

BRIDGES

BRIDGES is a recurring feature of J-NABS intended to provide a forum for the interchange of ideas and information between basic and applied researchers in benthic science. Articles in this series will focus on topical research areas and linkages between basic and applied aspects of research, monitoring policy, and education. Readers with ideas for topics should contact Associate Editors, Nick Aumen and Marty Gurtz.

Criteria for setting nutrient levels in lotic ecosystems are relevant to US states and other countries in the process of setting water-quality regulations. There are few articles in the peer-reviewed literature on this topic, and policy makers have had little information from which to base their decisions for streams. This lack of information is particularly troublesome because of the large number of streams and rivers that have impaired water quality, and the ever-increasing pace of urban and agricultural development. In addition to the effects of high nutrient concentrations on stream ecosystem structure and function, high nutrient concentrations, particularly nitrate, may have adverse effects on human health.

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Establishing nutrient criteria in streams

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“Every child deserves to grow up with water that is pure to drink, lakes that are safe for swimming, rivers that are teeming with fish. We have to act now to combat these pollution challenges with new protections to give our children the gift of clean, safe water in the 21st century.”

—President Clinton, 23 February 1999, Baltimore

The US Environmental Protection Agency (USEPA) and the US Department of Agriculture (USDA) recently have been directed to set criteria for nutrients in rivers, lakes, and estuaries. State reports compiled by the USEPA (National Water Quality Inventory: 1996 Report to Congress) claim that 40% of streams or rivers surveyed were impaired because of the nutrients N and P, but no well-defined standards have been proposed to determine if nutrients impair flowing waters (USEPA 1998). A rational framework for determining criteria is necessary because the USEPA has been charged with establishing maximum acceptable levels of nutrients in streams and rivers by 2001 as part of the Clean Water Action Plan. State and tribal governments

will use these criteria to set total maximum daily loads (TMDLs) for nutrients and adopt their own standards by 2003. The USEPA, the USDA, and other national governmental agencies (e.g., US Geological Survey [USGS], US Army Corps of Engineers), state and tribal officials, and private parties will set these criteria.

Data analyses are needed to explain the relationships between stream algae and nutrients, which previously have received attention from researchers. Given the potential economic impacts of nutrient control, the process by which nutrient levels are set likely will engender controversy. Basing the criteria on the best scientific data available will minimize conflict and maximize the potential benefits related to controlling nutrients in streams.

Although the question of how to set nutrient criteria is framed above in terms of US politics and policies, other countries also are interested in nutrient criteria for streams, particularly developed countries where industrialization, urbanization, and modern agriculture have resulted in extensive nutrient discharge into water courses. The following discussion will be based

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primarily on examples from the US, but the general principles could apply to any watershed.

We 1st will address why nutrient criteria are needed. Next, we will discuss the scientific basis behind possible criteria. Last, we discuss ways to account for variability in streams (i.e., factors that may decouple nutrient concentrations from biomass). The main thrust of this paper is to outline what scientific methods currently are available to managers for setting nutrient criteria, given specific reasons for setting the criteria.

Why do we need nutrient criteria?

Reasons for nutrient criteria include: 1) adverse effects on humans and domestic animals, 2) aesthetic impairment, 3) interference with human use, 4) negative impacts on aquatic life, and 5) excessive nutrient input into downstream systems. Each of these will be discussed in sequence; they share several characteristics, but they also have facets that may result in criteria being set at different levels.

High levels of some nutrients may have adverse effects on human health. Control of NO_3^- levels particularly is important to avoid methemoglobinemia. Furthermore, NO_3^- consumption has been correlated with stomach cancer (Hartman 1983). Although correlation does not guarantee causation, NO_3^- could be regulated as a carcinogen in the future.

Eutrophication from N and P causes proliferation of algal masses, some of which may be toxic. In one of the worst cases, eutrophication caused Cyanobacteria to bloom in the stagnant Murray-Darling River system (Australia) during a drought, leading to livestock deaths and concerns about impacts on humans (Bowling and Baker 1996). Such toxic blooms are most likely to occur in very enriched, slow-moving, and nonturbid rivers.

Eutrophication causes taste and odor problems in lakes (Arruda and Fromm 1989, Wnorowski 1992), but these negative effects have not been linked directly to trophic state of streams and rivers. Such problems often can be traced to production of odorous metabolites by Cyanobacteria (e.g., geosmin) and other algae and their subsequent leakage into surrounding waters. Algae that cause taste and odor problems can reach high biomass in eutrophic streams and rivers, both in the phytoplankton of slow-

moving rivers and the periphyton of shallow streams.

Aesthetic impairment is more difficult to quantify, but usually is associated with filamentous algal forms. Nuisance levels may be reached somewhere between 100 and 200 mg/m² chlorophyll (Horner et al. 1983, Nordin 1985, Welch et al. 1988, Quinn 1991). Enriched waters often have benthic chlorophyll concentrations >150 mg/m², and many stream users find high levels of algal growth objectionable (Welch et al. 1989, V. Watson, University of Montana, personal communication). A link also may exist between property values and trophic state in lotic waters, as has been documented for lakes (Michael et al. 1996). However, to our knowledge, such analyses have not been conducted for rivers and streams.

Excessive growth of algae and macrophytes can interfere with human uses of flowing waters. Such interference is exemplified by problems caused by the filamentous green alga *Cladophora*. Exorbitant amounts of this alga can slow water flow in canals (decreasing delivery rates and increasing water losses), interfere with swimming, and snag fishing lures (Dodds and Gudder 1992). Furthermore, excessive algal growth may clog screens on water intakes for water treatment plants and industries.

High concentrations of NH_3 in the water column clearly are toxic to aquatic animals (Russo 1985). For example, levels of >1 mg/L $\text{NH}_3\text{-N}$ in Ohio streams have negative impacts on the fish communities (Miltner and Rankin 1998). Negative impacts on aquatic life related to stimulation of algal biomass by increased nutrients are subtler. As a system becomes more productive, different species of algae may become more competitive and species composition can shift (Kelly and Whitton 1995, Pan et al. 1996, Kelly 1998). However, unless such species shifts cause specific water-quality symptoms (e.g., toxic algae) or aesthetic problems (e.g., very long streamers of filamentous algae), the public is unlikely to be concerned.

Nutrient enrichment may adversely affect stream animal communities. Enriched streams have increased invertebrate biomass and altered invertebrate communities (Bourassa and Cattaneo 1998). Community structure has been correlated directly with P concentration (Miltner and Rankin 1998). Excessive levels of algae were damaging to invertebrates (Nordin 1985).

Changes in community structure may be viewed as problematic, particularly if game fish are affected. In extreme cases, levels of primary production can be stimulated by nutrients; organic C will build up in the system and cause a subsequent low dissolved O₂ (DO) and high pH event. Fish and invertebrates will grow poorly and even die if the O₂ depletion and pH increases are severe (Welch 1992).

Because streams drain into lakes and oceans, eutrophication caused by influx of nutrients from flowing waters is a concern for downstream lake and coastal areas. Examples of marine eutrophication are the zone of hypoxia that develops in the Gulf of Mexico (Rabalais et al. 1998) and the production of toxic estuarine dinoflagellate blooms (Burkholder and Glasgow 1997). Eutrophication problems in lakes are well documented, and the control of external and internal nutrient loading necessary to minimize eutrophication can be calculated (Cooke et al. 1993). Requirements for control of nutrient loading to lakes and coastal marine systems may lead to more stringent nutrient criteria in rivers than those required for controlling instream eutrophication, especially in localities where stream algae are limited by factors other than nutrients.

What are the scientific bases for criteria?

In this section we discuss what nutrients and what forms of those nutrients should be used to set criteria. We also describe some models and approaches that can be used to set criteria. Last, we offer some discussion on how criteria may vary depending upon the reason for the criteria.

What nutrients and forms should be used to set criteria?

The traditional view is that P limits primary production in fresh waters (e.g., Correll 1998), and N limits it in the ocean. However, nutrient bioassays and correlation analyses do not substantiate this point of view. Data were compiled from 158 bioassays reported in the literature in which the response of stream periphyton to nutrient fertilization was measured. Of the studies, 13% showed stimulation by N alone, 18% by P alone, 44% by simultaneous N and P additions, and 25% by neither nutrient (W. K. Dodds, unpublished data). The absolute proportions as-

sociated with each type of limitation should not be viewed as a general guide to nutrient limitation in streams. However, bioassay results do suggest that both N and P can limit primary producers in streams.

Correlation analyses also do not support the idea of P as the sole limiting nutrient in rivers and streams. Mean and maximum benthic chlorophyll correlated better with total N (TN) than total P (TP) in the water column in several hundred streams. Nitrogen and P occur in several forms in rivers and streams, including dissolved organic and inorganic forms and in particulate material. All of these forms together are referred to as TN and TP. Total N does not include dissolved N₂ gas. A regression model using both nutrients explained the highest proportion of the variance in biomass (Dodds et al. 1997). Thus, both N and P can control primary production in at least some streams and rivers.

Control of P alone may cause P to limit and lower algal biomass, as has occurred in many lakes (Sas 1989, Cooke et al. 1993). However, if pulses of P occur, they can be taken up in excess of requirements and stored inside algal cells in a process called *luxury consumption*. This stored P can allow algae to grow even if P concentrations are low in the water column. If controlling such P pulses is impossible (e.g., pulses associated with high runoff events in spring), control of N could become necessary. For example, Lohman and Prisco (1992) demonstrated that intracellular P concentrations in *Cladophora* increased, while P availability in the surrounding water decreased. Thus, *Cladophora* was N limited, even though analyses of available N and P in the river water column suggested P limitation. Given the bioassay and correlation data, and that periphyton can consume P in excess of immediate needs, setting nutrient criteria for both N and P makes sense.

Unless clear limitation by other nutrients has been demonstrated in a particular system, N and P should be assumed to be the dominant nutrients controlling the trophic states of streams and rivers. Fortunately, nuisance and some toxic heterocystous Cyanobacteria that can use N₂ gas as a N source generally are not part of eutrophic stream periphyton, but may occur in the plankton of slowly flowing rivers (Bowling and Baker 1996). The decreased dominance of heterocystous Cyanobacteria in streams leads to some situations where N control alone may

lead to decreases in algal biomass. Controlling N and not P inputs in lakes can encourage blooms of nuisance Cyanobacteria (Stockner and Shortreed 1988). However, the strategy of controlling N alone should be viewed with caution, especially in plankton-dominated rivers. More data on cyanobacterial problems in eutrophic streams are necessary before we can be certain that N control will not lead to cyanobacterial dominance.

Control based on measured levels of dissolved inorganic N and P may not be effective because these pools are replenished rapidly by remineralization in surface waters (Dodds 1993). Correlation of algal biomass with dissolved inorganic nutrients was poor in some studies (Dodds et al. 1997), but not all (Biggs and Close 1989). Also, lake managers are aware of problems with using dissolved inorganic nutrient concentrations to set nutrient criteria. Last, most of the data linking land-use practices to N and P loading have been reported in TN and TP (Loehr 1974), so basing criteria on total nutrients for calculating TMDLs is more practical than using dissolved inorganic nutrients.

Two caveats are necessary to the generalization that TN and TP should be emphasized. First, if nutrients are released directly into streams in dissolved inorganic form, their influence may be more intense and localized near the point source of release. Dissolved inorganic nutrients will be taken up rapidly, which can lead to a very high, localized concentration of biomass (Hynes 1969). Second, some models using seasonal means of dissolved inorganic nutrients to predict algal biomass have been very successful (Biggs 1995, 2000), and some sites have considerably more data on dissolved than total nutrients on which to base decisions.

At what concentrations should criteria be set?

One difficulty in setting criteria involves assessment of the trophic state of a stream or river. Stated another way, how can we declare that a river or stream is in an unacceptable trophic state if there is no basis for scaling the trophic state relative to other rivers? A generally accepted system for classifying the trophic states of streams and rivers is lacking (Dodds et al. 1998). In general, trophic state is classified by nutrients and algal biomass. System metabolism may be more relevant to ecosystem function, but

difficulties with methods and limited data have precluded use of production and respiration to classify trophic states of lakes and streams. One classification system proposed for streams relies upon the cumulative frequency distributions of chlorophyll and nutrients. The lower 1/3 of the distribution sets the range for oligotrophic streams, and the upper 1/3 for eutrophic streams. This approach is consistent with the convention of classifying trophic state into 3 categories, while basing classification on the actual distribution of biomass and nutrient levels found in streams (Dodds et al. 1998). The published classification was based on only 286 temperate streams. More data are necessary to determine how well this classification scheme applies to rivers from different ecoregions, how distributions of nutrients correlate to algal biomass, and how well such classification represents pristine conditions. Analyses of existing databases may provide a valuable tool in extending this approach to trophic classification.

A few models directly link TN and TP to benthic algal biomass in streams (e.g., Lohman et al. 1992, Dodds et al. 1997, Bourassa and Cattaneo 1998, Chételat et al. 1999). Such models can be applied to estimate algal biomass as a function of water column nutrients. A similar correlation approach has been very successful in managing eutrophication in lakes and reservoirs. Extension of these models to link in-stream nutrient concentrations to known sources of nutrient loading also has been described (Dodds et al. 1997).

Models describing the correlation between nutrients and chlorophyll in lakes differ from those for streams because benthic chlorophyll may be much more variable in streams as a result of the effects of floods, turbidity, and grazing. This difference is exemplified by the ratios of maximum to mean chlorophyll. This ratio describes the variance in level of chlorophyll, with high numbers denoting a high variance. The maximum/mean chlorophyll ratio is 4.5 for stream benthos compared to 1.7–2.6 for lake phytoplankton (Dodds et al. 1998). Furthermore, total water column nutrients usually are correlated strongly with chlorophyll because phytoplankton contain chlorophyll, N, and P. This linkage leads to high correlation coefficients between total nutrients and algal biomass in lakes. This relationship is not as highly coupled in streams when *benthic* chlorophyll and *water col-*

umn TN and TP are considered. Thus, the correlation models developed for stream benthic algae contain a much greater degree of uncertainty than those for lakes.

Biggs (2000) proposes a correlation method that considers hydrodynamic disturbance and inorganic nutrients in New Zealand streams, that is pertinent for predicting benthic algal biomass. Such an approach may prove useful within an ecoregion, and could be used to provide a sliding scale of nutrient criteria, with higher nutrient content allowed in more hydrodynamically unstable rivers (i.e., criteria may be more lenient because of regular scouring of algal biomass in rivers that flood frequently).

An alternative approach to correlation models also has been developed. This method consists of sampling nutrients in reference stream reaches where chlorophyll levels are deemed acceptable. Gary Ingman (Montana Department of Environmental Quality) and Vicki Watson (University of Montana) proposed this technique for use in the Clark Fork River in Montana (Dodds et al. 1997). General regional criteria have yet to be established using this method. In systems where the entire stream receives nutrient loading, or regions where all watersheds are enriched, locating suitable reference reaches may be impossible. Data from other similar streams should be used to identify the obtainable baseline nutrient concentrations in those cases.

A regression model linking TP to river phytoplankton is available (Van Nieuwenhuysse and Jones 1996). This model can be used to set TP criteria. The TP levels can be used to calculate corresponding TN concentrations with the Redfield ratio (Harris 1986). This model captures additional variance when watershed area is considered.

Setting nutrient criteria is difficult based on subjective impressions of what constitutes excessive levels of benthic algae. However, >200 mg/L of benthic chlorophyll generally produces a very green stream bottom (Welch et al. 1988). To further complicate matters, filamentous green algae have a less desirable appearance than brown-colored diatoms, even when the biomass of the 2 is similar. Moreover, a large amount of the variance in benthic chlorophyll levels in streams is not related to nutrient levels. We simply do not have the data in the US to predict when benthic algal community structure will shift to more nuisance forms with changes

in nutrients. Preliminary data from Canada indicate that rhodophytes make up a large portion of the algal community when biomass is low, and *Cladophora* and *Melosira* prefer high nutrient water (Chérelat et al. 1999). More research clearly is needed in this area, both original research and analysis of existing data. Thus, criteria based on current data will need to be set based on what amount of chlorophyll is acceptable, not on how nutrient amounts and ratios will influence algal communities.

Dissolved O₂ deficit and high pH are perhaps the most severe algal-related problems affecting the aquatic life-support characteristics of a river or stream. Deficits of DO can occur when respiration of organic C produced by photosynthetic processes in the stream exceeds the ability of reaeration to supply DO. Depletion of DO in streams was described years ago (Odum 1956). However, the severity of the deficit is difficult to predict in specific situations. Deficits of DO are most likely to occur in rivers with laminar flow (slow, non-turbulent flow), when a large algal biomass is present, with high water temperature, and during times of low light (early morning or after protracted cloudy periods). Given that such events rarely are recorded (though they may occur frequently), and that so many factors are related to DO depletion rates, existing data for most streams are insufficient to develop nutrient criteria for avoiding DO deficits. Such models probably will be developed in the future and development will be facilitated by recent improvements in tools for measuring and storing temporal data on instream DO concentrations. As more data become available, it will be possible to directly link frequency and severity of low DO events with nutrient loading.

Similar problems exist for predicting pH excursions. High pH is promoted by laminar flow and sunny conditions that, respectively, minimize atmosphere-to-water transport of CO₂ and maximize photosynthetic uptake of CO₂. Again, limited data for most streams hamper prediction of the degree of pH excursions as a function of TMDLs of N and P.

Nutrient criteria also could be set relative to other streams on a regional or national basis. Dodds et al. (1998) combined data from the EPA eutrophication survey (Omernik 1977) and several hundred streams and rivers in the US and analyzed the resulting cumulative frequency distributions. Half of the systems had TP >0.04

mg/L, and $\frac{1}{2}$ had TN >0.9 mg/L. If the target is to bring streams and rivers to nutrient levels at or below current means, then using frequency distributions would be a viable approach to setting nutrient criteria. Problems with using such frequency distributions are discussed below.

Nutrient criteria may be more stringent when potential eutrophication of systems fed by rivers is a factor driving adoption of criteria. A common classification system suggests that $35 \mu\text{g/L}$ TP and a mean of $8 \mu\text{g/L}$ chlorophyll constitutes the dividing line between eutrophic and mesotrophic lakes (Organization for Economic Cooperation and Development [OECD] as cited in Rast et al. 1989). In contrast, data from Dodds et al. (1997) suggest that maximum benthic chlorophyll values are likely to exceed 200 mg/m^2 at $90 \mu\text{g/L}$ TP, and mean values of chlorophyll of 50 mg/m^2 are likely with $55 \mu\text{g/L}$ TP. Thus, unacceptable levels of chlorophyll may occur at much lower nutrient concentrations in lakes than streams.

Streams and rivers are less likely to accumulate as much algal biomass as lakes, given the same TP, because the lentic planktonic habitat is considerably more benign. Thus, there is fairly low chlorophyll yield per unit nutrient in streams. Comparing streams that flood at moderate frequency to more hydrodynamically stable artificial and spring-fed streams substantiates this view. Much higher benthic chlorophyll yield per unit TP than predicted by Dodds et al. (1997) is possible in controlled laboratory streams, outdoor artificial streams, or spring-fed rivers (Welch et al. 1992, Walton et al. 1995, Anderson et al. 1999, Welch et al., in press).

Likewise, planktonic chlorophyll yield is less in flowing waters than in lakes. A river with $8 \mu\text{g/L}$ chlorophyll would have $\sim 48 \mu\text{g/L}$ TP, using the relationship proposed by Van Nieuwenhuysse and Jones (1996) for suspended chlorophyll in rivers as a function of TP. This value is ~ 1.4 times greater than the proposed mesotrophic/eutrophic boundary value for lakes and reservoirs (OECD as cited in Rast et al. 1989).

Last, a missing link in the above discussion is how to relate instream TN and TP concentrations to nonpoint and point sources of nutrients (i.e., to set TMDLs). Models predicting nutrient loading in streams need to be developed if mitigation strategies based on water column nutrients are to be successful. 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 in various regions may yield a 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.

What factors may alter responses to nutrient control?

Variation of benthic algal biomass occurs among areas with different geology, land-use practices, and as a function of other biotic and abiotic factors. In this section, we discuss how regional differences (ecoregions) may play a role in setting nutrient criteria. In general, the relationships described above that can be used to set criteria based on algal biomass response, represent average responses.

Nutrient criteria should be set after considering the natural state of streams and rivers in an ecoregion. For example, in watersheds with high- PO_4^{3-} rock that is weathering at significant rates, low P concentrations may never occur. Large rivers will have higher TP, and yield of suspended algae will be different than in smaller streams (Van Nieuwenhuysse and Jones 1996). Furthermore, some watersheds have very high natural NO_3^- weathering rates (Halloway et al. 1998). Such areas naturally high in nutrients occur in several places in the US (Omernik 1977). Clearly, if nutrient levels naturally are high in a watershed, restrictive nutrient criteria cannot be met. Furthermore, when pristine systems are absent, determining natural baselines could be impossible.

Considerably greater levels of accuracy for prediction of benthic algal biomass with regression models are possible if region-specific data are available. For example, the general data sets used in regression models relating water column nutrients to benthic algae developed by Dodds et al. (1997) have a maximum r^2 of 0.43. Data from Missouri streams alone have r^2 values ranging from 0.47–0.60, depending upon year and whether TN or TP is used to predict algal biomass (Lohman et al. 1992). Biggs (1995) was able to construct a model for algal biomass with an r^2 of 0.89 in a region of New Zealand by normalizing for the effect of floods and using conductivity as a surrogate for nutrients. Fur-

TABLE 1. Various potential nutrient criteria set using different outcomes of concern related to instream nutrient concentrations. TN = total N, DIN = dissolved inorganic N.

Outcome	N (mg/L)	Total P (mg/L)	Comments
Toxicity, human	10 NO ₃		US national standard
Toxicity, aquatic life, acute	0.03–5 NH ₃		Fish and invertebrate data (Russo 1985)
Toxicity, aquatic life, chronic	0.005–1 NH ₃		Fish data (Russo 1985, Miltner and Rankin 1998)
Oxygen deficit, pH excursion	?	?	Probably greater than levels presented below
Mean benthic chlorophyll <50 mg/m ²	0.47 TN	0.055	Large data set (Dodds et al. 1997)
Mean benthic chlorophyll <50 mg/m ²	0.25 TN	0.021	Lohman et al. (1992)
Maximum benthic chlorophyll <200 mg/m ²	3.0 TN	0.415	Calculated from Dodds et al. (1997)
Significant effect on biotic integrity index using invertebrates and fish	1.37 inorganic N	0.17	Headwater streams, Ohio (Miltner and Rankin 1998); effects less apparent in larger rivers
Systems with nutrient concentrations in upper ½	0.9 TN	0.04	Dodds et al. (1998)
Planktonic stream chlorophyll <8 µg/L	0.29 TN	0.042	Calculated from Van Nieuwenhuysse and Jones (1996); chlorophyll level from Organization for Economic Cooperation and Development (OECD, as cited in Rast et al. 1989); TN set by Redfield ratio (Harris 1986)
Lake mesotrophic/eutrophic boundary (planktonic chlorophyll <8 µg/L)	0.25 TN	0.035	OECD (as cited in Rast et al. 1989); TN set by Redfield ratio
Values set by State of Montana and co-operators	0.30 TN	0.020	Tri-State Implementation Council, Clark Fork Voluntary Nutrient Reduction Program
Levels leading to periphyton and macrophyte control	1.0 DIN	<0.020 (total dissolved)	Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication)
Levels set to control summer phytoplankton		0.07	Tualatin River, Oregon (R. Burkhart, Oregon Department of Environmental Quality, personal communication).
Levels recommended to control maximum periphyton below 200 mg/m ² for 50 d accrual	0.019 DIN	0.002 (soluble reactive)	(Biggs 2000)

thermore, all relationships that have been developed to date are from temperate regions, with most data from North America and New Zealand. Subtropical or polar regions could have quite different relationships. Thus, if data are available for an ecoregion, they should be used to set criteria for that region. Extant data such as state and tribal water quality records, USGS National Water Quality Assessment Program data, and Environmental Monitoring and Assessment Program results may serve as sources for such analyses.

Streams in a local region also may exhibit different relationships between TN or TP and benthic chlorophyll than those observed with larger-scale data sets. Thus, large, generalized data sets should not be the 1st choice for setting criteria, if local data are available. For example, the TN and TP values that yield a mean benthic chlorophyll of 50 mg/m², were lower for the detailed data set from Missouri than those from a larger data set (Table 1).

Nevertheless, one should not expect that the nutrient concentration yielding a given peri-

phytic biomass will be markedly different among regions if other factors (i.e., light, grazing, etc.) are similar. For example, most regression relationships for chlorophyll-TP in lakes show slopes or chlorophyll:TP ranging from 0.5–1.0 (Ahlgren et al. 1988). Invertebrate grazing may result in low chlorophyll yield per unit nutrient in streams regardless of ecoregion (Bourassa and Cattaneo 1998), as is the case in lakes.

One potential problem with the ecoregion approach is that variation over time and space within a small area may be as great as the variation among ecoregions. The nutrient bioassays of Wold and Hershey (1999) demonstrate high variation of responses to N or P additions in 6 watersheds within 100 km of each other. The responses also were variable across season. Similar seasonal responses have been documented in New Zealand streams (Francoeur et al. 1999).

All the data sets that have been published linking algal biomass to water column nutrients in rivers and streams have a potential statistical problem (Lohman et al. 1992, Biggs 1995, Van Nieuwenhuysse and Jones 1996, Dodds et al. 1997, Chételat et al. 1999). Investigators may have introduced bias in site selection because sites were not selected randomly. In many cases, study sites are selected specifically to represent the broadest possible range of site types. Thus, extremely eutrophic and oligotrophic systems may be overrepresented. Such models may work well for the streams used to construct the models, but their application should be viewed with caution. For example, Dodds et al. (1997) reported relationships among nutrients and chlorophyll derived from literature values. The investigators who conducted this literature analysis had no way of knowing why investigators choose to investigate particular sites or if all data were reported. Streams with low amounts of periphyton may have been excluded, or researchers may have preferred to work in pristine systems. Last, much ecological investigation has concentrated on temperate, forested streams, which may have low levels of nutrients and where canopy cover may have restricted algal growth. Temperate forested streams may not be globally representative of all streams because they provide <1/3 of the runoff from the earth's continents (Dodds 1997). Thus, future sampling strategies to generate data that will be used to link stream eutrophication with nutrients should attempt to

avoid investigator-specific biases. The models for setting criteria should be based on representative streams with data taken from the full population of streams and with each type of stream sampled in proportion to its relative occurrence. Such an approach has been taken in lakes (Peterson et al. 1999). Large data sets such as those collected by the USGS water quality monitoring network of the National Water Quality Assessment Program may be useful because sites could be selected from the databases to provide data specific to individual ecoregions.

If streams and rivers are turbid as a result of suspended particles, nutrient enrichment will have less influence on trophic status of the entire system. Sediments attenuate light, which becomes the factor limiting ecosystem production. However, even in turbid systems, enrichment may increase periphyton and macrophyte production in shallow portions of the river. Similarly, extensive shading by a riparian canopy will inhibit algal growth. Both conditions reduce chlorophyll yield per unit nutrient.

If macrophyte production predominates in streams and rivers, setting nutrient criteria will be difficult. We are not aware of any general published relationships between water column nutrients and macrophyte biomass. Such relationships may be very difficult to establish for macrophytes that are able to acquire nutrients from sediments through their root systems. However, nutrient control resulted in lowered macrophyte biomass in the Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication), so future work on macrophyte-nutrient relationships could yield useful predictive models.

Conclusions

Many factors can regulate primary producers in streams, including nutrient availability, hydrodynamics, grazing, turbidity, riparian shading, and human impacts (e.g., addition of toxic compounds, global change, introduced species, watershed development). However, nutrient inputs are usually the most effectively managed factor. Factors in addition to nutrients need to be considered mainly because they can lead to cases of low algal biomass with high nutrients. Although these additional factors may decouple nutrient enrichment from algal biomass, most of these (e.g., flooding, grazing, turbidity) are not

easily controlled at most sites. Thus, we are left with setting nutrient criteria as the primary way to mitigate problems of excessive algae.

Developing a single value that can be used for nutrient criteria in streams and rivers will be difficult, given the variety of reasons for setting the criteria (Table 1). To protect human health, no more than 10 mg/L NO_3^- -N should be present. To avoid chronic toxicity by NH_3 , no more than 0.02 mg/L NH_3 -N should be present. If the concern is eutrophication, then setting criteria for TN and TP is most reasonable.

If streams are not turbid, preventing maximum benthic chlorophyll levels from exceeding 200 mg/m² is reasonable because streams with higher levels are not aesthetically pleasing, and their recreational uses may be compromised. For benthic chlorophyll to remain below 200 mg/m² at the very least, TN should remain below 3 mg/L, and TP below 0.4 mg/L. Based on cumulative frequency distributions of nutrients, and assuming that $\sim\frac{1}{2}$ the systems in the US have been impaired by excessive nutrients, levels of TN and TP would be set at 0.9 and 0.4 mg/L, respectively. If a mean of 50 mg/m² chlorophyll is the target (thus ensuring chlorophyll is <100 mg/m² most of the time), TN should be 0.47 and TP 0.06 mg/L. Lower levels for nutrient criteria should be considered for regions with more pristine systems (e.g., TN and TP levels of 0.3 and 0.02 mg/L, respectively, were chosen for the Clark Fork River in Montana, Table 1). If systems downstream are to be protected, even lower stream nutrient concentrations will be necessary in some situations.

A significant amount of monitoring data are necessary to refine recommendations for nutrient criteria. Some regions and agencies have data that can be used for this purpose. Data that would be useful to collect or glean from existing sources for many more systems include seasonal means and maxima for benthic and planktonic chlorophyll, associated water column nutrients, and diurnal DO concentrations for a variety of stream types. Such data should be collected in a way that avoids sampling bias. Data on macrophyte abundance related to nutrients, reference streams with acceptable algal and macrophyte biomass, and factors related to dominance by nuisance algal and macrophyte species also are sorely lacking for many regions.

Establishing rational criteria will require bridging the gap between managers and scien-

tists. The managers will provide the realistic assessment of what needs to be accomplished, whereas the scientists can suggest the best available means to reach the management goals. Continued interplay between applied and basic approaches will be necessary if eutrophication in streams is to be controlled in an efficient manner.

Acknowledgements

W. D. thanks Kirk Lohman for early conversations and subsequent collaborations that led to much of this work. Jack Jones, Vicki Watson, Val Smith, and Jan Stevenson also provided useful input into developing these ideas. Barry Biggs, Debra Hart, Marty Gurtz, Amanda Parker, and 2 anonymous reviewers commented on the manuscript and Dolly Gudder provided technical assistance. This research was supported in part by the Konza LTER grant from the National Science Foundation. This paper is contribution no. 99-375-J from the Kansas Agricultural Experiment Station.

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