, in that they have known biological functions. Macronutrients such as nitrogen and phosphorus, are at once biological necessities and, in excess quantity, agents of change to community and ecosystem attributes. As such, great care must be taken in the characterization of nutrient regimes in stream ecosystems. Streams are by their very nature dynamically changing ecosystems that must be studied at ecologically meaningful temporal and spatial scales (O'Neill et al. 1986). The characterization of "ambient" conditions with a few grab samples is inappropriate, if not reckless. The researcher must first learn what limits autotrophic productivity, the major nutrient sources and sinks and how and where nutrient transformations take place in order to make informed decisions and avoid the adoption of water quality standards that may allow or even cause shifts in stream community structure or ecosystem process.

The nutrient regime of streams in general can be complex, however, desert streams present particular complexities not found in more homogeneous, mesic landscape stream ecosystems. Spatial and temporal variability in physical structure, community composition, materials availability and the interactions between these elements strongly control nutrient processes in desert streams. Dent and Grimm (in press) found a high coefficient of variability (as high as 145%) in the spatial distribution of nutrients in Sycamore Creek, Arizona, with coefficients of variation increasing over successional time. Part of this is due to hydrologic variability, in all its temporal, spatial and amplitude scales. However, stream ecosystems are complex in time, space, composition and process, and extend beyond the limit of the wetted surface stream. These ecosystems must be considered as a whole, temporally and spatially, and not as disconnected, stand alone components in order for accurate characterization to take place.

The following discusses a number of the many determinants of nutrient regimes in desert streams. As the subject matter of this section deals with desert streams, and in particular, southwestern hot desert streams, the literature cited will concentrate on research done in those ecosystems. A hierarchical structure (sensu Stevenson 1997) will be used to organize determinants into ultimate, intermediate, and proximate categories. These hierarchical levels operate as interconnected units, with a particular level being limited by constraints imposed by higher levels with processes structured at lower levels (Pickett et al. 1989). Because of the interconnectedness of the differing hierarchical scales, it will be sometimes necessary to "mix" scales in the discussion that follows. Hydrologic variability and its effects on desert stream nutrient regimes is also discussed.

ULTIMATE DETERMINANTS

Ultimately, the stream is a product of its parent geology, catchment configuration and climate. In desert streams, these structural determinants combine to organize processes at lower hierarchical levels forming ecosystems unique from more mesic ecosystems. These desert stream ecosystems consist of four interconnected and interacting subsystems, the surface stream, the hyporheic zone (zone of subsurface flow), the parafluvial zone (lateral sandbars within the active channel) and the riparian zone. These subsystems interact with and are ultimately a product of the geology and the climate/precipitation regime.

Parent geology can play a large role in the availability of nutrients to aquatic ecosystems. Because of the impermeability of desert soils and the low biomass per unit area found in terrestrial desert ecosystems, materials entrained by precipitation events readily move into aquatic ecosystems, assuring an ample supply of nutrients associated with the parent material. Soils of the arid southwest are rich in calcium phosphate (Fuller 1975) and transfer that nutrient readily to stream ecosystems. A survey of 196 sites on 157 streams in Arizona found soluble reactive phosphorus (SRP) and total dissolved phosphorus did not differ significantly among stream types (Fisher and Grimm 1983; Grimm and Fisher 1986a). This uniformity in concentration may be due to solubility equilibria (Stumm and Morgan 1981) and indicate physical rather than biological control of this nutrient. The ample supply of phosphorus coupled with the low overall input of nitrogen from the surrounding landscape has lead to the condition in which nitrogen, rather than phosphorus, is the nutrient that limits primary productivity (Grimm and Fisher 1986b).

The size of the soil particles moving into and staying within the channel can also have a large impact on the nutrient dynamics of streams (Jones 1995). In Arizona desert streams, the unglaciated terrain provides little silt and clay to the stream (Fisher 1986) and the flashy hydrologic regime may cause what little sediment that makes it into the stream to be deposited either laterally or in unconstrained reaches. Valett et al. (1990) found that in Sycamore Creek, Arizona, most sediment ranged in size from coarse sand to fine gravel (0.5 -5.0mm). This paucity of fine sediments allows a relatively high rate of hydrologic conductivity within the hyporheic (Valett et al. 1990), parafluvial (Holmes et al. 1994), and riparian zone (Marti et al. in press [a]).

The hydrologically conductive sediments beneath and lateral to the wetted stream are an important zone of biologically mediated nutrient processes. Jones et al. (1995) observed significant rates of nitrification (mineralization of organic nitrogen to ammonia and a subsequent transformation to nitrate) within the hyporheic zone of Sycamore Creek. This nitrate rich hyporheic water may then exchange with nitrogen poor surface water (Dahm et al. 1987; Valett et al. 1990; Stanley and Valett 1992) where it is an important nutrient source for primary producers and may strongly influence biomass and community composition (Valett et al. 1994).

The parafluvial zone consists of sediments within the active channel but outside the wetted stream. Results from Holmes et al. (1994) indicate that the parafluvial zone can be an area of net nitrification, and increased algal productivity has been observed on the downstream edges of parafluvial sandbars. However, Holmes et al. (1996) found significant denitrification (the reduction of nitrate to nitrous oxide or dinitrogen) potential existed in hyporheic and parafluvial sediments. Conservative estimations by these authors indicated that 5-40% of nitrate produced by nitrification may be consumed by this process.

The riparian zone can also be an important area of nutrient storage and transformation. Denitrification and uptake by vegetation within riparian areas may constitute a significant sink of nutrients within a watershed (Peterjohn and Correll 1984; Pinay and Decamps 1988). Chauvet and Decamps (1989) found denitrification and nutrient retention to be important processes in these areas. Marti et al. (in press [b]) found retention of nutrients in riparian areas to be affected by the length of the interflood period rather than by the magnitude of the flood.

The overall geomorphological structure of the stream, which is a product of geology and climate, determines the pattern of the four subsystems; surface stream, hyporheic, parafluvial, and riparian. This spatial mosaic can have a strong influence on the retention, transformation, uptake, and emission of nutrients (Fisher et al. 1998a).

The climate, while a contributing determinant of desert stream physical structure, is also an ultimate determinant of biological structure and function. Desert streams receive high rates of insolation due to the low amount of shading contributed by the relatively open riparian canopy. This open canopy itself is a product of the low precipitation rates and high pan evaporation rates found in southwestern deserts. Although precipitation rates are low in desert landscapes, this sparse precipitation may contribute substantial amounts of nitrate and ammonium to desert stream ecosystems (Grimm 1992).

INTERMEDIATE DETERMINANTS

Community structure and function are shaped in part by the constraints imposed from the higher geological and climatic scale. Species composition and life history are ultimately products of the physical components of aquatic ecosystems and, in turn, alter physical structure and chemical processing at the ecosystem level. These interactions can significantly affect nutrient dynamics in aquatic ecosystems.

The change in the biotic community that occurs over time after a flood disturbance shapes and changes nutrient processing. The concept of temporal succession has been put forward to conceptualize this temporal change that occurs after a flood disturbance. Fisher et al. (1982) described the recovery of community and ecosystem attributes after flooding disturbance in Sycamore Creek, Arizona, as a model of temporal succession. In these ecosystems, biologically driven nutrient transformation, uptake, and emission can vary significantly over spatial and temporal scales.

Changes in nutrient processing and species composition over successional time can drastically alter the nutrient regime of streams in general, and, due to their open, autotrophic nature, in desert streams in particular. A flood disturbance of sufficient magnitude can scour and shuffle hyporheic and parafluvial sediments, removing attached biomass and essentially homogenizing ecosystem structure and function at the reach level. The emergent physical structure confers organization to the biotic community recolonizing the reach post flood. During the interflood period, biological processes become a progressively more prominent organizational force which, given a sufficient interdisturbance interval, can then confer organization to higher hierarchical levels.

The complexity of nitrogen processing interactions increases with successional time. Dent and Grimm (in press) found nutrient spatial heterogeneity in the surface stream to increase with time post flood. Over successional time, nitrogen uptake length decreases due to increased biological uptake (Fisher et al. 1982; Marti et al. 1997) and retention of nitrogen in the surface stream increases from early to mid successional stages and then declines during late succession (Grimm 1987). The changes seen in nutrient processing length due to disturbance may vary according to the relative resistance of the individual stream subsystem and recovery may vary over successional time due to relative resilience (Fisher et al. 1998b).

Streams by their very nature are spatially heterogeneous. In desert streams, the spatial heterogeneity of nutrients and nutrient processing is manifested in several ways. Hydrologic linkages between the surface stream and the parafluvial and hyporheic ecosystem components vary spatially due to the underlying geomorphological structure of the stream ecosystem. As stated earlier, these are areas where important nutrient processing occurs. The extent and intensity of hyporheic upwelling and downwelling can change or even reverse in response to flooding or drying (Stanley and Valett 1992; Valett et al. 1994) considerably affecting concentrations of nitrate in the surface stream and influencing algal community composition (Valett et al. 1994).

One of the most striking successional events in desert streams is the drying and contraction of the surface stream ecosystem. This phenomena can take place either at large spatial and temporal scales, or at the scale of the reach during a 24 hour period. In Sycamore Creek, as the quantity of water delivered to the surface stream ecosystem begins to diminish after an extended wet period or post flood, drying begins to occur.

During dry periods, the wetted surface stream area can contract as much as eight fold at the scale of the entire basin. (Stanley et al. 1997). At the scale of the individual run, drying may begin at the downstream terminus and continue upstream to the area of hyporheic upwelling at the head of the run (Stanley et al. 1997). This contraction of the surface water ecosystem can strongly affect algal community composition. Nitrogen fixing cyanobacteria species inhabiting the downstream extremity of the drying run are exposed to the atmosphere long before upstream mats of filamentous green algae situated closer to the source of nitrogen rich hyporheic upwelling, potentially altering nitrogen cycling (Stanley et al. 1997). At all scales, drying of the surface stream increases the relative contribution of hyporheic processes to overall ecosystem function (Stanley and Valett 1992).

The increase in the relative proportion of the wetted stream ecosystem occupied by subsurface subsystems could have a profound effect on nitrogen processing and retention. With the decline of surface stream area available to autotrophic production and nitrogen fixation, the nitrogen transformations associated with subsurface flowpaths, nitrification, and denitrification will become proportionally more prevalent (Stanley and Valett 1992). This, coupled with the ample organic carbon available from decaying algae (Jones et al. 1995) and oxygen depletion due to respiration over extended hyporheic flowpaths, could possibly cause subsurface subsystems to become net nitrogen emitters. This emission of nitrogen gas would represent a real loss of nitrogen from the stream ecosystem.

The riparian component of the desert stream ecosystem, while ultimately a product of climate, geology, and catchment configuration, also modifies environmental factors within the ecosystem of which it is a part. At successional and reach scales, the presence of riparian vegetation is a strong determinant of the shape of the surface stream channel. Given a long period between stand destructive floods and ample surface or near surface water, fast growing woody species such as seep willow (Baccharis salicifolia) will progressively impinge on the surface stream. As the surface stream narrows, parafluvial ecosystem components gradually convert to riparian, considerably changing nutrient processing pathways.

The high pan evaporation rate (>300 cm year $^{-1}$) found in hot desert ecosystems coupled with high consumptive water use by phreatophytes (water-loving riparian plants) (1514 L day $^{-1}$) (Blaney and Criddle 1962) can significantly influence the size of a surface stream reach on a diel basis. Large changes in reach length during a 24 hour period have been observed on Sycamore Creek (Stanley pers. com.). This type of short term stress can effectively eliminate desiccation intolerant organisms from the benthic community of large portions of a stream reach.

In contrast to more mesic streams, many desert stream riparian zones occupy a relatively narrow band lateral to an underfit surface stream, providing minimal shading to the surface stream itself. The resultant high rate of insolation to the surface stream favors high rates of primary productivity, high relative water temperatures, and attendant increases in metabolic rates (Busch and Fisher 1981). The elevated rates of primary productivity and metabolism are controlling factors in the short uptake lengths for nitrogen covered earlier in this document.

PROXIMATE DETERMINANTS

The nitrogen cycle, as it occurs within a desert stream, is essentially a biologically driven process. Given the physical organization conferred from higher hierarchical levels, the resultant biotic communities of the surface stream, riparian, parafluvial, and hyporheic control fixation, mineralization, nitrification and denitrification.

Nitrogen fixation by heterocystous cyanobacteria such as Nostoc or Calothryx and diatoms with phycoendosymbionts such as Epithemia sorex may be significant in nitrogen poor desert stream ecosystems. Grimm and Petrone (1997) measured in-situ N_2 fixation rates as high as 51mg N_2 m^{-2} h^{-1}. These rates were high in comparison to published values from more mesic inland systems. In this study, as much as 85% of the net nitrogen flux to the benthos was accounted for by N_2 fixation on five dates for which nitrogen input/output budgets were constructed. Nitrogen fixation may be an extremely important vector of nitrogen into desert stream ecosystems, as between precipitation events, little fixed nitrogen from the surrounding uplands is transported to the stream ecosystem (Grimm and Petrone 1997).

As stated earlier, nitrification, or the biologically mediated oxidation of ammonium to nitrate takes place within hyporheic and parafluvial sediments. Jones et al. (1995) reported mean nitrification rates of 13.1 mg $NO_3 \cdot$ L sediments^{-1} h^{-1} in downwelling zones in Sycamore Creek and Holmes et al. (1994) reported increases in nitrate concentrations in water moving through parafluvial flowpaths. In both studies, the highest rates of biotic activity occurred at the interface where the surface stream infiltrated into hyporheic/parafluvial sediments. This effect suggests the importance of dispersed interfaces in a heterogeneous system (Dahm et al. 1998)

Denitrification is well documented in anoxic environments such as riparian soils (Peterjohn and Correll 1984; Lowrance et al.1984), but in well oxygenated environments such as the coarse sand/gravel hyporheic/parafluvial subsystems found in desert streams, the occurrence of denitrification is somewhat of a conundrum. Holmes et al. (1996) investigated denitrification potential in hyporheic, parafluvial and riparian sediments and found field measured rates in excess of 150 mg $N \cdot m^2 \cdot h^{-1}$ at the stream/parafluvial interface. This study also found the highest rates of biotic activity (denitrification) at the point of infiltration at the surface-water-sediment interface.

Despite the low overall availability of the most probable limiting nutrient in southwestern hot desert streams, nitrogen, high mid-summer instantaneous standing crops of algae (191 mg $^{-1}$ m^{-2} chlorophyll a) have been measured in Sycamore Creek (Busch and Fisher 1981). The spatial distribution of algal standing crop has been linked with areas of hyporheic upwelling and downwelling. Valett et al. (1994) found significantly higher areal concentrations of chlorophyll a in upwelling zones when compared to areas of downwelling. These high quantities of algal biomass and the associated autotrophic uptake of nitrogen are the most probable cause for the declines in nitrogen concentrations of surface water found downstream of spring sources (Grimm et al., 1981) and the lower concentrations found at points of downwelling (Valett et al., 1994).

Organic carbon released as a result of autochthonous primary production has been hypothesized as the energy source utilized in nitrification (Holmes et al., 1994; Jones et al., 1995) and denitrification (Holmes et al., 1996). However, allochthonous input of organic matter from riparian leaf litter was found to play an insignificant role in nitrogen dynamics in Sycamore Creek (Schade and Fisher 1997).

Macroinvertebrates may also significantly affect nitrogen processing in desert streams. Grimm (1988) found that during a 20 day successional period, collector gatherer invertebrate standing stock increased from 32,000 to 108,000 individuals $\cdot$ m^2 . Twenty seven percent of the nitrogen ingested by the collector gatherers during this period was converted to biomass, of which only 26% (7% of total ingested nitrogen) remained in the stream as macroinvertebrate biomass. One percent of collector gatherer biomass was lost to the surrounding upland ecosystem due to the emergence of adults, 19% was lost to mortality and 9-31% was excreted as ammonia. The transformation of organic nitrogen to ammonia may be particularly significant, as the ammonia form of nitrogen is readily taken up and utilized by primary producers or available for utilization in nitrification/denitrification transformations.

HYDROLOGIC VARIABILITY IN ARID LANDS STREAMS

While hydrologic variability is an important consideration in the development of nutrient standards for all streams, the spatial and temporal heterogeneity found in arid regions, the stark contrast between wet and dry, brings this variability into sharper relief. 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 may make the use of broad generalizations to explain nutrient regimes impossible.

When analyzing stream nutrient regimes in the context of hydrologic variability, there is a continuum of spatial and temporal scale (sensu Pickett et al. 1989; Fisher and Grimm 1991) beyond and including discreet disturbance flows which must be considered. Ecologically important spatial and temporal scales can vary from that of a discreet patch at a single point in time, to the fluvial geomorphological and climatic factors determining the physical structure of an entire catchment. These spatial and temporal scales exist as nested hierarchies, with structure at smaller scales being influenced by higher scales (Pickett et al. 1989).

Use of a coherent hierarchical schema can confer useful organization to the analysis of nutrients and primary productivity. The heterogeneity of benthic algal assemblages is determined at several hierarchical levels, with proximate and intermediate determinants such as nutrient regime and flow stability being governed by the ultimate determinants of climate and geology (Stevenson 1997). It is important to consider the determinants of structure and function at different scales when designing ecological studies.

In arid landscapes, stream ecosystems are more dynamically linked with the surrounding upland ecosystem than streams in more mesic regions. This close linkage is due to the higher percentage of uninterrupted vectors of runoff and entrained materials from the surrounding uplands to the aquatic ecosystem. The extensive riparian buffers and dense upland terrestrial vegetation found in more mesic ecosystems are largely absent in spatially intermittent and ephemeral watercourses. The sparse vegetative cover (5-50%; Barbour et al. 1980) and high orographic relief found in the upland terrestrial catchments promote increased rates of short-term, sheetflow runoff during intense precipitation events, leading to larger, more rapid movements of precipitation and entrained materials into watercourses (see Graff 1988). This "spiky" oscillation in the hydrograph is then transferred downstream to the more perennial sections of a stream.

In desert streams, 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 land lands streams (Davies et al. 1994). The aquatic ecosystems structured by these often catastrophic and always chaotic flow regimes exhibit spatially and temporally heterogeneous structures and functions (sensu Thoms and Sheldon 1996) which may not allow the application of nutrient criteria derivation techniques applicable to more homogeneous environments.

Short-term disturbance of small spatial extent may cause considerable alteration in the chemical and biological structure of a stream. Flooding may scour the benthic surface, reset the stream ecosystem to an earlier successional stage (Fisher 1983) and transport large, short-term pulses of nutrients (Fisher and Minckley 1978). Drying of a surface stream reach due to diel changes in evapotranspiration can strand algal mats (Stanley pers. com.) causing a stress disturbance (sensu Pickett et al. 1989). Recovery from these types of small scale disturbances may be rapid in ecosystems where the biota is disturbance adapted (Gray 1981; Grimm and Fisher 1989) and when observed in the context of larger spatial and temporal scales, these types of disturbances may represent normal oscillations in a steady state equilibrium.

Often, hydrologic regimes that effect a particular ecological structure or function may exist at spatial and temporal scales that can only be measured using multiple measurements over space and time. While the flood pulse itself may cause considerable disturbance to a stream ecosystem, the entire hydrologic regime must be considered biologically significant (Poff and Ward 1989). Variability in rates of rise and fall, timing, duration, magnitude and frequency of the flood pulse can have a significant effect on the biota of a stream (Puckridge et al. 1998). At slightly longer temporal scales, it is the relatively short interval (1.5 year) bankfull discharge that forms and maintains the physical structure of the wetted channel (Dunne and Leopold 1978) rather than the catastrophic long return interval flood.

During interflood periods, flow regimes are comparatively stable as precipitation stored within the watershed moves into the stream. The stable flow allows the control of ecosystem state variables, such as primary productivity, to shift from disturbance to morphometric/biotic controls. If the interflood period is of sufficient duration, a phase shift from wetted surface to dry occurs as flow from the watershed diminishes (Fisher and Grimm 1991). During this interval, primary productivity is a partial function of the number of days post flood (Fisher 1986; Fisher and Grimm 1988). Characterization of the interflood period is an important tool which may allow the researcher to locate the point in successional time when indexing biological data for inter or intra-stream comparison.

One portion of the hydrologic regime that is often overlooked is drying. Drying disturbance, or more specifically the contraction and fragmentation of a stream ecosystem, occurs as a spatially or temporally intermittent stream recedes after a wet period. Differing reach types (e.g., riffles, runs, constrained, unconstrained) respond to this contraction and fragmentation differentially (Stanley et al. 1997) and hyporheic, or subflow processes may come to dominate as a larger portion of the wetted volume of a stream is subsurface. (Stanley and Valett 1992; Valett et al. 1990). Drying is likely to be an important determinant of biological pattern and process (Stanley et al. 1997; Stanley and Boulton 1995), especially in streams where the dry period and extent may be greater than the wet.

Longer-term hydrologic/disturbance regimes are also an important consideration. Decadal climate variability such as the El Niño and La Niña phenomena can cause large, prolonged fluctuations in stream flow (Molles and Dahm 1990). This long return interval climate variability and the attendant change in short-term weather patterns can significantly affect the structure and function of aquatic ecosystems. The establishment, maintenance and species composition of riparian associations are strongly dependant on the seasonality, periodicity, duration, sequentiality and magnitude of storm events and subsequent flow regimes (Baker 1990; Stromberg et al. 1991).

Large, long return interval disturbances can also greatly alter the physical structure and pattern of watercourses at greater than reach scales. These large alterations may affect the physical equilibrium of the watercourse in several ways. If the system is stable or dynamically stable, internal feedback mechanisms will cause physical values such as bed load transport to return to original following a disturbance. If a system is unstable or metastable, the system may adjust to a new value causing changes in channel pattern and shape (sensu Chorley and Kennedy 1971).

In the event a large scale, destructive flood event significantly restructures a stream, changes may occur in mean particle size, pattern of reach types or the ratio of the different stream ecosystem components (surface, riparian, parafluvial, hyporheic). These changes in physical structure may significantly alter the prevailing nutrient regime (sensu Fisher et al. 1998a).

In order to properly characterize the nutrient regime of a stream ecosystem, the flow of water, surface or subsurface, flood or base flow, wet or dry must be considered at ecologically significant temporal and spatial scales. It is also important that the researcher 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 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 discreet 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, it is important to recognize that all stream ecosystems posses complexities that will only yield to proper inquiry.

CONCLUSIONS

The characterization of nutrient dynamics in streams with high temporal and spatial variability, such as southwestern hot desert streams, may prove to be difficult without a commitment to addressing questions at the appropriate ecological scale. Variability in the products of climate and geology; precipitation, flow regime and physical structure, define the limits of community composition and nutrient processing. The predictive power of any model designed to characterize nutrient dynamics without considering the ultimate determinants is extremely limited. Conversely, relying on coarse generalizations generated at the spatial and temporal scale of the ecozone to predict process at the scale of the reach is also inappropriate. Nutrient dynamics must be characterized and nutrient water quality standards developed considering the constraints and processes dictated at the different and interacting scales.

STEPS FOR CHARACTERIZING THE NUTRIENT REGIME OF DESERT STREAM SYSTEMS

Selecting Index Sites and Periods

As in any investigation, the researcher must remove or account for as many of the differing sources of variability as possible prior to gathering nutrient data. In streams with high spatial and temporal variability, the best case scenario would be to characterize the entire stream, source to terminus, in space and time. While this may be the most scientifically sound methodology, it is infeasible in all but the smallest basins. An alternative is to carefully choose and compare index sites (and periods) from which reasonable extrapolations can be made. This can be done using a similar hierarchical approach to that outlined above, however, extrapolation beyond the specific index is risky. The number of sights required to accurately characterize the nutrient regime in a stream type will vary with the complexity of that nutrient regime.

First, at the largest spatial scale, the position of the stream reach within the watershed must be determined. The areal extent of the basin above the sample reach, the watershed aspect (orientation to weather patterns), mean stream gradient, parent material(s) and stream order are all attributes that should be considered. At the largest temporal scale, the time since the last flood that restructured all of the stream compartments (surface, hyporheic, parafluvial, and near stream riparian) should be determined as well as any long-term fluctuations or trends in the hydrograph.

At the scale of the sample reach, the researcher should consider the landscape setting surrounding and upstream of the sampling point. It is important that the sample reach and surrounding landscape be consistent with that found for a reasonable distance upstream. This distance will depend on stream velocity, with greater distances being required in faster flowing streams. The following is a list of the major elements of a sample reach that should be addressed to characterize nutrient regime and increase data conformity between sampling sights:

Physical/Structural Elements

- altitude
- terrestrial vegetation association
- terrestrial land use
- Rosgen stream type
- physical setting - constrained or unconstrained? (canyon or open plane)
- reach gradient
- solar aspect - is the sun blocked by canyon walls at times during the day or year?
- riparian association - including understory plants
- riparian cover percent
- riparian canopy density - stream shading
- stream discharge and velocity

- substrate particle size distribution
- estimated subsurface compartment volume (hyporheic, riparian and parafluvial)
- location of upwelling and downwelling zones
- water temperature

Temporal Elements

- season
- photoperiod
- time since last flood
- flow regime the previous 30 days
- temperature regime the previous 30 days (air and water)

Chemistry (other than TN and TP)

- NH_3/NH_4, NO_3, SRP
- CO_2
- potassium (K), calcium (Ca), magnesium (Mg), sulphur (S), boron (B), chlorine (Cl), copper (Cu), iron (Fe) manganese (Mn), molybdenum (Mo) and zinc (Zn)
- O_2 - dissolved and % saturation
- pH - field measured
- electrical conductivity
- total dissolved solids
- total organic carbon
- turbidity - field measured
- total suspended solids
- volatile suspended solids

Biological Elements

- algal community composition
- benthic chlorophyl *a*
- benthic organic matter
- benthic community productivity and respiration

When taking physical, chemical or biological samples, it is extremely important to choose the sampling point(s) and times carefully in order to accurately characterize the element in question for a particular reach at a particular time. Multiple samples taken within the reach and analyzed separately is the preferred method, however composite samples, or carefully taken grab samples can work well. The researcher should avoid or account for samples taken in areas of the stream that differ from the main body. Anoxic backwaters, upwelling or downwelling zones, highly aerated areas below waterfalls and other sections that differ physically, chemically, or biologically from the main stream, usually account for only a small portion of total stream area but may contribute significantly to materials processing. Rather than characterizing these sections individually, a point can be chosen that integrates these areas into the greater flow. A fast "run" with relatively uniform flow, biological, and bank characteristics for 20 meters that has neutral subsurface hydraulic head may be a good selection. Insolation rate (solar energy per unit area per unit time) and diel curve should also be considered, although a viable alternative would be careful consideration of time of day, time of year, riparian shading, and cloud cover.

It is extremely important that ***enough*** data be gathered to characterize a nutrient regime. While the ancillary data requirement may seem large, lack of one or more of these data points may preclude accurate interpretation of the nutrient data.

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Contact: Suesan Saucerman, Region 9 Nutrient Coordinator
United States Environmental Protection Agency
75 Hawthorne Street ◆San Francisco, CA 94105
saucerman.suesan@epa.gov

APPENDIX B. METHODS OF ANALYSIS FOR WATER QUALITY VARIABLES

Several methods used to analyze certain water quality variables are discussed briefly in this section. Additionally, relevant publications are referenced as appropriate. Some tips on analysis will also be presented to help users to be efficient in determinations. As with any environmental analysis, the most efficient strategy when learning a new technique is to visit a laboratory where it is routinely performed. The methods that will be discussed include those for light transmission; total suspended solids; total nitrogen, total phosphorus, and dissolved inorganic nitrogen and phosphorus; conductivity; chlorophyll *a* and ash free dry mass (AFDM) for algal biomass; and microscopic identification to determine the algal taxa present. Brief discussion of secondary indicators of eutrophication (algal production, dissolved oxygen concentrations, limiting nutrients and macroinvertebrates) will also be presented. As discussed above, determination of other factors such as hydrology, geology, soil characteristics may also be necessary.

PHYSICAL WATERBODY CHARACTERISTICS

LIGHT TRANSMISSION

Total suspended solids and dissolved humic compounds can absorb light and limit algal biomass. As periphyton biomass increases, particulate matter sloughed and eroded from the periphyton also increases, reducing transparency. Light transmission measurements may be required. Light transmission can be measured using turbidity meters (transmissometers or turbidometers). Use of these meters is described in Standard Methods (APHA 2000). A quick method for determining light transmission is use of a black disk and an underwater periscope (Davies-Colley 1988). The path length for transparency is measured horizontally in shallow streams, as opposed to vertically in lakes, reservoirs and deep rivers or estuaries. The vertical water column in relatively clear-water, gravel/cobble bed streams/rivers is usually insufficient to determine Secchi disk depth.

LIGHT AVAILABILITY

Light availability can be measured directly with a light meter as photon flux density (μmole quanta m^{-2} s^{-1}), but such measurement vary temporally. Measures of % canopy cover, TSS and average water depth, light transmission with a black disk or periscope, and stream direction provide measures of relative availability of light which can be related to a regional average. Light intensity varies so much during a day or with weather from day to day that indicators of relative light intensity may be a more precise indicators of light availability than one-time measurements of light intensity.

Light availability for photosynthesis can be reduced by the amount of total suspended sediment (TSS) in the water column, light attenuation caused by dissolved compounds, river depth, and channel shading. In addition to scouring algae, TSS also attenuates light to benthic algae. Dissolved organic humic compounds can absorb light, and if they are present in high enough concentrations, they can prohibit algal growth. Similarly, forest canopies can shade stream channels (Dodds et al. 1996). This shading can lead to rivers with relatively high nutrient concentrations, but with negligible sestonic or benthic algal biomass. In such cases, nutrient control may have few immediate benefits.

If there is seasonally high TSS or shading (e.g., deciduous forests), the high nutrients may cause excessive periphyton algal biomass only during certain times of the year. An example of this would be streams where snow melt is common in the spring; this could lead to high levels of TSS and low algal biomass, but during stable flows in summer, low TSS and high algal biomass. Finally, very deep channels will not usually have excessive algal biomass except at the margins, since limited amounts of light reach most of the bottom (Allan 1995), and sestonic algae are mixed frequently throughout the water column, which reduces available light while increasing respiration (Welch 1992). Therefore, net productivity (gross production minus respiration) decreases with depth of mixing.

FLOW AND VELOCITY

Flow and velocity measurements are important for determining nutrient loadings, concentrations, and distributions. Flow volume or discharge is easily calculated based on stream channel area and velocity. Velocity is typically measured with a stream gauge or current meter. See http://water.usgs.gov/pubs/circ1123/collection.html for more details.

TOTAL SUSPENDED SOLIDS AND VOLATILE SOLIDS

It can be useful to quantify total suspended solids because of their effect on light attenuation, and the determination of volatile solids may be of interest to determine if the total suspended solids are from organic sources. The methods for total suspended solids and volatile solids are presented in Standard Methods (APHA 2000).

TEMPERATURE

Temperature can be an important variable in determining alkalinity, saturation, and rates of chemical and biological reactions. It is a simple but useful measurement to include in a sampling regime. Methods for temperature measurement in the field and laboratory are described in Standard Methods (APHA 2000).

CHEMICAL WATERBODY CHARACTERISTICS

NUTRIENT ANALYSES

Nutrient analyses are the most important indicators for determining sources of nutrients and for monitoring the effectiveness of control programs. The analyses for soluble reactive phosphorus and dissolved inorganic nitrogen are mentioned first because they are the forms available for algal uptake and because they are the forms determined (after digestion) for total nitrogen and total phosphorus. In general, determinations of nutrient concentrations by field kits are only adequate to identify potential problems. If many nutrient assays are required to define the problem accurately, laboratory procedures are more cost effective and have greater sensitivity.

In nutrient-poor systems, levels of dissolved inorganic nutrients are generally near the limits of detection of the assays used. For example, phosphate levels in excess of 30 µg/L saturate uptake by algae, but this is the lower limit of detection in many laboratories. Care must be taken that the assay procedure used matches the question being asked.

The assay for dissolved or soluble reactive phosphorus from Standard Methods (APHA 2000) should be followed. A common source of contamination that causes problems with soluble reactive phosphorus analysis is the use of phosphate containing detergents to wash laboratory equipment. It is good practice to use phosphate-free detergents in the laboratory for this reason. Another important problem is the source of low-phosphorus water for dilution and blanks. Absorbance should be very low (0.001-0.003 absorbance units per cm) for such purposes.

The soluble reactive phosphorus assay does not determine only phosphate, because the chemicals in the assay react with some dissolved organic compounds that contain phosphorus other than ortho-phosphorus. It has been demonstrated that increased phosphorus deficiency in algae in natural systems leads to a lower percentage of biologically available phosphate in the chemically determined soluble reactive phosphorus (e.g., Dodds 1995). Unfortunately, the identity of the remaining fraction of soluble reactive phosphorus is unclear, so soluble reactive phosphorus values from natural waters are difficult to interpret, unless the values are fairly high (e.g., above 10 mg/L). In some such cases (e.g., groundwater or wastewater input), a large portion of the soluble reactive phosphorus may actually be in the form of phosphate, and the assay will provide a fairly accurate measure of the phosphate immediately available for algal consumption. The soluble reactive phosphorus assay is particularly useful to determine phosphate in sewage (where most soluble reactive phosphorus is phosphate) and to analyze digested samples for total phosphorus. A method for the analysis of PO_4^{3-} (orthophosphate) is also available in Standard Methods (APHA 2000).

Analysis of ammonium is straightforward with the phenate method (APHA 2000). Note that ammonium (NH_4^+) is the ion that identifies the available nutrient, and ammonia (NH_3) is the gas, known as unionized ammonia, which is the fraction that can cause toxicity. Contamination of ammonium assays can occur from scratched glassware and airborne ammonia gas, which can come from smoke (tobacco and otherwise), cleaning products with ammonia, and newly cut grass. Care should be taken to avoid these potential sources of contamination.

Nitrate is commonly measured by reduction to nitrite in a copper-cadmium reduction column (APHA 2000). Nitrite can be analyzed alone to correct estimates of nitrate, but in most studies of streams, nitrite is assumed to be a relatively small fraction of nitrate and as such is not accounted for. Cadmium is toxic and difficult to handle and dispose. Some packaged nitrate kits use cadmium pillows that are added to the sample. Appropriate precautions for handling and disposing of samples are recommended if these kits are used. Other (e.g., hydrazine) nitrate techniques may be more prone to interference or reduced efficiency. Automated analysis methods with segmented flow autoanalyzers are commonly used to speed processing and maintain sensitivity. Ion chromatography can be used successfully for nitrate determinations, but it should be kept in mind that this method is not sensitive enough for nitrate values typical of many moderately productive systems.

Total nitrogen and total phosphorus require digestion to dissolved inorganic forms before analysis. There are a number of available techniques. An important point is that the efficiency of digestion of organic materials varies with procedures and waters being analyzed. Regardless of the procedure chosen, solutions with known concentrations of organic compounds (e.g., urea for nitrogen, ATP for phosphorus) should be added to natural water samples in known concentrations and analyzed to check for complete digestion.

Persulfate digestion is commonly used for total phosphorus. This procedure can be modified to oxidize organic phosphorus to phosphate, as well as organic nitrogen to nitrate (Ameel et al. 1993). Careful attention to pH of the samples is necessary in these digestions (the digest must remain alkaline for nitrogen digestion, but if too much persulfate is used, it may not become acidic later in the digestion and incompletely decompose the phosphorus) and appropriate concentrations of fresh reagents should be used to allow for complete digestion of both organic nitrogen and phosphorus.

Persulfate digestion converts all forms of nitrogen except N_2 gas to nitrate. If nitrate analysis is not easily accomplished in the laboratory, it may be desirable to use a Kjeldahl digestion procedure (APHA 2000) for total nitrogen analysis. In this procedure, all nitrogen forms but nitrate, nitrite and nitrogen gas are digested to ammonium. If this procedure is employed, it is still necessary to analyze for nitrate and nitrite to determine total nitrogen. Because the sensitivity and accuracy of the cadmium reduction method for nitrate are greater than analyses for ammonium, and the toxicity and corrosiveness of the digestion procedure are less, persulfate digestion and nitrate analysis is usually preferred to Kjeldahl.

Analyses for TN, TP, phosphate, and nitrate can also be used to calculate water column N:P ratios.

CONDUCTIVITY AND PH

Conductivity may serve as a first indicator of total nutrients (although it indicates total ions which are much more abundant than, and not always closely correlated to, nutrients), and pH may be of interest as a variable indicating impairment. Both of these analyses are most easily accomplished with electronic probes. Refer to Standard Methods (APHA 2000) for the particulars of the analysis.

DISSOLVED OXYGEN

Analyses of dissolved oxygen, for measurements of primary production and determination of low oxygen demand should be done with a titrametric method or polarographic sensor. The titrametric is more accurate, but more time consuming. Standard Methods should be consulted for these analyses (APHA 2000). If diurnal measurements of oxygen are performed, procedures outlined by Marzolf et al. (1994) should be followed for small streams.

ORGANIC CARBON

Analysis of organic carbon (dissolved) may be problematic because incomplete digestion of dissolved organic carbon is common. This has been most thoroughly investigated for marine samples (Perdue et al. 1993). However, similar problems have been documented for freshwater samples (Kaplan 1992). High temperature catalyzed analyzers provide more complete digestion and generally yield reliable results.

ALGAL AND PLANT ATTRIBUTES

COLLECTION OF ALGAL SAMPLES

The choice of methods for sample collection is dependent upon the intent of algal sample analysis. These methods are reviewed in the Revised Rapid Bioassessment Protocols for Streams and Rivers (Stevenson and Bahls 1999), so only a brief overview will be presented in this document. Sampling for assessments of the biotic integrity of algal assemblages should be more thorough and extensive spatially than sampling for algal assessments of water quality. Thorough assessments of biotic integrity would call for multihabitat sampling over large reaches of the stream to find as many species and habitats within the stream or river as possible. Targeted habitat sampling (most commonly samples of algae from rocks in riffles) can provide collections that provide indications of biotic integrity or water quality. A third major alternative is to the use of artificial substrata that have the advantage of controlling variability among streams due to substratum type, but the disadvantage of having to visit the field two times (to place and retrieve substrata) and the concern that non-natural assemblages are being sampled. Targeted habitat sampling is usually recommended, is employed by most State programs, and is known to be successful. Efforts should be made to sample more than one riffle, particularly if an important goal of sampling is to assess benthic algal biomass in a stream.

The collection of algal samples can be a complex exercise due to the variability of stream features such as depth, substrata, flow velocity, and bottom characteristics. Holding some of these variables relatively constant by selecting a habitat zone with a narrow range for these variables was suggested earlier. Another approach is using artificial substrata which are easier to sample than natural substrata but which have several drawbacks. Artificial substrata are more likely to be vandalized, and they often tend to alter the flow regime around them resulting in silt deposition. The use of artificial substrata limits the ability to move to a different area where conditions are more acceptable, as can be done when using natural substrata. Perhaps the biggest drawback of artificial substrata is their inability to promote colonization by certain forms of algae, especially the massive filamentous forms. This issue is discussed further below in connection with the best methods of sampling various algal growth forms.

While there is a great variety of algal taxa, there are two main growth forms of algal communities: thin biofilms and long filaments. Many single-celled and colonial forms of attached algae appear to the naked eye as a biofilm of slippery, gelatinous material (often referred to technically as slime) on river rocks. This material can be easily sampled by using a template method.

Template Method

A template is a piece of flat, flexible, waterproof material in which a window of about 2.5 cm to 5 cm per side is cut. This template is placed in the center of the upper surface of a rock collected from the sample site, and a razor blade is used to scrape together all the material in the window. The material is then placed in a small water tight container (snap-shut plastic petri dishes, vials, or a piece of aluminum foil), and stored on ice in the dark until frozen.

This procedure is greatly facilitated by selecting smooth rocks. To avoid bias, sample points should be selected randomly. Then, rocks are selected blindly until one is chosen that is between 10 and 20 cm (in some regions of the country one is allowed to take a quick look for snakes first). If the rock's surface is too rough to sample, it should be replaced and the process continued until a rock of the right size and smoothness is selected.

The biofilms sampled by the above method form fairly quickly on artificial substrata and often the thickness and composition of this film is quite similar to that on nearby natural substrata in a matter of weeks (Watson unpublished). However, some of the more complex attached algae, most noticeably *Cladophora glomerata*, attach to rocks using a basal holdfast cell which supports a long filament. *Cladophora* holdfasts often survive short exposure and drying out and the scour that removes the filament. The holdfasts spread over the rock and support more massive growths in subsequent years. After several years of flows that are too low to dislodge and roll the river rocks over, *Cladophora* may take the form of massive tangled branched filaments streaming several meters long. Since the massive growths take several years to develop, they can not be produced on artificial substrata which are likely to wash away during spring floods. Hence, such growth forms must be sampled from natural substrata. An example of a method for sampling from natural substrates is the hoop method.

Hoop Method

The template method does not work well for sampling the massive growth form (long string filaments) mentioned above. It is possible to be standing in a sea of waving *Cladophora* and pick up a random rock that has no *Cladophora* on it, or that has a tangled mass hanging by a few threads a few inches from the rock. The preferred way to sample such a growth form is to place a heavy metal hoop about 0.3 to 0.5 meter in diameter on the bottom of the stream (at the randomly selected point) and collect all the filamentous material inside the hoop. This often involves cutting the filaments around the hoop and picking up the filaments and rocks inside the hoop. The collection should be brought to shore in a tub where the filaments should be removed from the rocks. Razor blades and paint scrapers work well; hack saws are generally unnecessary. Wrapping these large samples in aluminum foil will facilitate the drying, weighing and ashing process. The collection of 10 to 20 replicates of such samples at a series of high biomass sites will represent a large volume.

Freezing samples not only helps preservation, but cells are ruptured, facilitating chlorophyll extraction. Samples collected by templates should be frozen at -10°C on return to lab, and analyzed for chlorophyll *a* and AFDM within 2 weeks to a month. Laboratory methods for analyzing algal biomass for chlorophyll and ash free dry weight (AFDM) are discussed below. The same sample can be analyzed for both chlorophyll *a* and AFDM. After chlorophyll analysis, the extracted sample is poured into an aluminum weigh boat, the solvent is evaporated, and AFDM analysis is performed on the boat. This facilitates determining the chlorophyll to AFDM ratio on these samples.

Due to the abundance of material collected using the hoop method, it is not possible to extract all the chlorophyll from these samples. Hence, only small subsamples of each large sample are analyzed for chlorophyll and AFDM, and the large samples are analyzed for AFDM only. Their chlorophyll content is estimated using the chlorophyll to AFDMratio determined from the subsamples. These samples are handled as follows: 1) before freezing the samples collected by the hoop method, take each sample and spread it out; 2) collect many tiny subsamples from all over this bulk sample; 3) chop and mix the subsamples; 4) make at least 4 replicate composite samples from this well mixed pile; 5) place these in small containers and process as you do the template samples; 6) analyze the remaining bulk sample for AFDM; and 7) use the chlorophyll to AFDM ratio of the small composite samples to estimate the chlorophyll in the bulk sample (consider the variability in the chlorophyll/AFDM ratio of the composite subsamples as well as the variability in biomass of the large samples).

COLLECTION OF MACROPHYTE SAMPLES

Macrophyte sampling is commonly performed 1) to qualitatively assess the distribution of vegetation in an area or 2) to quantitatively measure primary productivity (gauged by changes in biomass). Caution must be taken when sampling macrophytes for biomass determination to ensure that the appropriate portions of macrophytes (above and below ground) are collected. (Macrophytes may have up to 90%

underground biomass.) After collection, macrophyte samples may be dried and combusted to determine AFDM in a manner comparable to that for algal samples. Macrophyte sampling and biomass determinations are discussed in Wetzel and Likens (1991).

ALGAL BIOMASS - % COVER OF BOTTOM BY NUISANCE ALGAE

Methods have been described in the literature (e.g., Sheath and Burkholder 1985) to estimate algal biomass in the stream by visual observation. In the Revised Rapid Bioassessment Protocols for Rivers and Streams (Stevenson and Bahls 1999), a rapid periphyton survey is described that provides an in-stream assessment of algal biomass. The technique is simple and can be used by professionals or volunteers with little training. Two steps are involved as the stream bottom is observed at multiple sites (usually >9) through a viewing bucket (clear-bottom bucket submerged in stream for clear observation of the stream bottom). First, percent cover of filamentous algae over the stream bottom is assessed. Then thickness and percent cover of microalgae is assessed. A ranking system is used to quantify thickness of microalgal mats. The advantages of this rapid periphyton survey are that it allows for rapid assessment of algal biomass, particularly filamentous algal green biomass, and it covers large regions of the stream (thus accounting for the great spatial variability in algal biomass).

CHLOROPHYLL *a*

The most commonly used determinant of benthic algal biomass is chlorophyll *a*. Chlorophyll *a* is often a superior indicator of biomass compared to determination of AFDM because non-algal material can contribute to biomass. Chlorophyll *a* is used because it occurs in all common photosynthetic organisms. Other forms of chlorophyll can inflate estimates of algal biomass, because the amount per cell can be more variable. In addition, counts of algal cells and biovolume are often used as a determinant of biomass. These counts are time consuming and require taxonomic expertise, and thus are rarely done and will not be considered here. The general methods for biomass determination are well described by Steinman and Lamberti (1996) and Stevenson (1996); the interested reader should consult these references and others cited herein.

Chlorophyll is determined in seston on filtered material and from benthic material either from cores, artificial substrata, or scraped and extracted substrata. In general, artificial substrata yield higher chlorophyll *a*/AFDM values than natural substrata (Dodds et al., unpublished), and this should be kept in mind when selecting the method to be used. However, measurement of area and extraction of pigment is easier with artificial substrata.

Chlorophyll analyses without an acidification step to correct for chlorophyll degradation products (phaeophytin correction) are occasionally encountered. This acidification is essential for periphyton because dead cells that contain phaeophytin can remain in the assemblage, and lead to biomass overestimates. A fluorometric method with narrow band filters that correct for phaeophytin but omit the acidification step was recently introduced (Welschmeyer 1994).

Determination of phaeophytin concentrations may be useful not only for correcting chlorophyll *a* concentrations, but also as an indicator of periphyton degradation. Wetzel and Likens (1991) give a method for determining both chlorophyll *a* and phaeophytin concentrations. The ratio of chlorophyll *a* to phaeophytin gives an indication of periphyton growth and activity.

Generally, a ratio of 9:1, acetone:water, is used as an extractant. We have found that hot 90% ethanol extraction (Sartory and Grobbelaar 1984) offers some advantages. Primarily, material need not be scraped from the substratum, and grinding of the sample is not required. Rather, the entire sample of substratum and periphyton is placed in a heat resistant (autoclavable) plastic bag with extractant and heated to 80 ºC for 5 min. Ethanol fumes are also less noxious than acetone fumes.

The preferred procedure is to use a spectrophotometer to read absorbance, because the relatively dense solutions of extracted chorophyll are common for periphyton samples. Very dense solutions of chl must be avoided for spectrophotometry and fluorometry to prevent analytical errors; the problem is of greater concern in fluorometry. In spectrophotometry, solutions of greater than 1.5 absorbance units per cm at 665 nm should not be analyzed. Fluorometric analysis should not be attempted with samples having more than 0.5 absorbance units per cm at 665 nm. Dilution with extractant can bring samples to within the appropriate absorbance range.

AFDM AND ALGAL CELL BIOVOLUME

Methods for AFDM and algal cell biovolume are covered by Steinman and Lamberti (1996) and Wetzel and Likens (1991), respectively. Ash-free dry-weight values have been used in conjunction with chlorophyll a as a means of determining the trophic status (autotrophic vs. heterotrophic) of streams (Weber 1973). The Autotrophic Index (AI) is calculated as:

AI = AFDM (mg/m2) / chlorophyll *a* (mg/m2).

As suggested before, these should be relied upon as supplementary methods, and the large degree of time required for biomass determinations by cell counts and biovolume estimates should be considered.

ALKALINE PHOSPHATASE ACTIVITY

Analysis of alkaline phosphatase activity (APA) is used to determine phosphorus limitation in algae. Alkaline phosphatases are enzymes produced by algae to break down organic phosphorus compounds and release bioavailable (PO_4) P (Steinman and Mullholland 1996). Studies have shown that lower levels of P result in higher levels of APA and vice versa (Klotz 1992). The most common method for APA analysis is a fluorometric method described by Hill et. al. (1968).

ALGAL SPECIES COMPOSITION

Different methods can be used to assess algal species composition depending upon the objective of the assessment (Whitton et al. 1991; Whitton and Rott 1996; Lowe and Pan 1996; Stevenson 1998; Stevenson and Pan 1999; Stevenson and Bahls 1999). For example, if the objective of the assessment is to determine if nutrient conditions meet a drinking water use, then analysis of all algae in samples may be desirable to determine if taste and odor algae are present. If the objective is to get an indication of nutrient conditions, trophic status, or biotic integrity, then analysis of species composition of diatoms only may be sufficient. The latter is less time consuming that an analysis of all algae in samples. The methods for analysis of algal species composition in samples can be found in Standard Methods (APHA 2000), the Revised Rapid Bioassessment Protocols (Stevenson and Bahls 1999) or in Lowe and LaLiberte (1996).

Although time consuming, it may be desireable to determine the types of algae present that are thought to be creating problems. The methods necessary for such determinations are described in Lowe and LaLiberte (1996) and in Standard Methods (APHA 2000). In general, taxa determination, especially to species, requires expertise, similar to that required for precise water chemistry and macroinvertebrate assays. Such fine level determinations may be useless if not conducted by experienced taxonomists. Some companies provide algal identification and analysis services that may be useful for those lacking such expertise. The reputation of prospective companies should be verified. In general, total algal biomass is of greater concern than taxonomic composition to those wishing to control eutrophication and its effects.

MACROINVERTEBRATE ANALYSIS

Macroinvertebrates may indicate water quality problems and some monitoring programs may want to evaluate biomass and diversity of macroinvertebrates. There is little precedence for this in stream eutrophication studies, and the analysis of macroinvertebrates to species is time consuming. Methods to assess stream macroinvertebrates have recently been reviewed (Hauer and Resh 1996). Generally, identification of most animals to species is required for accurate indices to be constructed, so it is important that such analyses be carried out by individuals with taxonomic expertise.

COMMUNITY METABOLISM ANALYSES

Productivity/respiration (P/R) ratios can be determined by the upstream-downstream method with dissolved oxygen data and estimates of atmospheric reaeration (Odum 1956; Marzolf et al. 1994) or light/dark, flow-through chambers (Hickey 1987; Dodds and Brock 1998). P/R ratios measured using chambers are generally higher than those measurements obtained from upstream-downstream methods. Even in streams with heavy algal growths, it is rare to find P/R ratios in excess of one (1) using upstream-downstream methodology. Both methods convert the diel changes in dissolved oxygen into actual rates of productivity. The diel range in dissolved oxygen indicates the magnitude of gross productivity and can be used to monitor ecological integrity in streams and rivers of similar velocity, depth, and turbulence.

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APPENDIX C. STATISTICAL TESTS AND MODELING TOOLS

STATISTICAL ANALYSES

In order to use parametric tests, (Student *t* test, ANOVA, MANOVA, etc.) assumptions about the population distribution must be made. When the data are not normally distributed, transformations of the data to obtain a normal distribution are commonly made (e.g., log transformation). Less powerful, non-parametric tests of significance must be used in cases where the data do not fit the assumption of a normal distribution (Atlas and Bartha 1993).

STUDENT *t* TEST

The validity of hypotheses is frequently tested using the Student *t* test. There is a family of distributions for the *t* statistic that vary as a function of degrees of freedom. The *t* distributions are symmetrical about a mean of 0, as are normal distributions, though the *t* distributions are more spread out than on the normal curve. As degrees of freedom increase, the *t* distribution more closely approximates the normal curve. There are published tables of critical values for *t* that allow one to compare a calculated *t* value from one's own data with a *t* value determined by the level of significance. This comparison allows one to decide whether or not to reject the null hypothesis (Atlas and Bartha 1993). An in-depth discussion of the use of the *t* statistic for analyzing environmental data can be found in Ott (1995). Most statistics texts include the published tables of the *t* statistic and information on it's application.

ANOVA

The analysis of variance (ANOVA) method is used to determine the significance or validity of data when information is collected from different populations. ANOVA is generally used to confirm that there are not significant differences in sample population means. ANOVA determines whether there is greater variability among sample populations or within population groups. ANOVA is performed by summing the variance of all sample points and comparing it to the sum of the variance of all the sample means (Remington and Schork 1985). ANOVA is useful for calculating the unbiased variance of samples that have been composited or parts of samples (such as a 10 mL water sample analyzed for TP taken from a 50 mL total sample) (Gilbert 1987).

CHI SQUARE TEST

A common non-parametric statistical test is the χ^2 (chi square) test. When attempting to analyze the apportionment of a characteristic within a population, the chi square test is valuable for determining the independence of categorical variables. The raw data for a chi square test should be on a scale for which data are placed into discrete groups (nominal scale) (Atlas and Bartha 1993).

MANN WHITNEY *U* TEST

The Mann Whitney *U* test is one of the most powerful non-parametric statistical tests. This test may be employed in place of the *t* test when data are on an ordinal scale. This test is used with two independent groups. The null hypothesis is that both samples are drawn from populations with the same distributions. The alternative hypothesis is that the parent populations from which the samples are taken have different medians. This test assumes that the distributions have the same form, but have different medians. This test ranks scores from lowest to highest while retaining the identity of the group from which they came (control or experimental group), to determine the distribution of the *U* statistic. *U* represents the number of times the n_1 value precedes the n_2 value. *U* is large if the n_1 population is located below the n_2 population.

LINEAR REGRESSION

In regression analysis a relationship of best fit is used to describe the data. The experimenter must decide the type of relationship that best describes the data. If the relationship is linear, a linear regression may be appropriate. When the data are not linear, they may be log transformed to fit the linear assumption. The slope of the regression line is called the regression coefficient. In constructing a regression line of best fit, it is necessary to define the slope of the line and the intercept of an axis. Regression analysis minimizes the variance, though a residual variance remains. The statistical significance of the regression coefficient using the student *t* test described above. The null hypothesis in such a test, is that there is no difference between the calculated regression coefficient and a true population regression coefficient 0. In other words, the population regression coefficient indicates that no prediction of *y* can be made from *x*, nor of *x* from *y* (Atlas and Bartha 1993).

MULTIPLE REGRESSION

Multiple regression is based on the same principle as linear regression (where y=mx+b), but involves more than one regression variable (i.e., multiple sets of *x* values). Multiple regression is often performed using matrices and least squares approximations (Myers 1990). Applications may include developing relationships between response variables for various indicators.

BAYESIAN ANALYSIS

Bayesian analysis is most useful when incorporating historical data or comparing probabilities of various competing hypotheses. It allows use of all available data from various studies and weighing of different outcomes. A discussion of the uses of Bayes Theorem in statistical analysis can be found in Hilborn and Mangel (1997).

MODELS

The models discussed in this appendix may be used in criteria derivation when data are not sufficient. However, only the WASP model predicts periphyton biomass. The other models described here use periphyton as a forcing function for predicting nutrients or DO.

BETTER ASSESSMENT SCIENCE INTEGRATING POINT AND NONPOINT SOURCES (BASINS)

Better Assessment Science Integrating Point and Nonpoint Sources, or BASINS, is a tool developed by EPA to facilitate water quality analysis on a watershed level and for specific waterbodies or stream segments. BASINS was designed to integrate national water quality data, modeling capabilities, and geographic information systems (GIS) so that regional, State, local and Tribal agencies can easily address the effects of both point and nonpoint source pollution and perform sophisticated environmental assessments.

BASINS is made up of five components: (1) national databases; (2) assessment tools (TARGET, ASSESS, and Data Mining) for evaluating water quality and point source loadings at a variety of scales; (3) utilities including local data import, land-use and DEM (Digital Elevation Model) reclassification, watershed delineation, and management of water quality observation data; (4) watershed and water quality models including NPSM (Nonpoint Source Model), HSPF (Hydrologic Simulation Program Fortan), TOXIROUTE, and QUAL2E; and (5) post processing output tools for interpreting model results.

The three analytical tools (TARGET, ASSESS, and Data Mining) within BASINS allow the user a range of environmental assessment options. TARGET examines large area watersheds on a State/Tribal or regional level to analyze point source loads or general water quality. ASSESS gives information about specific water bodies and the monitoring stations or discharge points near them. Data Mining integrates historical, geographic, and water quality data using maps and tables. In addition, models such as the NPSM, QUAL2E, and TOXIROUTE can be used to predict the fate, transport, and effects of loadings from various sources. The BASINS package can be used for many water quality management analyses, particularly the development of total maximum daily loads (TMDLs). In addition, the GIS component of BASINS allows the user to virtually traverse the watershed.

BASINS is a software package that is installed on the user's computer. It may be downloaded from the EPA website (http://www.epa.gov/ost/BASINS/download.htm) or ordered on CD-ROM from the National Service Center for Environmental Publications (NSCEP). A printed copy of BASINS version 2.0 Users' Manual is also available through NSCEP. BASINS training courses are available in some areas of the country. For more information on BASINS, see the BASINS website (http://www.epa.gov/ost/BASINS/).

HYDROLOGICAL SIMULATION PROGRAM - FORTRAN (HSPF)

HSPF is a comprehensive package developed by EPA for simulating water quantity and quality for a wide range of organic and inorganic pollutants from agricultural watersheds (Bicknell et al. 1993). The model uses continuous simulations of water balance and pollutant generation, transformation, and transport. Time series of the runoff flow rate, sediment yield, and user-specified pollutant concentrations can be generated at any point in the watershed. The model also includes instream quality components for nutrient fate and transport, biochemical oxygen demand (BOD), dissolved oxygen (DO), pH, phytoplankton, zooplankton, and benthic algae. Statistical features are incorporated into the model to allow for frequency-duration analysis of specific output parameters. Data requirements for HSPF are extensive, and calibration and verification are recommended. The program is maintained on IBM microcomputers and DEC/VAX systems. Because of its comprehensive nature, the HSPF model requires highly trained personnel. It is recommended that its application to real case studies be carried out as a team effort. The model has been extensively used for both screening-level and detailed analyses. Moore et al. (1992) describe an application to model BMP effects on a Tennessee watershed. Scheckenberger and Kennedy (1994) discuss how HSPF can be used in subwatershed planning. The HSPF model can be downloaded at EPA's BASINS website given above.

QUAL2E

The Enhanced Stream Water Quality Model (QUAL2E), originally developed in the early 1970s, is a one-dimensional water quality model that assumes steady-state flow but allows simulation of diurnal variations in temperature or algal photosynthesis and respiration (Brown and Barnwell 1987). QUAL2E represents the stream as a system of reaches of variable length, each of which is subdivided into computational elements that have the same length in all reaches. The basic equation used in QUAL2E is the one-dimensional advection-dispersion mass transport equation. An advantage of QUAL2E is that it includes components that allow quick implementation of uncertainty analysis using sensitivity analysis, first-order error analysis, or Monte Carlo simulation. The model has been widely used for stream waste load allocations and discharge permit determinations in the United States and other countries. EPA's Office of Science and Technology recently developed a Microsoft Windows-based interface for QUAL2E that facilitates data input and output evaluation, and QUAL2E is one of the models included in EPA’s BASINS tool. More information on QUAL2E, including downloadable program files, can be found at EPA’s website (www.epa.gov/docs/QUAL2E_WINDOWS/index.html).

CE-QUAL-RIV1

The one-dimensional Hydrodynamic and Water Quality Model for Streams (CE-QUAL-RIV1) was developed through the Waterways Experiment Station of the Corps of Engineers. The model was designed to simulate water quality conditions associated with the highly unsteady flows that can occur in regulated rivers (e.g., storm water flows and streams below peaking hydropower dams). The model has two submodels for hydrodynamics (RIV1H) and water quality (RIV1Q). Output from the hydrodynamic solution is used to drive the water quality model. Water quality constituents modeled include temperature, dissolved oxygen, carbonaceous biochemical oxygen demand, organic nitrogen, ammonia nitrogen, nitrate nitrogen, and soluble reactive phosphorus. The effects of algae and macrophytes on water quality can also be included as external forcing functions specified by the user. A limitation of CE-QUAL-RIV1 is that it is only applicable to situations where flow is predominantly one-dimensional. Currently, this model can only be downloaded for USCOE use. More information on CE-QUAL-RIV1 can be found at the WES website (www.wes.army.mil/el/elmodels/riveinfo.html).

CE-QUAL-W2

CE-QUAL-W2 is a two-dimensional, longitudinal/vertical water quality model that can be applied to most waterbody types. It includes both a hydrodynamic component (dealing with circulation, transport, and deposition) and a water quality component. The hydrodynamic and water quality routines are directly coupled, although the water quality routines can be updated less frequently than the hydrodynamic time step to reduce the computational burden in complex systems. Water quality constituents that can be modeled include algae, dissolved oxygen, ammonia-nitrogen, nitrate-nitrogen, phosphorus, total inorganic carbon, and pH. http://www.wes.army.mil/el/elmodels/w2info.html

WASP5

The Water Quality Analysis Simulation Program is a general-purpose modeling system for assessing the fate and transport of conventional and toxic pollutants in surface waterbodies. Its EUTRO5 submodel is designed to address eutrophication processes and has been used in a wide range of regulatory and water quality management applications. The model may be applied to most waterbodies in one, two, or three dimensions and can be used to predict time-varying concentrations of water quality constituents. The model reports a set of parameters, including dissolved oxygen concentration, carbonaceous biochemical oxygen demand (BOD), ultimate BOD, phytoplankton, carbon, chlorophyll *a*, TN, total inorganic nitrogen, ammonia, nitrate, organic nitrogen, total inorganic nitrogen, organic phosphorus, and inorganic phosphorus. Although zooplankton dynamics are not simulated in EUTRO5, their effect can be described by user-specified forcing functions. Lung and Larson (1995) used EUTRO5 to evaluate phosphorus loading reduction scenarios for the Upper Mississippi River and Lake Pepin, while Cockrum and Warwick (1994) used WASP to characterize the impact of agricultural activities on instream water quality in a periphyton-dominated stream. http://www.epa.gov/earth100/records/wasp.html

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).