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.

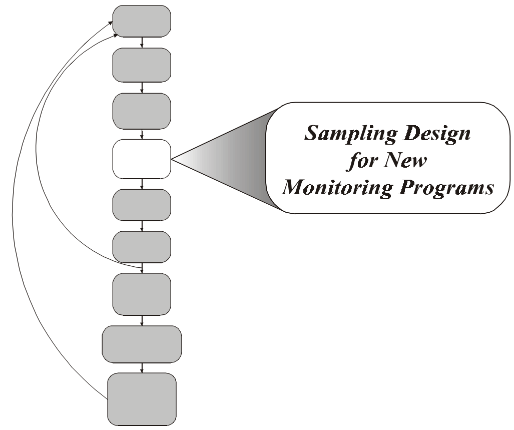
Current velocity, sediment type, and light availability to a large extent determine the plant types that occur in rivers (Hynes 1969; Goldman and Horne 1983; Chambers et al. 1999).

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.

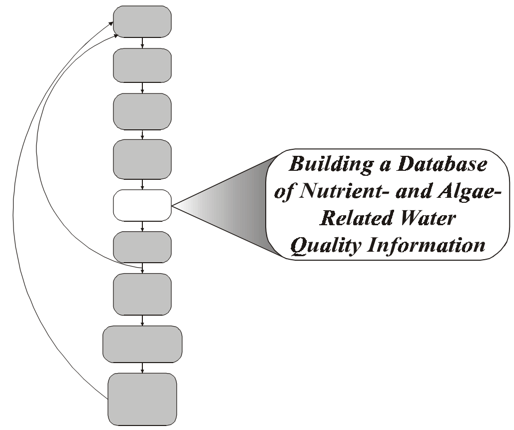
INVOLVEMENT OF CITIZEN MONITORING PROGRAMS

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