APPENDIX A. NUTRIENT CRITERIA CASE STUDIES

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

TENNESSEE ECOREGIONAL NUTRIENT CRITERIA

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

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

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

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

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

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

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

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

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

PHASE I: DELINEATE SUB-ECOREGION BOUNDARIES

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

PHASE II: REFERENCE STREAM SELECTION

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

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

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

PHASE III: INTENSIVE MONITORING OF REFERENCE STREAMS

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

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

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

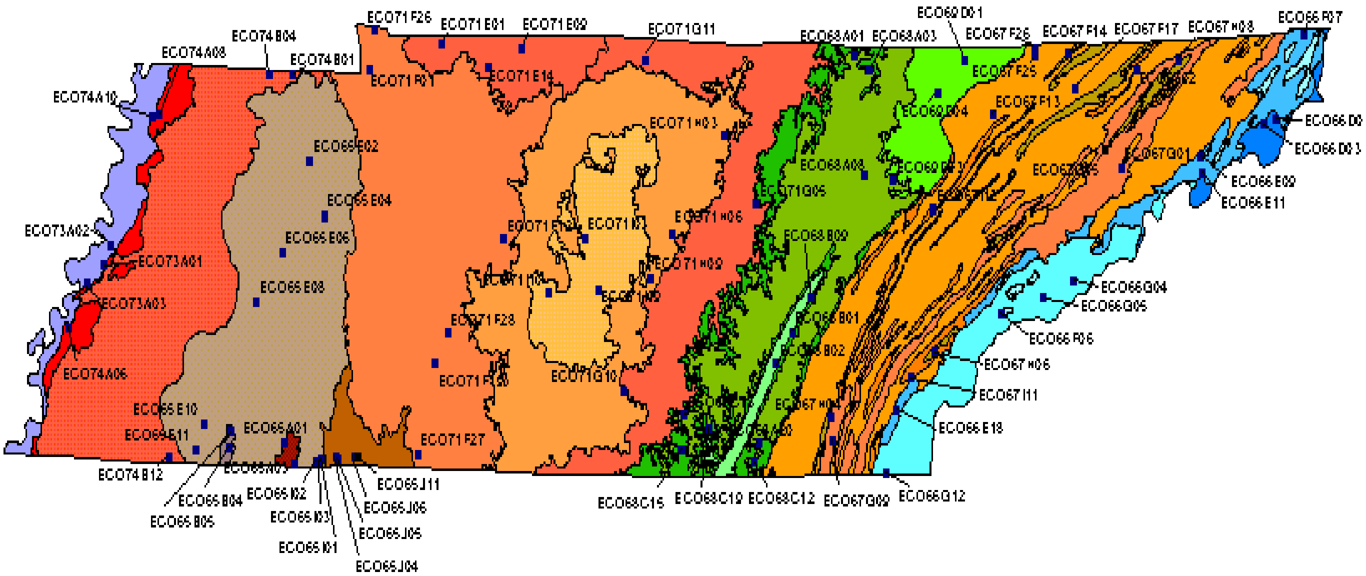

Figure A-1. Tennessee Level IV ecoregions and locations of reference streams.

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

How Are Reference Stream Data Being Used?
For the first time, the DWPC has regionally-based chemical, physical, and biological data representing least impacted conditions in Tennessee. These data are important to our program and have multiple applications.

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

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

Finalizing the Ecoregion Reference Stream Nutrient Database
Additional steps are needed to finalize the ecoregion nutrient database:

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

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

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

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

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

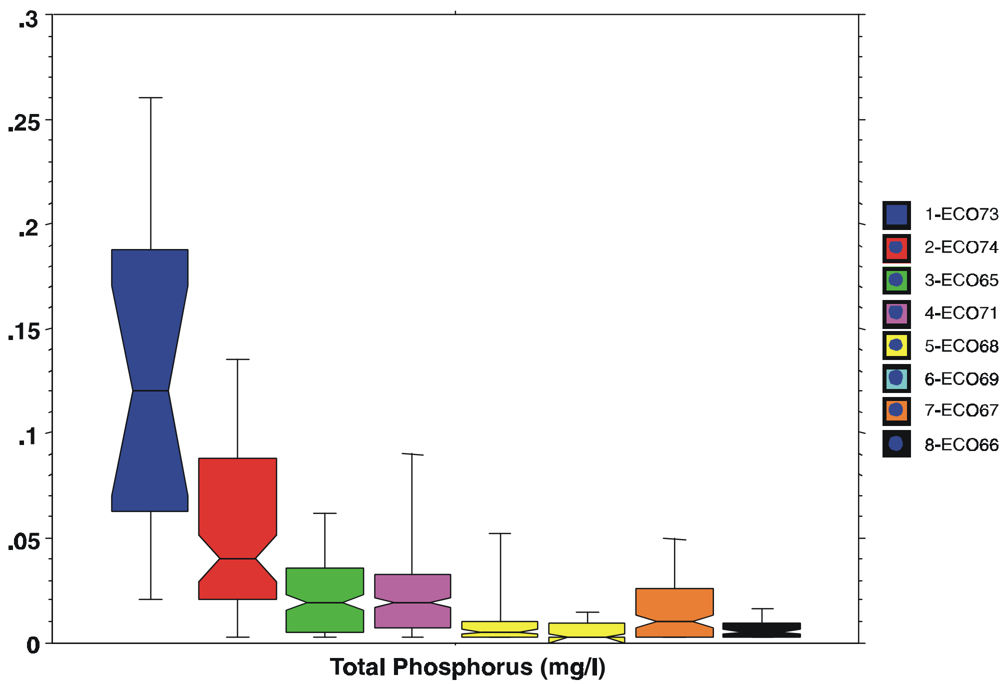

Figure A-3. Total phosphorus concentrations (μg/L) for reference streams within each ecoregion.

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

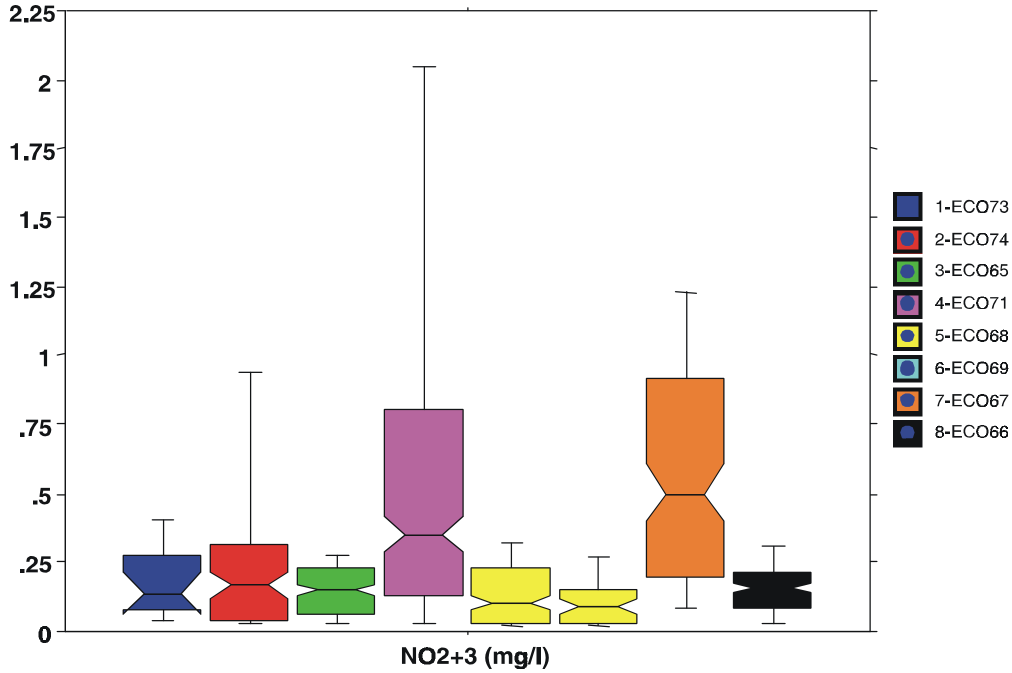

Figure A-4. Total nitrate-nitrite concentrations (mg/L) for reference streams within each ecoregion.

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

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

Data Relationships

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

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

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

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

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

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

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

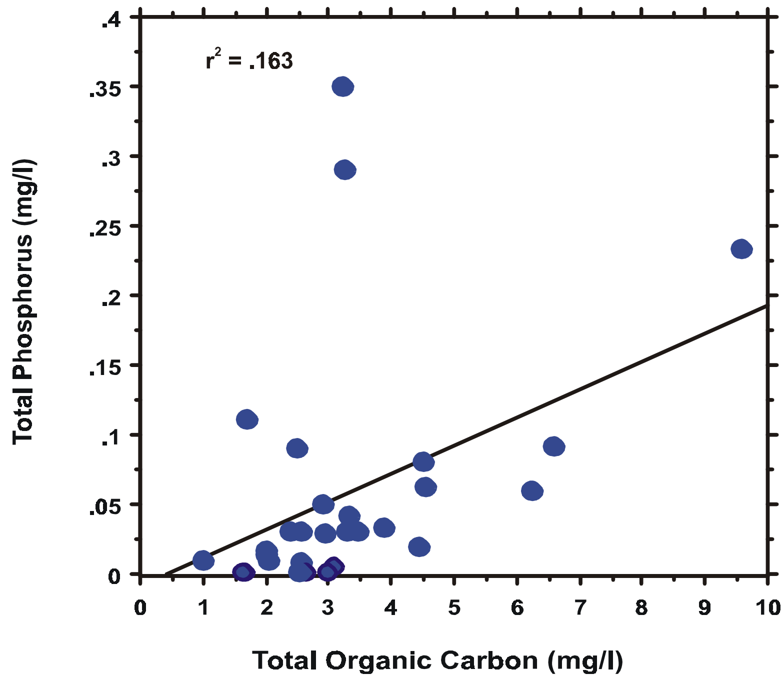

Figure A-5. Relationship between total phosphorus and TOC (r^2 value = 0.163) in ecoregion 67g.

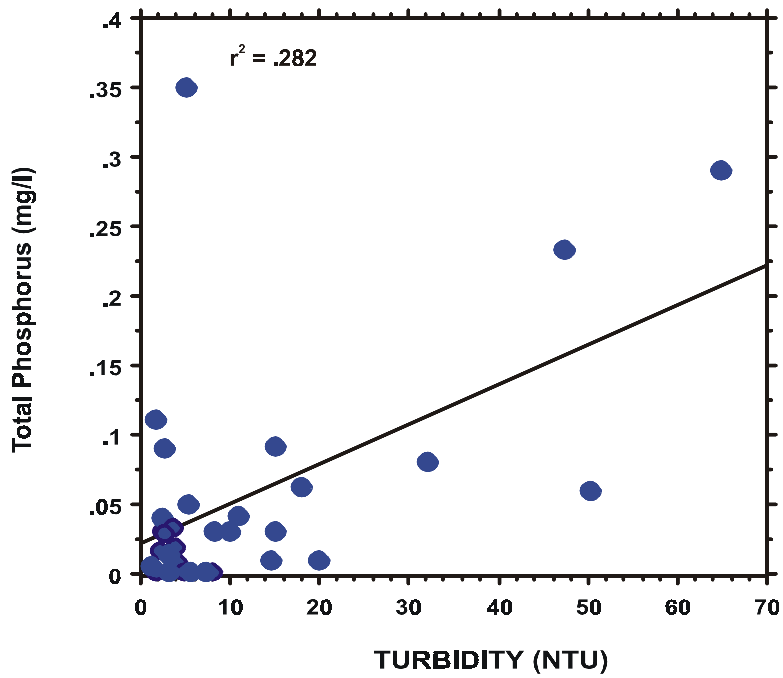

Figure A-6. Relationship between total phosphorus and turbidity (r^2 value = 0.282) in ecoregion 67g.

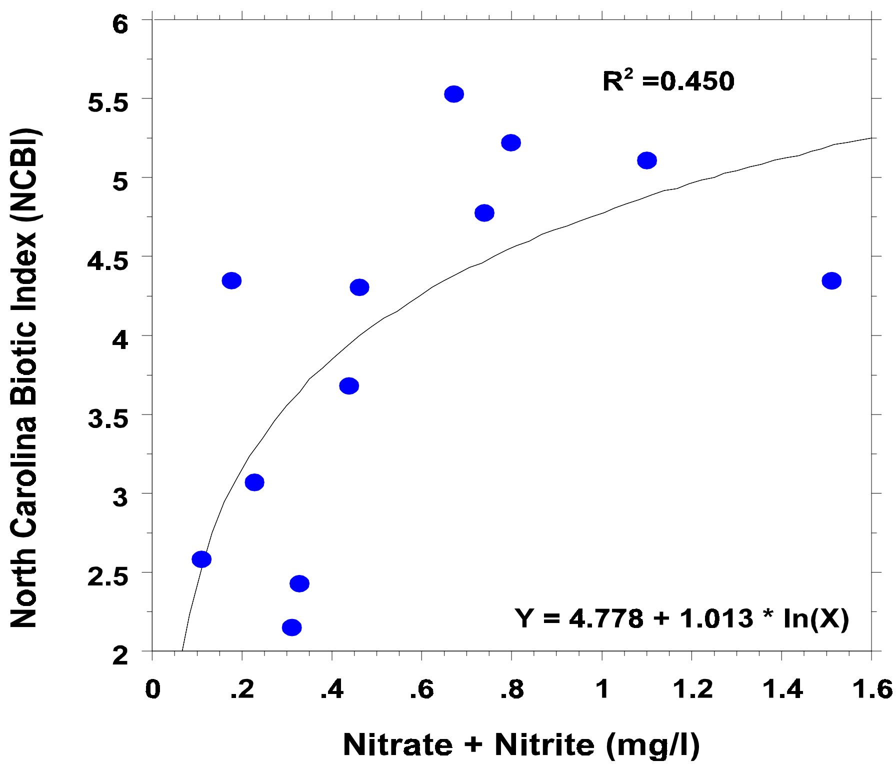

Figure A-7. Relationship between nitrate-nitrite levels and the Hilsenhoff Biotic Index.

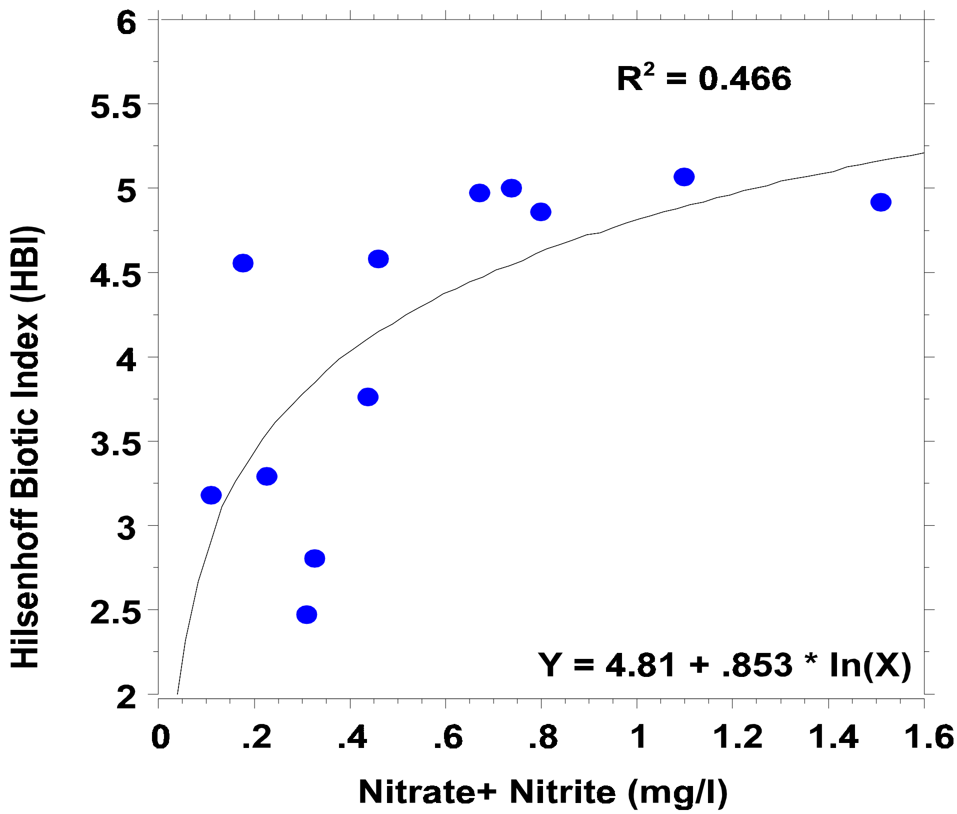

Figure A-8. Relationship between nitrate-nitrite levels and the North Carolina Biotic Index (NCBI).

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

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

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

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

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

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

SUMMARY

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

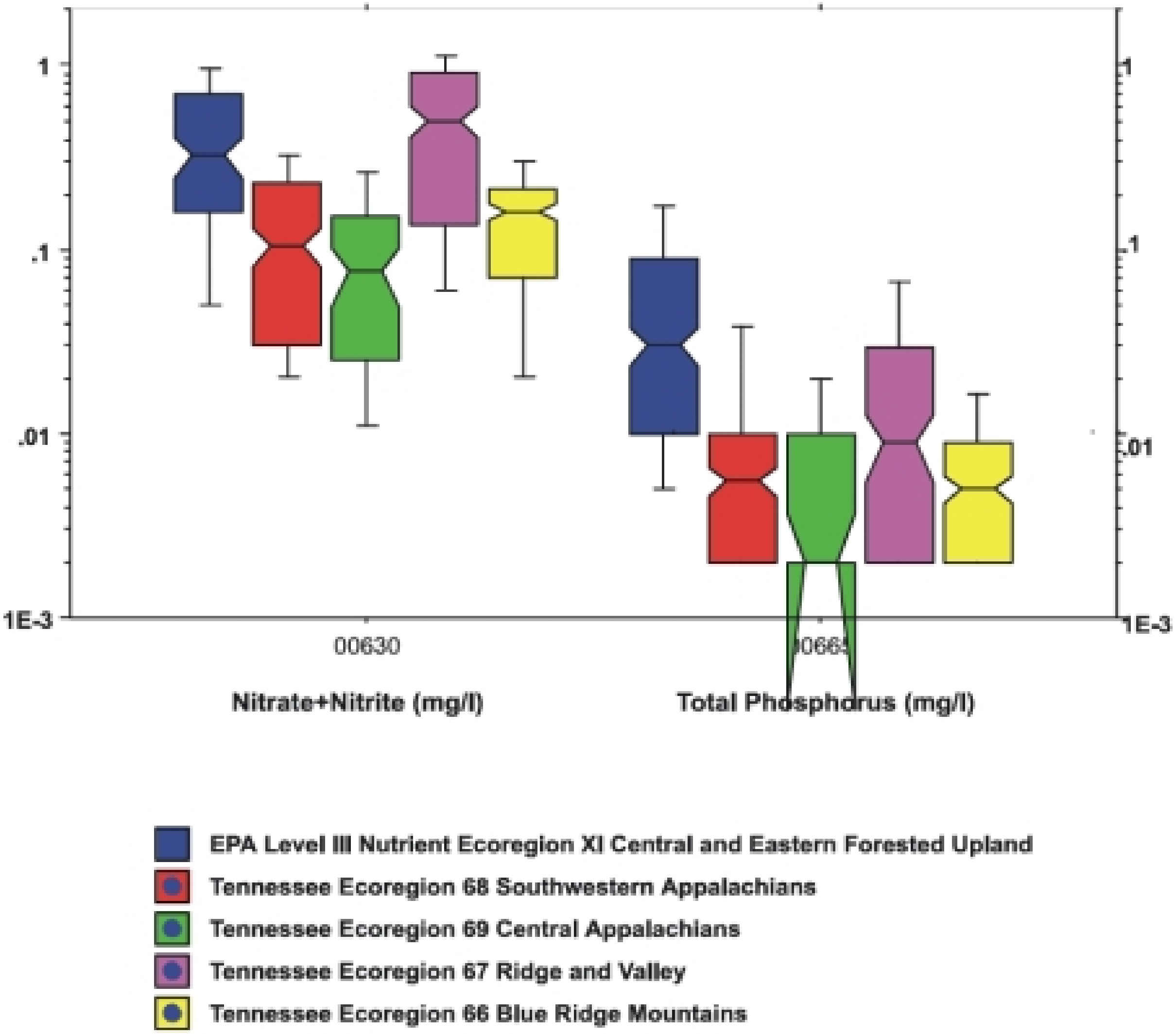

Figure A-9. Comparison of EPA Nutrient Ecoregion Region XI data to the Tennessee reference stream database for the same geographic area.

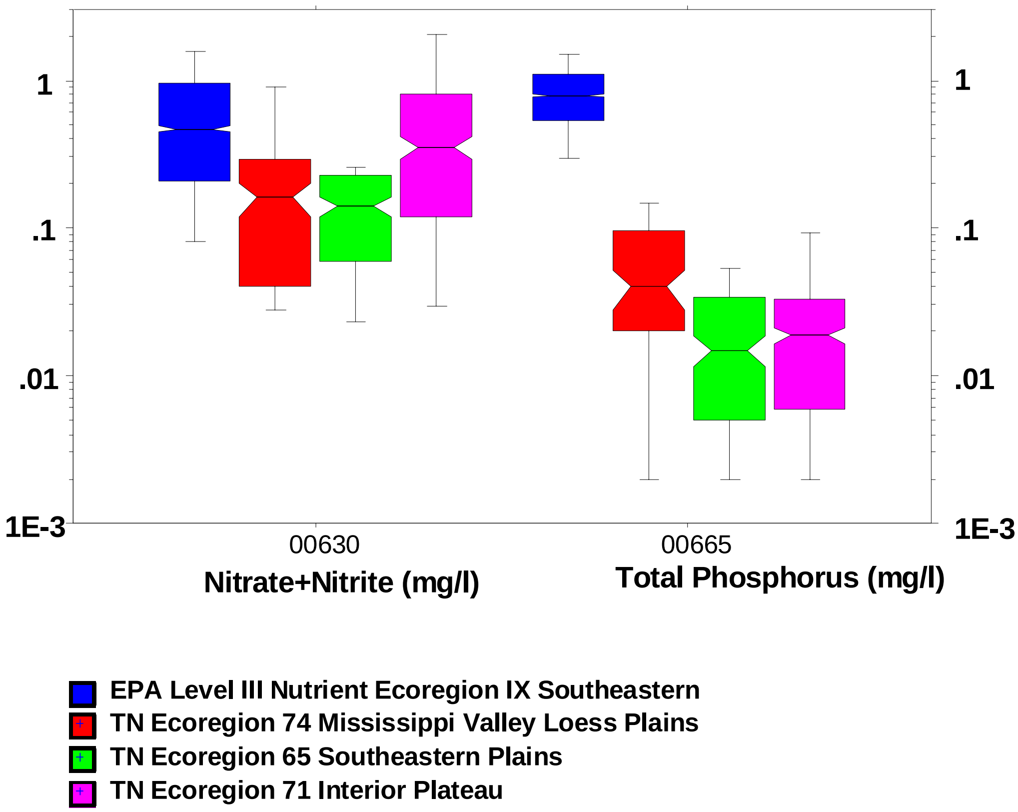

Figure A-10. Comparison of EPA Nutrient Ecoregion Region IX data to the Tennessee reference stream database for the same geographic area.

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

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

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

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

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

REFERENCES

TNDEC (Tennessee Department of Environment and Conservation). 1996. Standard operating procedure for modified clean technique sampling protocol. Tennessee Department of Environment and Conservation, Division of Water Pollution Control, Nashville, Tennessee.

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

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

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

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

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

INTRODUCTION

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

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

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

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

QUESTIONS ADDRESSED BY THE VNRP

What Are Current and Desired Algae Levels in the River?

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

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

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

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

What Actions Seem Most Likely to Reduce Algal Levels?

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

How Much Must Nutrients Be Reduced to Achieve Algal Targets?

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

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

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

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

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

How Was a Margin of Safety Incorporated in the VNRP?

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

What Actions Are Expected to Achieve Needed Load Reductions?

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

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

How Will Progress Towards the Targets Be Determined?

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

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

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

References

Bothwell, M. L. 1989. Phosphorus-limited growth dynamics of lotic periphytic diatom communities: areal biomass and cellular growth rate responses. Can. J. Fish. Aqua. Sci. 46:1293-1301.

Braico, R. D. 1973. Dissolved oxygen and temperature diurnal variations in the Clark Fork River, August 1973. Rep. to MT Dept. Health and Env. Sciences (now the Dept. Env. Quality).

Brick, C. & J. Moore. 1996. Diel variation of trace metals in the upper Clark Fork River, MT. Env. Sci. Tech. 30(6):1953-60.

Dodds, W. K., V. H. Smith & B. Zander. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: a case study of the Clark Fork River. Water Res. 31(7):1738-50.

Ingman, G. 1992. A rationale and alternatives for controlling nutrients and eutrophication problems in the Clark Fork River basin. Mt Dept. Health and Env. Sciences, Helena, MT.

Johnson, H. E. & C. L. Schmidt. 1988. Clark Fork basin status report & action plan. MT. Governor's Office, Helena, MT.

Land & Water Consulting. 1996. Water quality status and trends monitoring system for the Clark Fork-Pend Oreille Watershed. Rep. To MT Dept. Env. Quality.

Nordin, R. N. 1985. Water quality criteria for nutrients and algae (technical appendix). British Columbia Ministry of the Environment. Victoria, BC. 104 pp.

USEPA. 1993b. Clark Fork-Pend Oreille Basin Water Quality Study. EPA 910/R-93-006.

Watson, V. J. 1989a. Dissolved Oxygen levels in the Clark Fork River. Proc. Mont. Acad. Sci. 49:146-162.

________. 1989b. Control of attached algae by nitrogen and phosphorus in the Clark Fork River. Proc. Symp. Headwaters Hydrology, Amer. Water Res. Assn. Missoula, MT pp 287-97.

________ and B. Gestring. 1996. Monitoring algae levels in the Clark Fork River. Intermountain J. Sci. 2(2):17-26.

Welch, E. B., J. M. Jacoby, R. R. Horner and M. R. Seeley. 1988. Nuisance biomass levels of periphytic algae in streams. Hydrobiologia 157:161-8.

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

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

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

INTRODUCTION

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

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

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

METHODS

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

NUTRIENT INDICATORS OF TROPHIC CONDITION

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

NATURAL FACTORS THAT INFLUENCE NUTRIENT INDICATORS OF TROPHIC CONDITION

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

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

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

[1] Sum of dissolved NH_4-N + dissolved NO_2+NO_3-N + total organic N
[2] Sum of dissolved ortho-PO_4 + particulate phosphorus
[3] Total phosphorus as P (USGS WATSTORE code 00665)

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

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

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

ALGAL INDICATORS OF TROPHIC CONDITION

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

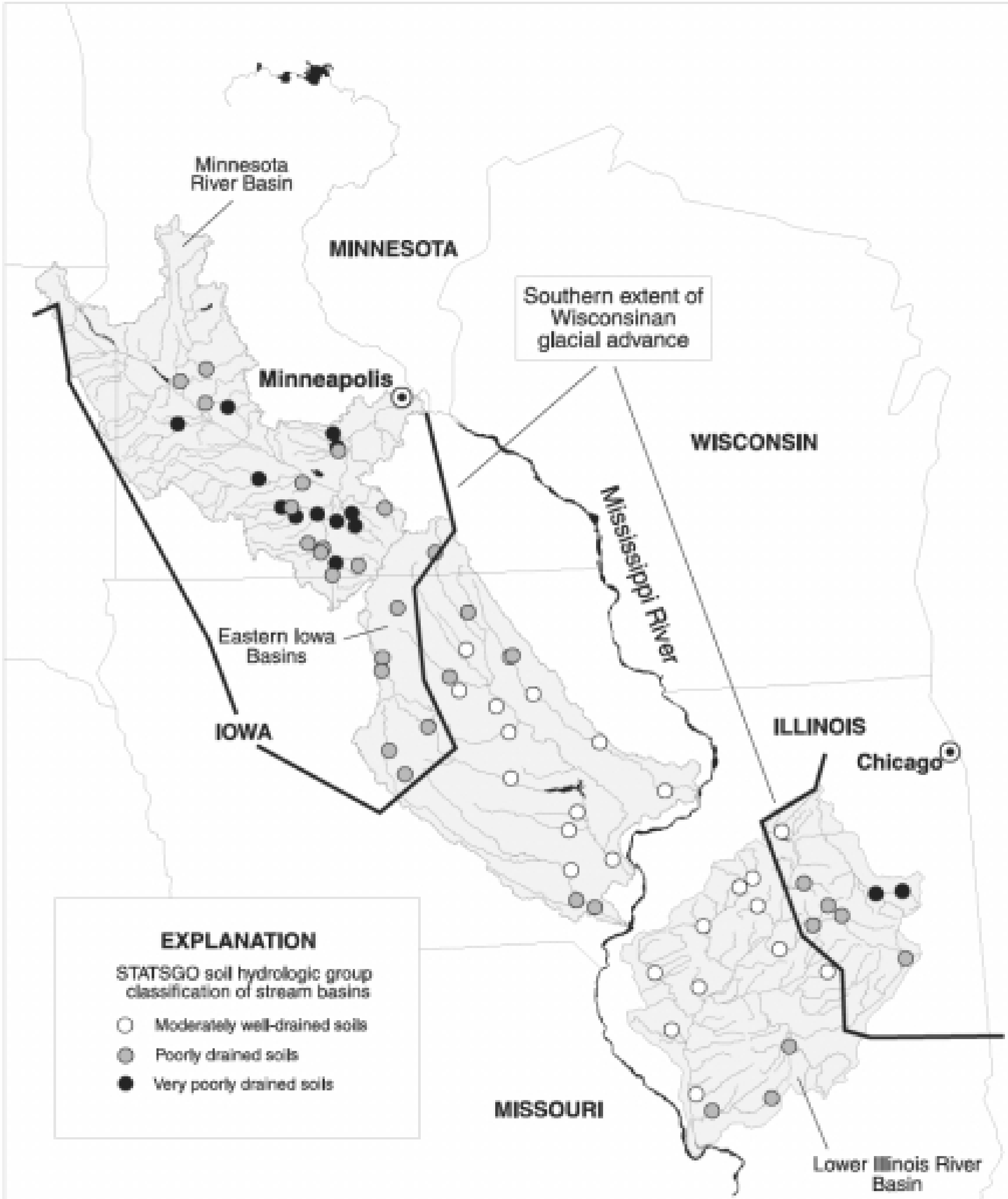

Figure A-11. Classification of Midwestern streams and rivers relative to basin soil-drainage characteristics and relation with southern extent of Wisconsin glacial advance.

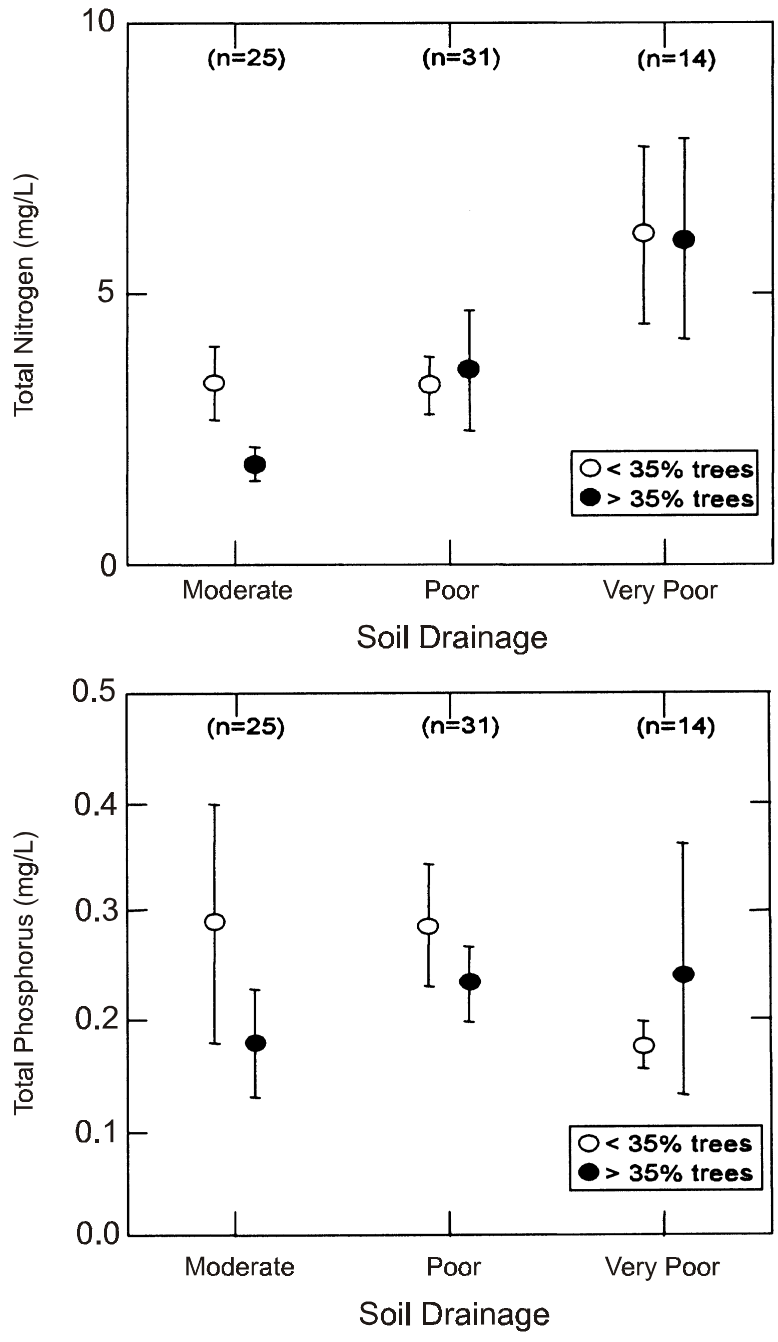

Figure A-12. Total nitrogen and phosphorus concentrations relative to soil drainage and riparian conditions in Midwestern streams and rivers.

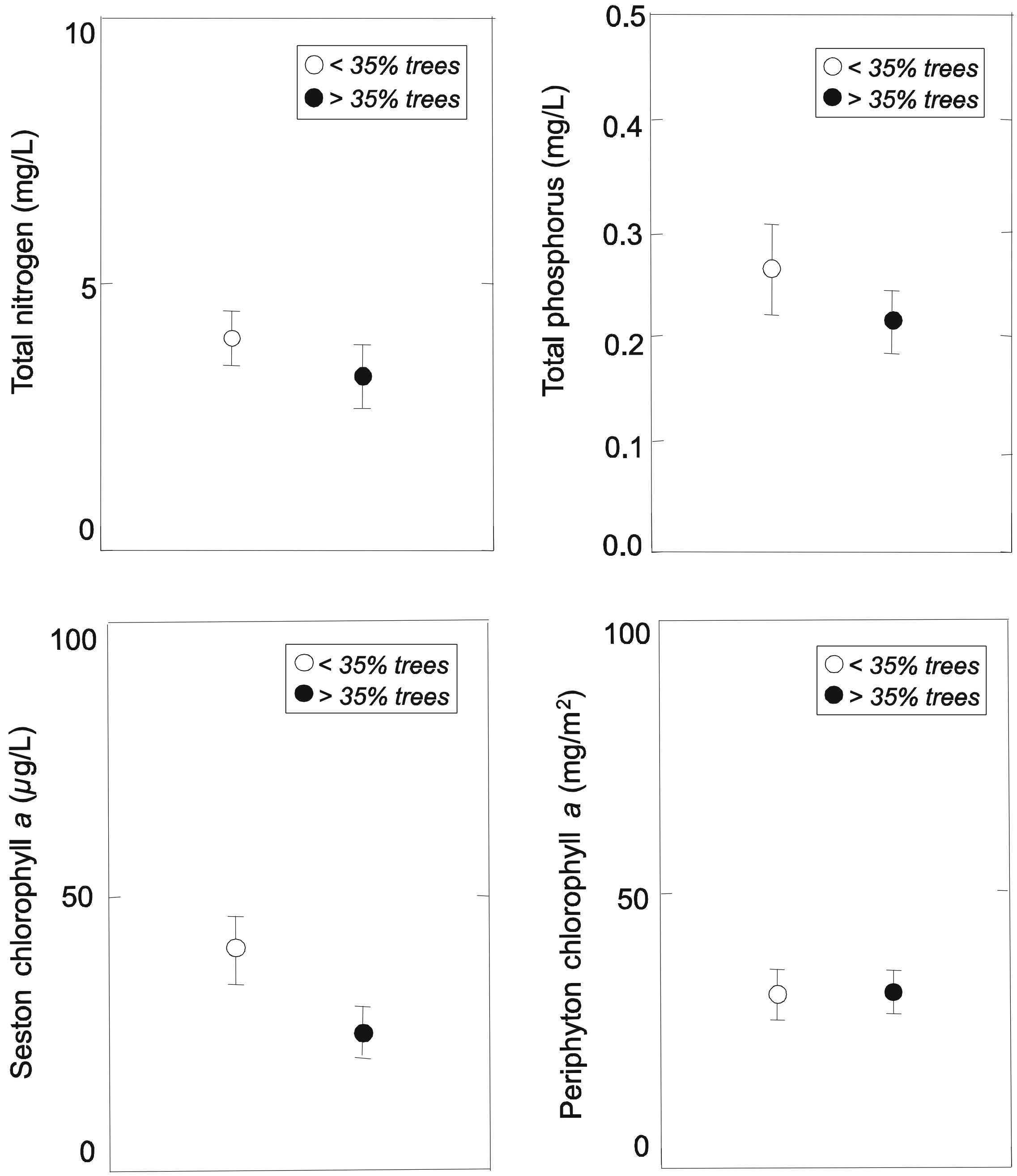

Figure A-13. Average concentrations of total nitrogen, total phosphorus, seston chlorophyll *a*, and periphyton chlorophyll *a* in relation to riparian-zone conditions.

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

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

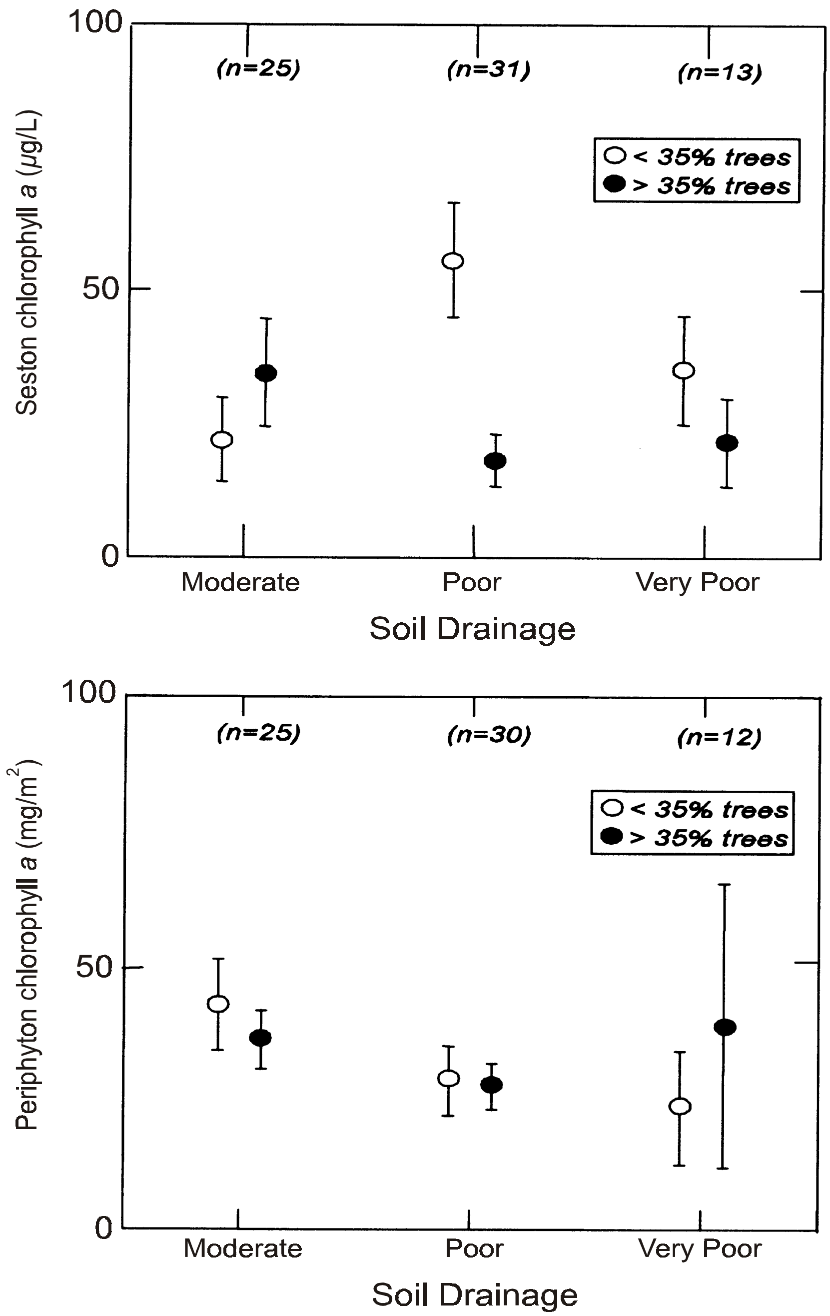

Figure A-14. Seston and periphyton chlorophyll a values relative to soil drainage and riparian conditions in Midwestern streams and rivers.

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

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

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

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

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

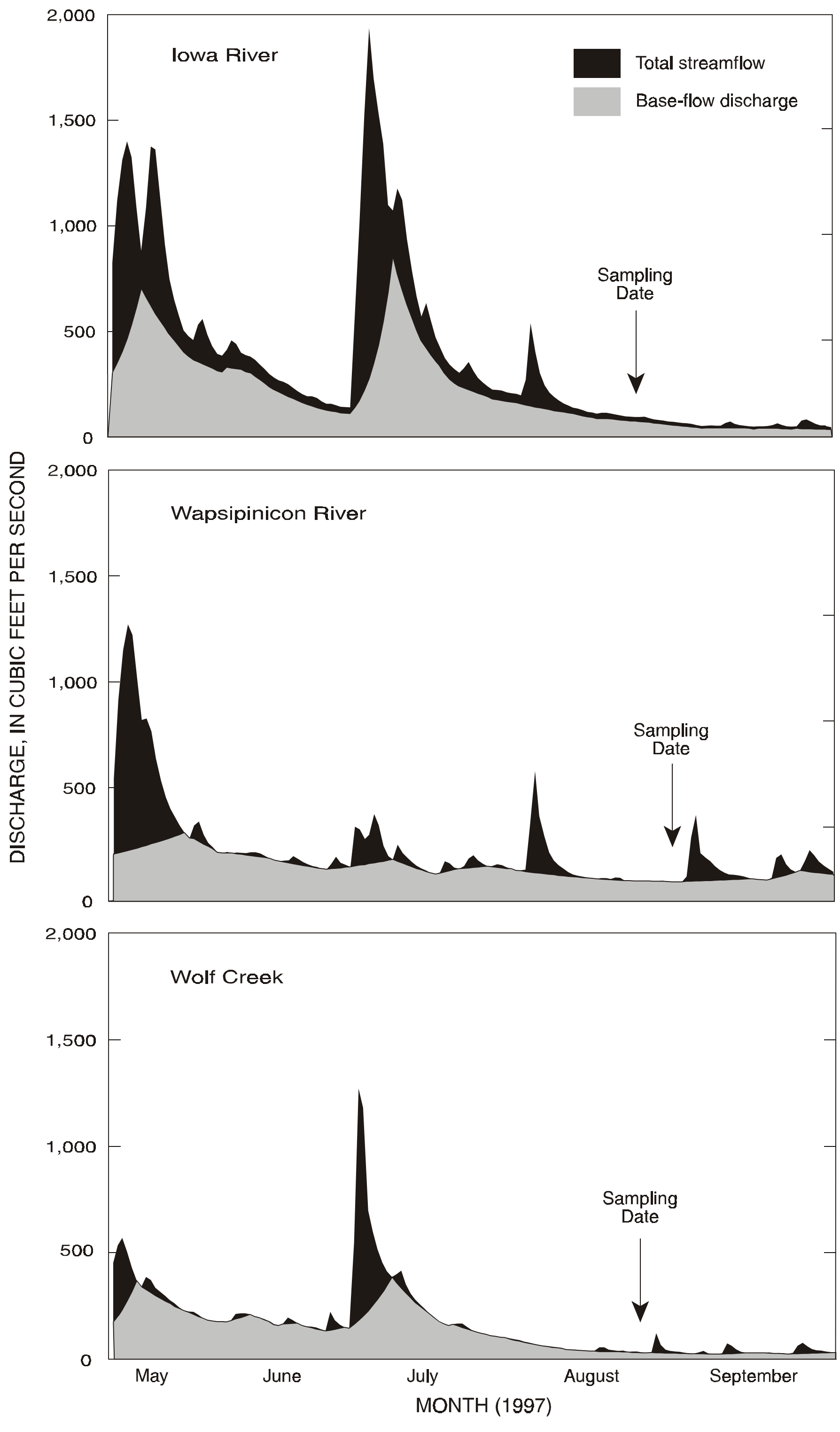

Figure A-15. Total and base flow discharge for selected Midwestern streams and rivers in relation to collection date of water quality samples.

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

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

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

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

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

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

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

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

SUMMARY AND IMPLICATIONS FOR ESTABLISHING AND MONITORING ALGAL-NUTRIENT CRITERIA

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

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

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

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

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

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

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

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

LITERATURE CITED

Biggs, B.J.F. and Close, M.E., 1989, Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients. Freshwater Biology, v. 22, p. 209-231.

Biggs, B.J.F., 1996, Patterns in benthic algae in streams. *In*: Algal Ecology—Freshwater benthic ecosystems, Stevenson, R.J., Bothwell, M.L., and Lowe, R.L., eds., Academic Press, San Diego, CA, p. 31-56.

Carlson, R.E., 1977, A trophic state index for lakes. Limnology and Oceanography, v. 22, p. 361-369.

Coupe, R.H., Goolsby, D.A., Iverson, J.L., Markovichick, D.J., and Zaugg, S.D., 1995, Pesticide, nutrient, water-discharge and physical-property data for the Mississippi River and some of its tributaries, April 1991-September 1992. U.S. Geological Survey Open-File Report 93-406, 66 p.

Darley, W.M., 1982, Algal biology: A physiological approach. Blackwell Scientific Publications, Oxford, U.K., 168 p.

Dodds, W.K., Jones, J.R., and Welch, E.B., 1998, Suggested classification of stream trophic state: Distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Resources, v. 32, no. 5 p. 1455-1462.

Dodds, W.K., Smith, V.H., and Zander, B., 1997, Developing nutrient targets to control benthic chlorophyll levels in streams: A case study of the Clark Fork River. Water Resources, v. 31, no. 7, p. 1738-1750.

Dodds, W.K., and Welch, E.B., 2000, Establishing nutrient criteria in streams. Journal of the North American Benthological Society, v. 19, p. 186-196.

Fitzpatrick, F.A., Waite, I.R., D'Arconte, P.J., Meador, M.R., Maupin, M.A., and Gurtz, M.E., 1998, Revised methods for characterizing stream habitat in the National Water-Quality Assessment Program. U.S. Geological Survey Water-Resources Investigations Report 98-4052, 67 p.

Fuhrer, G.J., Gilliom, R.J., Hamilton, P.A., Morace, J.L., Nowell, L.H., Rinella, J.F., Stoner, J.D., and Wentz, D.A., 1999, The quality of our Nation's waters. Nutrients and pesticides. U.S. Geological Survey Circular 1225, 82 p.

Goolsby, D.A., Coupe, R.C., and Markovchick, D.J., 1991, Distribution of selected herbicides and nitrate in the Mississippi River and its major tributaries, April through June 1991. U.S. Geological Survey Water-Resources Investigations Report 91-4163, 35 p.

Harris, M.A., and Porter, S.D., (unpublished manuscript), Relating epidendric macroinvertebrate communities to physical and chemical factors in upper Midwest streams. U.S. Geological Survey Water-Resources Investigations Report.

Heintz, A.J., 1970, Low-flow characteristics of Iowa streams through 1966: Iowa Natural Resources Council Bulletin No. 10, 176 p.

Lohman, K., Jones, J.R., and Perkins, B.D., 1992, Effects of nutrient enrichment and flood frequency on periphyton biomass in Northern Ozark streams. Canadian Journal of Fisheries and Aquatic Sciences, v. 49, p. 1198-1205.

O'Hearn, M.O., and Gibb, J.P., 1980, Groundwater discharge to Illinois streams. Illinois Institute of Natural Resources, State Water Survey Division, Groundwater Section, SWS Contract Report 246, Champaign, Illinois, 31 p.

Omernik, J.M., 1986, Ecoregions of the United States. U.S. Environmental Protection Agency, Corvalis Environmental Research Laboratory, 1 p.

Rabalais, N.N., Turner, R.E., Justic, D., Dortch, Q., Wiseman, W.J., and Sen Gupta, B.K., 1996, Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. Estuaries, v. 19, no. 2B, p. 386-407.

Reckhow, K.H., and Chapra, S.C., 1983, Engineering approaches for lake management, Volume 1. AnnArbor Science, Butterworth Publishing Company, Woburn, Mass., 340 p.

Rosen, B.H., and Lowe, R.L., 1984, Physiological and ultrastructural responses of *Cyclotella meneghiniana* (Bacillariophyta) to light intensity and nutrient limitation. Journal of Phycology, v. 20., p. 173-183.

Shelton, L.R., 1994, Field guide for collection and processing stream-water samples for the National Water Quality Assessment Program, U.S. Geological Survey Open-File Report 94-455, 42 p.

Sorenson, S.K., Porter, S.D., Akers, K.K.B., Harris, M.A., Kalkhoff, S.J., Lee, K.E., Roberts, L.R., and Terrio, P.J., 1999, Water quality and habitat conditions in upper Midwest streams relative to riparian vegetation and soil characteristics, August 1997: Study design, methods, and data. U.S. Geological Survey Open-File Report 99-202, 53 p.

Squillace, P.J., Caldwell, J.P., Schulmeyer, P.M., and Harvey, C.A., 1996, Movement of agricultural chemicals between surface water and ground water, lower Cedar River basin, Iowa, U.S. Geological Survey Water-Supply Paper 2448, 59 p.

Turner, R.E., and Rabalais, N.N., 1994, Coastal eutrophication near the Mississippi River delta. Nature,v. 368, p. 619-621.

Van Niewenhuyse, E.E., and Jones, J.R., 1996, Phosphorus-chlorophyll relationships in temperate streams and its variation with stream catchment area. Canadian Journal of Fisheries and Aquatic Sciences, v. 53, p. 99-105.

Walton, W.C., 1965, Ground water recharge and runoff in Illinois. Report of Investigation 48, Illinois State Water Survey, Urbana, Illinois, 55 p.

Watson, V., and Gestring, B., 1996, Monitoring algae levels in the Clark Fork River. Intermountain Journal of Sciences, v. 2, no. 2, p. 17-26.

Welch, E.B., Jacoby, J.M., Horner, R.R., and Seeley, M.R., 1988, Nuisance biomass levels of periphytic algae in streams. Hydrobiologia, v. 157, p. 161-168.

Winter, T.C., Harvey, J.W., Franke, O.L., and Alley, W.M., 1998, Ground water and surface water—A single resource. U.S. Geological Survey Circular 1139, 79 p.

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

BOW RIVER, ALBERTA

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

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

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

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

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

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

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

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

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

LITERATURE CITED

Anderson, E .L., J. M. Jacoby, G. M. Schimek, E .B. Welch, and R .R. Horner. 1999. Periphyton removal related to phosphorus and grazer biomass level. Freshwater Biol. 41:633-651.

Barko, J. W., D. Gunnison, and S. R. Carpenter. 1991. Sediment interactions with submersed macrophyte growth and community dynamics. Aquatic Botany 41:41-65.

Charlton, S. E. D., H. R. Hamilton, and P. M. Cross. 1986. *The Limnological Characteristics of the Bow, Oldman and South Saskatchewan Rivers (1979-82)*. Alberta Environment, Edmonton, AL.

Elswick, D. A., B. W. Mar, and E. B. Welch. 2000. The use of dynamic modeling to predict periphyton algal biomass in the Bow River, Canada. Department of Civil and Environmental Engineering, University of Washington, Seattle.

Horner, R. R., E. B. Welch, M. R. Seeley, and J. M. Jacoby. 1990. Responses of periphyton to changes in current velocity, suspended sediment and phosphorus concentration. Freshwater Biol. 24:215-232.

Sosiak, A. J., Alberta Environment Protection, Calgary, Alberta. Personal communication (unpublished manuscript, "Long-term response of periphyton and macrophytes to reduced municipal nutrient loading to the Bow River [Alberta, Canada].")

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

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

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

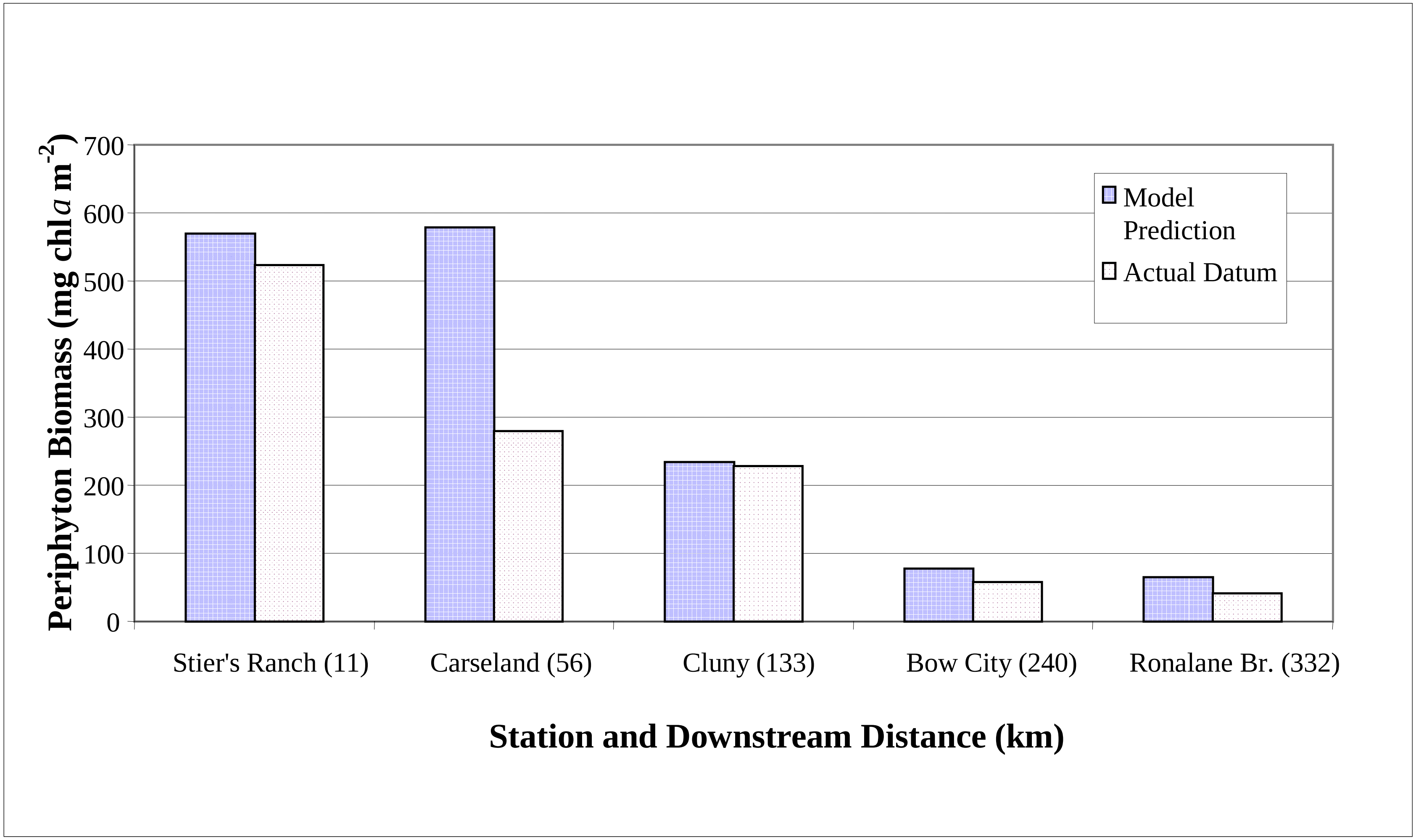

Figure A-16. Model simulation of periphyton biomass during low-flow conditions in the Bow River downstream from the Fish Creek STP, Calgary, Alberta, compared with actual 1997 data.

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

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

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

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

ULTIMATE DETERMINANTS

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

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

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

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

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

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

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

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

INTERMEDIATE DETERMINANTS

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

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

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

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

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

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

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

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

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

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

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

PROXIMATE DETERMINANTS

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

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

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

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

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

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

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

HYDROLOGIC VARIABILITY IN ARID LANDS STREAMS

While hydrologic variability is an important consideration in the development of nutrient standards for all streams, the spatial and temporal heterogeneity found in arid regions, the stark contrast between wet and dry, brings this variability into sharper relief. When viewing desert catchments from above, the observer is often presented with a dry landscape of high relief bisected by the string of glistening beads that is the spatially intermittent stream. The dry arroyos or quiet, disconnected pools and short reaches of wetted stream that characterize desert streams during dry periods are in complete contrast to the raging torrents that they can become at flood stage. This hydrologic variability and the unique chemical and biological characteristics of arid lands aquatic ecosystems may make the use of broad generalizations to explain nutrient regimes impossible.

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

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

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

In desert streams, surface discharge regimes may vary from completely dry, to flows as much as three to five orders of magnitude greater than mean annual flow, all within a period of hours or days. In comparison to streams in more mesic regions, the coefficient of variation of annual flow is 467% greater in arid land lands streams (Davies et al. 1994). The aquatic ecosystems structured by these often catastrophic and always chaotic flow regimes exhibit spatially and temporally heterogeneous structures and functions (sensu Thoms and Sheldon 1996) which may not allow the application of nutrient criteria derivation techniques applicable to more homogeneous environments.

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

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

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

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

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

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

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

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

CONCLUSIONS

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

STEPS FOR CHARACTERIZING THE NUTRIENT REGIME OF DESERT STREAM SYSTEMS

Selecting Index Sites and Periods

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

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

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

Physical/Structural Elements

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

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

Temporal Elements

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

Chemistry (other than TN and TP)

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

Biological Elements

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

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

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

REFERENCES

Baker, William L. 1990. Climatic and hydrologic effects on the regeneration of Populus angustifolia James along the Animas River, Colorado. Journal of Biogeography. 17(1): 59-73.

Barbour, M. G., J. H. Burk, and W. D. Pitts. 1980. *Terrestrial Plant Ecology*. The Benjamin/Cummings Publishing Company, Menlo Park, CA.

Blaney, H. and W. Criddle, 1962. Determining consumptive use and irrigation water requirements. USDA Technical Bulletin Number 1275, 59 pages.

Busch, D.E., and S.G. Fisher. 1981. Metabolism of a desert stream. Freshwater Biology 11:301-308.

Chauvet, E. and H. Decamps. 1989. Lateral interactions in a fluvial landscape: the River Garonne, France. Journal of the North American Benthological Society. 8(1) p 9-17.

Chorley, R. J. and B. A. Kennedy. 1971. Physical geography: a systems approach. Prentice-Hall, London.

Dahm, C.N., N.B. Grimm, P. Marmonier, H.M. Valett, and P. Vervier. 1998. Nutrient dynamics at the interface between surface waters and ground waters. Freshwater Biology 40:427-451.

Dahm, C. N., E. H. Trotter and J. R. Sedell, 1987. Role of anarobic zones and processes in stream ecosystem productivity. P 157-178 in R. A. Averett and D. M. McKnight, editors. Chemical quality of waterand the hydrologic cycle. Lewis, Chelsea, Michigan.

Davies, B. R., Thoms, M. C., Walker, K. F., O'Keeffe, J. H., and Gore, J. A. 1994. Dryland rivers: their ecology, conservation and management, in Calow, P. and Pets, G. E. (Eds) The Rivers Handbooy, Vol. 2, Blackwell Scientific, Oxford. 484-512

Dent, C.L., and N.B. Grimm. Spatial heterogeneity in stream water nutrient concentrations over successional time. Ecology: in press.

Dunne, T. and L. B. Leopold, 1978. Water in environmental planning. W. H. Freeman and Co. San Francisco.

Fisher, S.G. 1983. Succession in streams. Pages 7-27 in J. Barnes and G.W. Minshall, editors. Stream ecology: Application and testing of general ecological theory. Plenum Press, New York, New York, U.S.A.

Fisher, S.G. 1986. Structure and dynamics of desert streams. Pages 114-139. IN: W. Whitford (ed.). Pattern and Process in Desert Ecosystems. University of New Mexico Press, Albuquerque.

Fisher, S.G., L.J. Gray, N.B. Grimm, and D.E. Busch. 1982. Temporal succession in a desert stream following flash flooding. Ecological Monographs 52:93-110.

Fisher, S.G., and N.B. Grimm. 1983. Water quality and nutrient dynamics of Arizona streams. OWRT Project Completion Report A-106-ARIZ. Office of Water Research and Technology.

Fisher, S. G., and N. B. Grimm. 1988. Disturbance as a determinant of structure in a Sonoran Desert stream ecosystem. Internationale Vereinigung für Theoretische und Angewandt Limnologie, Verhandlungen 23:1183-1189.

Fisher, S.G., and N.B. Grimm. 1991. Streams and disturbance: are cross-ecosystem comparisons useful? pp 196-221 in J.C. Cole, G.M. Lovett, and S.E.G. Findlay, editors. Comparative analyses of ecosystems: patterns, mechanisms and theories. Springer-Verlag, New York, New York, U.S.A.

Fisher, S.G., N.B. Grimm, E. Marti, and R. Gomez. 1998a. Hierarchy, spatial configuration, and nutrient cycling in streams. Australian Journal of Ecology 23: 41-52.

Fisher, S.G., N.B. Grimm, E. Martí., J.B. Jones, Jr., and R.M. Holmes. 1998 (b). Material spiralling in river corridors: a telescoping ecosystem model. Ecosystems 1:19-34.

Fisher, S.G., and W.L. Minckley. 1978. Chemical characteristics of a desert stream in flash flood. Journal of Arid Environments 1:25-33.

Fuller, W. H. 1975. Soils of the desert southwest. University of Arizona Press. 102p

Graff, W. L. 1988. *Fluvial Processes in Dryland Rivers*. Springer-Verlag, New York.

Gray, L.J. 1981. Species composition and life histories of aquatic insects in a lowland Sonoran Desert stream. American Midland Naturalist 106:229-242.

Grimm, N.B. 1987. Nitrogen dynamics during succession in a desert stream. Ecology 68:1157-1170.

Grimm, N. B. 1988. Role of macroinvertebrates in nitrogen dynamics of a desert stream. Ecology 69: 1884-1893.

Grimm, N.B. 1992. Biogeochemistry of nitrogen in arid-land stream ecosystems. Journal of the Arizona-Nevada Academy of Science 26:130-146.

Grimm, N.B., and S.G. Fisher. 1986a. Nitrogen limitation potential of Arizona streams and rivers. Journal of the Arizona-Nevada Academy of Science 21:31-43.

Grimm, N.B., and S.G. Fisher. 1986b. Nitrogen limitation in a Sonoran Desert stream. Journal of the North American Benthological Society 5:2-15.

Grimm, N.B., and S.G. Fisher. 1989. Stability of periphyton and macroinvertebrates to disturbance by flash floods in a desert stream. Journal of the North American Benthological Society 8:293-307.

Grimm, N.B., S.G. Fisher, and W.L. Minckley. 1981. Nitrogen and phosphorus dynamics in hot desert streams of Southwestern U.S.A. Hydrobiologia 83:303-312.

Grimm, N.B., and K.C. Petrone. 1997. Nitrogen fixation in a desert stream ecosystem. Biogeochemistry 37:33-61.

Holmes, R.M., S.G. Fisher, and N.B. Grimm. 1994. Parafluvial nitrogen dynamics in a desert stream ecosystem. Journal of the North American Benthological Society 13:468-478.

Holmes, R.M., J.B. Jones, Jr., S.G. Fisher, and N.B. Grimm. 1996. Denitrification in a nitrogen-limited stream ecosystem. Biogeochemistry 33:125-146.

Jones Jr., J.B. 1995. Factors controlling hyporheic respiration in a desert stream. Freshwater Biology 34:101-109.

Jones, J.B., Jr., S.G. Fisher, and N.B. Grimm. 1995. Nitrification in the hyporheic zone of a desert stream ecosystem. Journal of the North American Benthological Society 14:249-258.

Lowrance, R., R. Todd, J. Fail Jr., O. Hendrickson Jr., R. Leonard and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34(6) p 374-377.

Martí, E., S.G. Fisher, J.J. Schade, and N.B. Grimm. in press (b) Effect of flood frequency on hydrological and chemical linkages between streams and their riparian zones: an intermediate disturbance model. J.B. Jones, Jr., and P.J. Mulholland, editors. Surface-subsurface interactions in streams. Book chapter.

Martí, E., S.G. Fisher, J.J. Schade, J.R. Welter, and N.B. Grimm. in press (a) Hydrological and chemical linkages between streams and their riparian zones: an intermediate disturbance model. Internationale Vereinigung fur Theoretische und Angewandt Limnologie, Verhandlungen 27.

Marti, E., N.B. Grimm, and S.G. Fisher. 1997. Pre- and post-flood nutrient retention efficiency in adesert stream ecosystem. Journal of the North American Benthological Society 16:805-819.

Molles, M. C., Jr., and C. N. Dahm. 1990. A perspective on El Nino and La Nina: global implications for stream ecology. Journal of the North American Benthological Society. 9:68-76.

O'Neill, R. V. , D. L. DeAngelis, J. B. Waide and T. F. H. Allen, 1986. A hierarchical concept of ecosystems. Princton University Press, Princton, New Jersey.

Peterjohn, W. T. and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65(5) p 1466-1475.

Peterson, C.G., and N.B. Grimm. 1992. Temporal variation in enrichment effects during periphyton succession in a nitrogen-limited desert stream ecosystem. Journal of theNorth American Benthological Society 11:20-36.

Pickett, S. T. A., J. J Kolasa, and S. L. Collins. 1989. The ecological concept of disturbance and its expression at various hierarchical levels. Oikos 54: 129-136.

Pinay, G. and H. Decamps. 1988. The role of riparian woods in regulatimg nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. Regulated Rivers; Research and Management Volume 2. p 507-516.

Poff, N.L., and J.V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences. 46:1805-1818.

Puckridge, J.T., Sheldon, F., Walker, K.F. & Boulton, A.J. 1998. Flow variability and the flood pulse concept in river ecology. Australian Journal of Marine and Freshwater Research.

Schade, J.D., and S.G. Fisher. 1997. The influence of leaf litter on a Sonoran Desert stream ecosystem. Journal of the North American Benthological Society 16:612-626.

Stanley, E.H. 1999. Personal Communication Department of Zoology, University of Wisconsin, Madison, WI

Stanley, E.H. and A.J. Boulton. 1995. Hyporheic processes during flooding and drying in a Sonoran Desert stream. I. Hydrologic and chemical dynamics. Archiv fur Hydrobiologie 134:1-26.

Stanley, E.H., S.G. Fisher, and N.B. Grimm. 1997. Ecosystem expansion and contraction: a desert stream perspective. BioScience 47:427-435.

Stanley, E.H. and H.M. Valett. 1992. Interaction between drying and the hyporheic zone of a desert stream ecosystem. pp 234-249 in P. Firth and S.G. Fisher, editors. Climate Change and Freshwater Ecosystems. Springer-Verlag, New York, New York, U.S.A.

Stevenson, R. J. 1997. Scale dependent determinants and consequences of benthic algal heterogeneity. Journal of the North American Benthological Society. 16(1):248-262

Stromberg, J. C., D. T. Patten and B. D. Richter. 1991. Flood flows and dynamics of Sonoran riparian forests. Rivers 2(3):221-235.

Stumm, W. G. and J. J. Morgan, 1981. Aquatic chemistry, John Wiley and Sons, New York.

Thoms, M.C. and Sheldon, F. 1996. The importance of channel complexity for ecosystem processing: An example of the Barwon-Darling River. Pp:111-118 in Rutherfurd, I. (ed) Stream Management in Australia. CRC for Catchment Hydrology, Melbourne.

Valett, H.M., S.G. Fisher and E.H. Stanley. 1990. Physical and chemical characteristics of the hyporheic zone of a Sonoran Desert stream. Journal of the North American Benthological Society 9:201-215.

Valett, H.M., S.G. Fisher, N.B. Grimm, and P. Camill. 1994. Vertical hydrologic exchange and ecological stability of a desert stream ecosystem. Ecology75:548-560.

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