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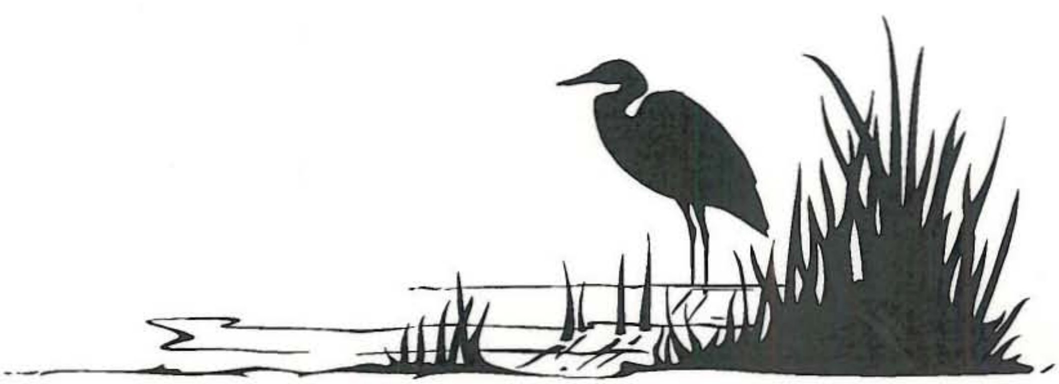

National Nutrient Assessment Workshop
December 4-6, 1995
Washington, D.C.

Introduction

Nutrient overenrichment is one of the leading causes of water quality problems in the United States. The National Water Quality Inventory 1994 Report to Congress Executive Summary cites nutrients (nitrogen and phosphorus) as one of the leading causes of water quality impairment in our Nation's rivers, lakes, and estuaries. While nutrients are essential to the health of aquatic ecosystems, excessive nutrient loadings can result in the growth of aquatic weeds and algae, leading to oxygen depletion, increased fish and macroinvertebrate mortality, and other water quality and habitat impairments.

Over the years, the Environmental Protection Agency's Office of Water has issued a number of technical guidance documents and supported the development of water quality simulation models and load estimating models to assess the impacts of urban, rural, and mixed land use activities on receiving waters. In addition, some States currently have water quality standards that incorporate criteria, primarily narrative, aimed at controlling problems associated with overenrichment. However, in order for State and local agencies to better understand and manage nutrient impacts to surface waters, additional work is required.

EPA established a Nutrient Task Force in July 1993 to gather existing data on nutrient problems and currently available tools. The Task Force recommended that EPA provide additional assistance to States in developing and implementing appropriate nutrient endpoints, assessment methodologies, and models. The first step in carrying out the recommendations of the Task Force was a nutrient assessment workshop, which was held in Washington, DC, on December 4-6, 1995. The workshop was organized around plenary and breakout group discussions on four major waterbody types: estuarine and coastal waters; lakes, impoundments/reservoirs, and ponds; rivers and streams; and wetlands. Issue papers describing the state-of-the science, gaps, and user needs in terms of nutrient assessment tools and methodologies for each waterbody type were developed and used as foundations for group discussion.

The primary goals of the workshop were to:

- Identify the full range of potential nutrient overenrichment endpoints, including early warning indicators, assessment methodologies, and models available for application to various types of waterbodies.
- Identify what gaps exist in providing a full range of simple to complex nutrient overenrichment endpoints assessment methodologies, and models for a wide range of management applications.

- Evaluate the existing and potential nutrient overenrichment endpoints, assessment methodologies, and models in terms of the state-of-the-science supporting their applications, data input requirements, and relative ease of transferability/application to various types of waterbodies and geographically diverse areas.

- Evaluate and begin to prioritize specific user needs for nutrient overenrichment endpoints, assessment methodologies, models to address nutrient problems in various waterbody types and in geographically diverse areas.

- Examine ways to apply nutrient assessment tools across various waterbody types, geographical areas, and ecoregions.

Recognizing the long history EPA and its partners have had in addressing nutrient overenrichment impacts on aquatic ecosystems, we view this workshop and the subsequent development of an Agency strategy as a turning point in our commitment to support development of the tools necessary to set clear, quantifiable endpoints for addressing nutrient overenrichment problems. Without definable endpoints - whether they are comprehensive narrative descriptions, site- or regional-specific numerical criteria, or correlative relationships defining water quality objectives - and the supporting assessment tools applicable at the watershed and individual waterbody level, it is difficult to design and implement effective management actions that are tailored to stem the increasing nutrient overenrichment of the nation's surface waters.

Through development and implementation of a national strategy, it is our hope that we can better support the development, validation, and implementation of the tools State and local natural resource managers require to address nutrient overenrichment problems in surface waters.

Robert H. Wayland III
Director
Office of Wetlands, Oceans and Watersheds

Tudor Davies
Director
Office of Science and Technology

Executive Summary

The National Nutrient Assessment Workshop was held with the purpose of asking for expert recommendations on the components of a national strategy addressing nutrient overenrichment issues. There were approximately fifty participants in attendance, representing a diversity of organizations and geographic regions (a list of the participants is attached at the end of these proceedings).

The participants met over the course of two and a half days to discuss pertinent eutrophication issues and potential nutrient assessment and management tools. Each expert participated in one of four groups that were organized by waterbody (i.e., Estuaries and Coastal Waters; Lakes and Reservoirs; Rivers and Streams; and Wetlands) and these groups discussed their specific waterbodies during breakout sessions held on the first two days of the workshop. Breakout groups reconvened at several points during the workshop to report to the plenary session on the results of their discussions.

Many tools and measurement endpoints were deemed useful by each of the waterbody breakout groups. Some waterbodies were better understood than others in this regard. For example, measurement parameters for lakes are well known and the interactions between these parameters and resulting eutrophication are generally understood. Because of this, the lakes and reservoirs breakout group, which was dealing with a well-developed historical database, was able to rate the various measurement parameters in terms of their usefulness; in some cases the group was able to assign quantitative thresholds for each parameter. For wetlands, on the other hand, the breakout group was limited by a very small historical database and therefore was restricted to identifying potential measurement parameters and assigning qualitative judgements to each of the parameters. The wetlands group was also handicapped by the high variability encountered in wetland types.

Some of the general recommendations arising from the workshop were:

- A set of national standards is not a realistic goal and it would be more appropriate to set nutrient standards on an ecoregional or watershed basis.

- Organizations, states, and societies should be involved in further discussions of nutrient assessment and implementation issues, as well as development of the national nutrient overenrichment assessment strategy.

- Cultural eutrophication should be recognized as a public health threat. The two examples that were mentioned were 1) nutrient overenrichment can stimulate harmful and toxic algal species that directly affect the safety of seafood products and 2) drinking water treated from eutrophic lakes and reservoirs can be contaminated with harmful byproducts of the disinfection process (e.g., trihalomethanes).

- Land use should be included as a separate early warning indicator.

- EPA should provide simple, user-friendly, desktop-based software models, where available, to state and local governments to aid them in waterbody decision making.

- While many models can be recommended for specific waterbody types and sites (e.g., lakes), more research is needed on investigating models for rivers and streams (especially periphyton dominated systems), estuaries, and wetlands.

- The use of "reference sites" to develop baseline data for comparisons to potentially impacted areas is recommended, especially for wetlands where information is scarce.

Some of the waterbody specific conclusions/recommendations arising from the workshop were:

Lakes and Reservoirs

Conclusions/Recommendations

- National nutrient criteria must be narrative based and quantitatively implemented on a local/regional basis. This is because public opinion of (and in fact) good water quality varies by geographic region.

- There is a need to develop standardized protocols for several of the monitoring techniques currently used in the field (e.g., total phosphorus, total chlorophyll concentration, Secchi depth). These and other secondary indicators should be described in a technical guidance manual for use by all states.

- Land use change around lakes and reservoirs is one of the most telling early warning indicators of water quality trends. It was said, "A lake is a product of its watershed."

- Develop desk-top software for States to use for lakes and reservoirs individually to help assess the data they gather.

- Use professional societies such as the North American Lake Management Society, as well as EPA Regional meetings with the States and Tribes, to promote nutrient criteria and management for lakes and reservoirs

Endpoints

- Total phosphorus, total nitrogen, total chlorophyll, Secchi depth and dissolved oxygen are good endpoints to consider as indicators of trophic status of lakes and reservoirs.

- If we use chlorophyll as a parameter, total chlorophyll is arguably the best way to measure (as opposed to chlorophyll *a* or *b*).

- Secchi disk is a good, simple tool to use on lakes and reservoirs, recognizing that its use is limited where water visibility is blocked by suspended inorganic sediment or the water has obscuring color such as is naturally found in bog lakes.

- The best indicators are (in chronological order of response):

Early warning indicators:	Land use changes
Sensitive in-lake indicators:	Total phosphorus, total chlorphyll, Secchi depth, total nitrogen
Lake status indicators:	Total organic carbon, dissolved oxygen, total suspended solids, turbidity, biological (fish, algae, zooplankton, macrophytes)

Modeling

- Simple and effective "desktop" models currently exist at the State level that incorporate the following parameters: dissolved oxygen, total phosphorus, Secchi depth, total chlorophyll, and total nitrogen.

- A trend promoted by some modelers is to use carbon as a replacement parameter for chlorophyll.

Information Needs/Next steps

- We need to better define the relationship between management policies and loading.

- We need to better understand the relationship between nutrient loading and the macrophyte response.

- We need additional simple, accurate desktop models that distinguish between lakes and reservoirs, as well as geographic and temperate differences.

- We need a better understanding of the effect that suspended solids have on nutrient release.

Rivers and Streams

Conclusions/Recommendations

- National nutrient criteria, whether numeric or narrative, must be ecoregion specific, because of natural variabilities across waterbody types.

- National guidance should encourage States to adopt a nutrient control strategy, which includes the following minimum set of endpoints: total nitrogen, total phosphorous, Secchi depth, dissolved oxygen, and soluble reactive phosphorous.

- States should include land use as a separate early warning indicator (i.e., if development is proposed in a watershed, an environmental impact study should be done to assess the potential impact of such development on the surrounding waterbody).

- Minimizing the turbulence by constructing channels in the waterbody would help in reducing the rapid nutrient movement from one segment of the waterbody to another.

- Shading the streams or canopy restoration can minimize eutrophication.

- Biological control - Managers should consider introducing biomass eating organisms (e.g., cadis fly larva *(Dicosmoecus gilvipes)*), which efficiently remove both periphytic diatoms and filamentous algae from rock substrata.

Endpoints

Plankton dominated systems

- Algal Biomass
- pH
- Dissolved Oxygen (DO)
- Chlorophyll
- Biointegrity (macroinvertebrate index)
- Total Nitrogen vs. Total Phosphorous
- Transparency (Secchi disk)
- Temperature
- Total suspended solid, Variable suspended solid ratio

Periphyton dominated systems

- Algal Biomass
- pH
- Dissolved Oxygen (DO)
- Chlorophyll
- Biointegrity (macroinvertebrate index)
- Dissolved inorganic nitrogen (DIN) vs. Soluble reactive phosphorous
- Transparency (Black disk)
- Temperature
- Total suspended solid, Variable suspended solid ratio

Modeling

- Desktop models that are easily transferred across waterbodies and use the following parameters should be encouraged: total phosphorus, total nitrogen, total chlorophyll, dissolved oxygen, temperature and light attenuation instrumentations (i.e., Secchi disk and black disk).

- Adding temperature simulation to WASP5 is recommended. WASP5 is widely used in both water quality assessment and toxic modeling. The model considers comprehensive dissolved oxygen and algal processes, but does not include the carbon and silica cycles or full sediment diagenesis. In addition, the model's use is limited because it does not account for temperature.

- Periphyton should be added as a parameter incorporated within QUAL2E and HSPF. QUAL2E and HSPF are one-dimensional models that capture the longitudinal transport (steady-state flow) which dominate in most rivers and streams. QUAL2E and HSPF both consider advection and dispersion. Adding periphyton to these models will allow for simulation of periphyton biomass in the riverine system.

- Introduce load/response relationship (plankton dominated system) and concentration/ response (periphyton dominated system), in order to pinpoint with high degree of accuracy where loading reduction can be targeted.

- Other models of note are those used in combination with other techniques in the state of Montana to set nutrient loading targets for Clark Fork River.

Information Needs/Next steps

- Use of community metrics as early warning indicator.

- EPA should conduct a comprehensive literature search to determine what work has already been done on nutrient-related issues in rivers and streams.

- Need to synthesize information on the relative sensitivity of different riverine types to nutrient enrichment and use this information to investigate regressional relationships.

- EPA should publish a national technical guidance document summarizing the Agency's position on methodology information in assessing and controlling nutrients.

- EPA should set up a nationwide database for storing state of the science information on nutrient overenrichment endpoints and modeling techniques to serve as a resource center for water quality managers at state and local levels.

- Involve organizations such as American Society of Limnologist and Oceanographers (ASLO), American Water Resources Association (AWRA), North American Benthological Society (NABS) etc., in the development and implementation of the national nutrient strategy.

- Investigate pH and DO amplitude.

- Investigate the role of fluvial geomorphology as a factor in controlling algal growth.

- Field research should be conducted on *Cladophora*, and diatom growth requirements.

Estuarine and Coastal Waters

Conclusions and Recommendations

- Coral Reefs - EPA should support research on the relationship between specific nutrient concentrations and algal coverage and coral reef die-off.

- Seagrass Dominated Systems - Tools for assessing nutrient enrichment impacts on seagrass population exist and have been field tested. EPA should help in the transfer of these tools and insights for assessing nutrient enrichment impacts to waterbody specific seagrass habitats in estuarine and coastal management programs.

- Plankton Dominated Systems - Quantitative relationships have been established between nutrient loading and chlorophyll *a* (primary productivity) and nutrient loading and dissolved oxygen. EPA should assist research into the use of Vollenweider type methods in developing nutrient response relationships for plankton dominated systems.

- Nuisance Algal Blooms/Brown Tide Blooms - The relationship between nutrient loadings and nuisance algal blooms (e.g., brown tide) is not well established outside of laboratory settings. EPA should support a comprehensive synthesis of correlative/experimental evidence for toxic algal blooms response to nutrient enrichment.

- Macroalgae Dominated Systems - Only a qualitative relationship exists for relating nutrient loadings to macroalgal cover. EPA should build on the existing information on the relationship between nutrient loading and macroalgal cover.

Information Needs/Next Steps

- Coral reefs - Need to further study the relationship between nutrient concentrations and algal coverage and coral reef die-off.

- Seagrass - Need to confirm that tools for assessing nutrient enrichment impacts from Chesapeake Bay and Tampa Bay are appropriate for use across other waterbodies (i.e., identify any limitations of tools).

- Plankton - Need to develop nutrient/response relationships for plankton dominated systems using Vollenweider type relationships (i.e., phosphorus loading to hypolimnetic dissolved oxygen concentration regressions)

- Nuisance algae - Need to establish the relationship between nutrient loadings and nuisance algal blooms such as brown tide in field settings (i.e., ground truth this research).

- Macroalgae - Need to compile nutrient loading data for: seagrass, macroalgae, and phytoplankton. In addition, confirm the different community responses and pull out thresholds relating to these community shifts. Also, quantify the macroalgae influence on dissolved oxygen, dissolved organic carbon, and lower trophic levels.

Wetlands

Conclusions/Recommendations

- There needs to be an accepted national wetlands classification system similar to the hydrogeomorphic classification system developed by the Army Corps of Engineers.

- A nutrient database including baseline data according to wetlands types/regions needs to be developed.

- Field experimentation should be conducted to determine nutrient limitation to wetland type and to isolate the effects of nutrients from other variables, such as hydrology, climate, and natural variability.

- Need to integrate parameter data collection with wetland indicators, or possibly other EPA wetlands programs.

- Need to improve upon molecular biology techniques (e.g., stress proteins in plants).

- Currently, standards for wetlands are primarily narrative since it has been difficult to apply certain numeric criteria to wetlands (e.g., dissolved oxygen is hard to standardize because wetlands can be dry at times).

- Variability is important to consider in wetlands, especially in extreme flow and rainfall events.

Endpoints

- A "Secchi disk" for wetlands could be feldspar (silicon dioxide) which shows accumulation within a wetland over time. The rate of accumulation can be related to changes in productivity.

- Most important indicators (short- and long- term) of nutrient impact are: bioavailable nitrogen and phosphorous in sediments, soils, water, and vegetation; hydrology; and loads. However, for most of these parameters there is no baseline data available with which to compare collected data.

- For immediate assessment, recommended endpoints are: bioavailable nitrogen and phosphorous in soils and water; plant species composition; plant species richness; plant species structure; plant indicator species (vascular and non-vascular); soil oxygen demand.

Modeling

- Very few models exist for wetlands. There is a model under development for the Everglades which considers water budget, flow fields, vegetation and biology. It has a spatial scale which is important in assessing wetlands.

- Other models of note are those done for the Des Plaines wetlands and Odum Creek (Florida). However, these lack spatial scale.

- Landscape models can be used but are more coarse in their ability to predict impacts. Grids can be very large (1 km by 1 km).

- Some members felt that numeric models may not be useful in predicting changes in wetlands. Rather, more simple (i.e., heuristic) models may be more helpful in understanding nutrient-related responses within wetlands.

Information Needs/Next steps

- EPA should develop a nationwide database for natural wetlands similar to that currently available for constructed wetlands. The database should include a wetland type and statistical characteristics that apply to each wetland type. A national database could be used to compare the measurement parameters of assessed (impacted) wetlands to an established set of reference conditions.

- EPA should conduct a comprehensive literature search to determine what work has already been done on nutrient-related issues in wetlands.

- Organizations such as the State Association of Wetland Managers, Society of wetland scientists, and private organizations (e.g., Ducks Unlimited) should be involved in the development and implementation of the national nutrient strategy.

- Need to synthesize information on the relative sensitivity of different wetland types to nutrient enrichment and use this information to investigate regressional relationships.

Day One Plenary Session
Summary

I. Welcome and Participant Introduction

The National Nutrient Assessment Workshop opened with Rich Batiuk, U.S. EPA Chesapeake Bay Program Office, who welcomed all of the participants and thanked them for attending. Participants then introduced themselves and described their area of expertise as it related to the workshop objectives. Rich noted that the workshop had succeeded in convening representatives from a diversity of organizations, interests, and geographic areas to focus on a set of issues of regional and national importance.

Rich then presented a brief synopsis of events leading up to and the purpose behind the workshop. He informed the group that the workshop grew out of a recommendation of the EPA Office of Science and Technology Nutrient Task Force draft report released December 30, 1994 and stressed that EPA and its partners are attempting to re-engage nutrient overenrichment issues and to identify and address the relevant technical and implementation issues. He stressed the importance of identifying endpoints and selecting assessment tools for use at the individual waterbody and watershed scales. He pointed out that the workshop should serve as a unique opportunity to pool the experience that is available in the scientific and resource management communities in helping set the short and long-term priorities for a national nutrient overenrichment assessment strategy.

II. Introductory Remarks

Tudor Davies, Director, U.S. EPA Office of Science and Technology

Tudor Davies emphasized that eutrophication continues to be a problem faced by many states and municipalities. He challenged the workshop participants to ask whether or not there are alternative ways to assess the nature and extent of nutrient enrichment problems and he reminded them that their discussions should take into account implementation, as well as technical and scientific issues. Tudor concluded his remarks by assuring the participants that EPA is taking this issue very seriously and will use the results of the workshop as it prepares its national strategy.

Bob Wayland, Director, U.S. EPA Office of Wetlands, Oceans, and Watersheds

Bob Wayland spoke about confronting the nutrient overenrichment issue from a management perspective. He mentioned the growing public awareness of eutrophication problems through programs such as the Clean Lakes Program, the Chesapeake Bay Program, and the National Estuaries Program. He stressed that a national nutrient strategy needs to have clear management objectives and recognize concerns of dischargers, especially nonpoint sources, about affordability. Bob expressed his hope that the workshop could serve to facilitate technology transfer and communication among parties from all of the different communities represented.

III. Past, Present, and Future Perspectives on the Assessment and Management of Nutrient Overenrichment Needs[1]

Robert Thomann, Manhattan College

Robert Thomann presented his perspective on the historical issues associated with the assessment and management of nutrient overenrichment. He divided the past seventy years into two periods: the Observational Descriptive period that began in the late 1920s and the Predictive Management period that began circa 1965. He characterized the Observational Descriptive period as a time when general observations were being made about nutrient enrichment issues, but little was understood concerning the cause and effect relationships between loadings and eutrophication.

The Predictive Management period, on the other hand, was characterized as a time when researchers attempted to relate the basic science that was available to practical management goals. The period from 1970 to 1980 saw a rapid expansion in the spatial detail and number of variables included in nutrient models. This progress slowed somewhat during the period from 1980 to 1990 (characterized as the "Dry Years") as many persons came to feel that the problem of eutrophication had been solved—the National Eutrophication Program came to an end and attention was diverted to chemical contaminants and acid rain. Interest began to grow again during the late 1980s, however, as the public began to realize that nutrient overenrichment could have far-reaching effects. By this time dynamic modeling had become much more advanced, with more variables being included in each model and key inputs being internalized into the model framework. This development was facilitated by the evolution in computer technology and a reduction in computer costs.

Bob concluded his remarks by outlining the requirements of a successful nutrient management program. He pointed out the need for a credible framework to support quantitative decision making that takes into account what level of improvement to expect, a time frame for this improvement, and a cost/benefit analysis, if possible. He also stressed that this framework should incorporate the perspectives of nutrient dischargers, the scientific community, and the general public.

Mimi Dannel, U.S. EPA Office of Wetlands, Oceans, and Watersheds

Mimi Dannel explained where EPA Headquarters and its regional offices are today in terms of addressing the eutrophication issue. EPA is aware that there is a need for management of excessive nutrient loadings, especially as evidenced by national and State findings summarized in Clean Water Act section 305(b) reports (National Water Quality Inventory) and section 303(d) lists (identifying water quality limited waters). EPA currently encourages consideration of nutrient enrichment issues in State's implementing watershed protection approaches, community-based environmental protection approaches, and in potential

[1]Summaries of the overheads used during these presentations are attached at the end of the proceedings.

opportunities for pollutant trading. Mimi told the participants that she hoped they develop a list of endpoints and assessment methodologies for each of the waterbodies, as well as a core set of models to be used for "routine" application across the different waterbody categories. She indicated that the focus should be on models that already exist and how they could be enhanced and simplified. She also indicated that participants need to critically look at the number of input parameters and how they are measured. Finally, she explained that there is a need to be able to demonstrate the costs of model application versus the cost of controls to avoid focusing on the tool more than the resource.

Steve Heiskary, Minnesota Pollution Control Agency

Steve Heiskary presented workshop participants with the nutrient control approach currently used in Minnesota as an example of how a state could address nutrient issues on a regional basis. He explained that most of the lakes in Minnesota are located in four main ecoregions and that phosphorus criteria have been established in each ecoregion based on user expectations and regional variations in attainable lake trophic status. This approach has been aided by the selection of "reference" (minimally impacted) lakes within each ecoregion that were chosen based on expert recommendations and input from citizen monitoring groups. Steve also explained how Minnesota implemented the lake user survey approach (developed in Vermont) to gauge the public's expectations of lake quality for each of the different ecoregions.

Steve mentioned that several other states have been doing progressive work in this area (e.g., Vermont with the user surveys, Oregon with a nuisance phytoplankton approach, and North Carolina with a nutrient sensitive approach). He ended his presentation by explaining that Minnesota uses its criteria to prioritize and target nutrient reduction projects, to develop water quality management plans, to serve as an educational tool, and to guide enforcement decisions.

IV. Purpose and Goals of the Workshop

After the introductory presentations, Rich Batiuk presented the workshop participants with the purpose and goals of the workshop and issued specific challenges to each of the breakout groups. Emphasis was placed on coming up with a practical strategy that encompasses local nutrient overenrichment assessment and implementation issues. The participants then separated into their assigned groups for the day one breakout sessions.

Breakout Group Discussion Questions

Lakes, and Reservoirs

Nutrient Enrichment Assessment Endpoints

- What are the currently available, ready to use, and most promising nutrient enrichment endpoints for application to lakes, impoundments, and ponds?
- Are there any early warning indicators of onset of nutrient enrichment conditions?
- How readily applicable are these endpoints at the regional, state, and local watershed and waterbody assessment scales? What are their advantages/disadvantages? Can they be applied to a wide range of waterbodies or are they geographically limited in their application?
- What is the balance that should be achieved between endpoints with a loading/response condition focus vs. endpoints with more of a user perception-based focus? Are they mutually exclusive or can they be married together in some form?
- Are there different sets of endpoints that apply only to large lakes vs. impoundments vs. small lakes and ponds? What are they?
- What gaps exist in this current array of endpoints that are necessary for regional/state/local watershed/waterbody-based assessment of nutrient enrichment status and, ultimately, management? What is the relative priority that should be given to the development/calibration/validation of new endpoints?
- Are some of the endpoints (or the science behind them) applicable to other non-lake/impoundment/pond waterbodies?

Dissolved Oxygen

- Are the existing EPA freshwater criteria sufficiently protective enough and/or comprehensive enough for full application to the entire range of lake, reservoir, and pond ecosystems?

Light Penetration/Nutrient Criteria

- Does the current state of the science support derivation of light attention criteria for lakes, impoundments, and ponds? What about nutrient, suspended solids, and chlorophyll *a* water quality criteria?
- Should EPA be actively promoting regional-based development and adoption of such criteria for the protection of lake and reservoir habitats?

Models

- What are the most appropriate models that should be highlighted within the national strategy? What are their relative advantages and disadvantages in terms of direct application, data calibration requirements, and amount of training/expertise necessary to allow effective use by regional, state, and local management agencies?
- How well do the identified models perform in relation to the preliminary list of endpoints that are concurrently being discussed at the workshop?

- Is there a good match between the available endpoints and the available load-response models?
- Which models and assessment tools are most likely to be used by local water quality managers in support of reaching the identified endpoints for each waterbody type?
- How can existing models be enhanced or what types of new models need to be developed to improve modelling capability and broaden use of models in assessment of nutrient enrichment?
- Are the existing "state of the science" lake and reservoir models currently being developed for and applied in large lakes and reservoirs at a point whereby simplified versions of the models can be extracted and used by smaller, less wealthy lake restoration and protection programs with the same basic management oriented results? If not, what steps need to be taken to get to that point?
- What about the same questions directed towards watershed models that feed inputs into lake/reservoir water quality models?

Issue Paper

- Are there obvious gaps in the lakes and reservoirs issue paper's coverage of the major literature that should be filled prior to publication as an EPA report and in the peer reviewed literature?
- Are there obvious gaps in the modeling issue paper's coverage of the major literature addressing lakes and reservoirs that should be filled prior to publication as an EPA report and in the peer reviewed literature?

Rivers and Streams

Nutrient Enrichment Assessment Endpoints

- What are the currently available, ready to use, and most promising nutrient enrichment endpoints for application to rivers and streams?
- Are there any early warning indicators of onset of nutrient enrichment conditions?
- How readily applicable are these endpoints at the regional, state, and local watershed and waterbody assessment scales? What are their advantages/disadvantages? Can they be applied to a wide range of waterbodies or are they geographically limited in their application?
- Are there different set of endpoints that apply only to large rivers vs. small streams? fast flowing vs. slow flowing rivers?
- What gaps exist in this current array of endpoints that are necessary for regional/state/local watershed/waterbody based assessment of nutrient enrichment status and, ultimately, management? What is the relative priority that should be given to the development/calibration/validation of new endpoints?
- Are some of the endpoints (or the science behind them) applicable to other non-riverine waterbodies?

Dissolved Oxygen

- Are the existing EPA freshwater criteria sufficiently protective enough and/or comprehensive enough for full application to the entire range of river and stream ecosystems?

Light Penetration/Nutrient Criteria

- Does the current state of the science support derivation of light attention criteria for rivers and stream? What about nutrient, suspended solids, and chlorophyll *a* water quality criteria? Are these really appropriate for these riverine systems (with the possible exception of slow flowing rivers and streams with longer retention times)?
- Should EPA be actively promoting regional-based development and adoption of such criteria?

Models

- What are the most appropriate models that should be highlighted within the national strategy? What are their relative advantages and disadvantages in terms of direct application, data calibration requirements, and amount of training/expertise necessary to allow effective use by regional, state, and local management agencies?
- How well do the identified models perform in relation to the preliminary list of endpoints that are concurrently being discussed at the workshop?
- Which models and assessment tools are most likely to be used by local water quality managers in support of reaching the identified endpoints for each waterbody type?
- How can existing models be enhanced or what types of new models need to be developed to improve modelling capability and broaden use of models in assessment of nutrient enrichment?
- Are the existing "state of the science" models currently being developed for and applied in rivers and streams at a point whereby simplified versions of the models can be extracted and used by smaller, less wealthy river/stream restoration and protection programs with the same basic management oriented results? If not, what steps need to be taken to get to that point?

Issue Paper

- Are there obvious gaps in the rivers and streams issue paper's coverage of the major literature that should be filled prior to publication as an EPA report and in the peer reviewed literature?
- Are there obvious gaps in the modeling issue paper's coverage of the major literature addressing rivers and streams that should be filled prior to publication as an EPA report and in the peer reviewed literature?

Wetlands

Nutrient Enrichment Assessment Endpoints

- Are there any available endpoints useful for assessing the relative impact of nutrient enrichment on wetland ecosystems?
- Are there any early warning indicators for nutrient enrichment conditions within wetlands?

Models

- Are there any models available for simulating/assessing the relative impact of nutrient enrichment on wetland ecosystems?
- What are the most appropriate wetlands models that should be highlighted within the national strategy? What are their relative advantages and disadvantages in terms of direct application, data calibration requirements, and amount of training/expertise necessary to allow effective use by regional, state, and local management agencies?
- How well do the identified models perform in relation to the preliminary list of endpoints that are concurrently being discussed at the workshop?
- Which models and assessment tools are most likely to be used by local water quality managers in support of reaching the identified endpoints for each waterbody type?
- How can existing models be enhanced or what types of new models need to be developed to improve modeling capability and broaden use of models in assessment of nutrient enrichment?

Issue Paper

- Are there obvious gaps in the wetlands issue paper's coverage of the major literature that should be filled prior to publication as an EPA report and in the peer reviewed literature?
- Are there obvious gaps in the modeling issue paper's coverage of the major literature addressing wetlands that should be filled prior to publication as an EPA report and in the peer reviewed literature?

Estuaries and Coastal Waters

Nutrient Enrichment Assessment Endpoints

- What are the currently available, ready to use, and most promising nutrient enrichment endpoints for application to estuarine and coastal waterbodies?
- Are there any early warning indicators of onset of nutrient enrichment conditions?
- How readily applicable are these endpoints at the regional, state, and local watershed and waterbody assessment scales? What are their advantages/disadvantages? Can they be applied to a wide range of waterbodies or are they geographically limited in their application?
- What gaps exist in this current array of endpoints that are necessary for regional/state/local watershed/waterbody based assessment of nutrient enrichment status and, ultimately, management? What is the relative priority that should be given to the development/calibration/validation of new endpoints?
- Are some of the endpoints (or the science behind them) applicable to other non-estuarine/coastal waterbodies?

Dissolved Oxygen

- Building on the estuarine/marine dissolved oxygen criteria document under preparation for northeast (Cape Cod to Cape Hatteras) coast, should EPA support development of the low dissolved oxygen effects databases necessary to publish estuarine/marine dissolved oxygen criteria for the southeast Atlantic (Cape Fear to Key West), Gulf, and Pacific coasts?
- What about development of the data sets necessary to extend the northeast criteria along the coastline north of Cape Cod up into the Gulf of Maine?

Light Penetration/Nutrient Criteria

- Does the current state of the science support derivation of light attention criteria for protection of submerged aquatic vegetation (seagrasses) along the U.S. coastlines? What about nutrient, suspended solids, and chlorophyll *a* water quality criteria based on protecting submerged aquatic vegetation?
- Should EPA be actively promoting regional-based development and adoption of such criteria for the protection of shallow water habitats?

Models

- What are the most critical nutrient enrichment related issues that the existing models have yet to be designed, calibrated, and/or applied to address?
- What are the most appropriate models that should be highlighted within the national strategy? What are their relative advantages and disadvantages in terms of direct application, data calibration requirements, and amount of training/expertise necessary to allow effective use by regional, state, and local management agencies?
- How well do the identified models perform in relation to the preliminary list of endpoints that are concurrently being discussed at the workshop?
- Are the existing "state of the science" estuarine/coastal models currently being developed for and applied in Chesapeake Bay, Long Island Sound and other larger estuaries at a point whereby simplified versions of the models can be extracted and used by smaller, less wealthy estuarine and coastal protection programs with the same basic management oriented results?
- If not, what steps need to be taken to get to that point?
- Which models and assessment tools are most likely to be used by local water quality managers in support of reaching the identified endpoints for each waterbody type?
- How can existing models be enhanced or what types of new models need to be developed to improve modelling capability and broaden use of models in assessment of nutrient enrichment?
- What about the same questions directed towards watershed models that feed inputs into estuarine/coastal water quality models?

Issue Paper

- Are there obvious gaps in the estuarine/coastal issue paper's coverage of the major literature that should be filled prior to publication as an EPA report and in the peer reviewed literature?

- Are there obvious gaps in the modeling issue paper's coverage of the major literature which address estuarine and coastal systems that should be filled prior to publication as an EPA report and in the peer reviewed literature?

Watersheds

Nutrient Enrichment Assessment Endpoints
- What is/what should be the relationship between nutrient enrichment endpoints for watersheds themselves vs. endpoints addressing watersheds as contributors to downstream waterbodies? Is there an inherent conflict here?

Models
- What are the most appropriate watershed models that should be highlighted within the national strategy? What are their relative advantages and disadvantages in terms of direct application, data calibration requirements, and amount of training/expertise necessary to allow effective use by regional, state, and local management agencies?
- How well do the identified models perform in relation to the preliminary list of endpoints that are concurrently being discussed at the workshop?
- Which models and assessment tools are most likely to be used by local water quality managers in support of reaching the identified endpoints for each waterbody type?
- How can existing models be enhanced or what types of new models need to be developed to improve modelling capability and broaden use of models in assessment of nutrient enrichment?

Issue Papers
- Are there obvious gaps in the four issue papers' coverage of the major literature addressing watershed models that should be filled prior to publication as an EPA report and in the peer reviewed literature?

Day Two Plenary Session
Summary of Reports from Breakout Groups and Discussion

Workshop participants reconvened on the morning of the second day to report on the results of their breakout group discussions. The following is a summary of what each of the facilitators had to say concerning their respective day one breakout group sessions.

I. Lakes and Reservoirs Summary

The breakout group began its discussion by outlining what it hoped to accomplish during the course of the workshop. It was agreed that the group would be making recommendations to EPA on the utility of using various parameters to assess waterbody status and to use as early warning indicators[2] of lake and reservoir nutrient overenrichment. The breakout group also agreed that they would be providing general recommendations on issues that would need to be addressed in a national nutrient assessment strategy, including the capabilities and limitations of existing modeling tools for lakes and reservoirs.

To organize the discussion, the group divided an initial list of parameters into three categories. The group realized that these categories could be differentiated in a number of ways (e.g., proactive parameters vs. reactive parameters; thermodynamic parameters vs. species succession/community structure parameters; parameters that are well understood vs. less well-understood parameters.) Eventually the group decided to separate the parameters into the following subsets:

Chemical/Biomass Parameters
- Land Use/Loading
- Nitrogen Concentration
- Secchi Depth
- Phosphorus Concentration
- Chlorophyll
- Dissolved Oxygen

Community Structure Parameters
- Algal Community
- Macrophytes
- Zooplankton
- Macroinvertebrate Structure
- Fish

Secondary Parameters
- Total Suspended Solids
- Total Organic Carbon

The group began to evaluate each of these parameters based on the following evaluation criteria:

How Measured What quantitative units are used to measure the parameter?

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[2] An early warning indicator was defined as "a change from a baseline condition within a region and classification of the water resource and watershed which is sufficiently rapid to initiate increased monitoring or management before degradation (unacceptable change) takes place."

Value	What advantages does using the parameter provide to the resource manager relative to using other parameters?
Scientific Validity	Are there well-accepted (i.e., within the scientific community), reproducible measurement techniques available for measuring the parameter?
Practicality	Is this a parameter that the states can realistically use given time, personnel, and resource constraints?
Cost	What is the cost of collecting and using this parameter?
Public Understanding	Is this a parameter that the public understands and will support?
Modeling Capability/ Load-Response	Is the relationship between loading and subsequent response well-understood? Can the parameter be modeled given existing tools or easy-to-develop tools?

In general, the group was able to reach consensus on a rating[3] for each of the evaluation criteria for each of the parameters. The results of their discussion are summarized in the tables below. Blank spaces in the "Rating" column indicate that either no consensus was reached or no formal designation was specified.

Land Use/Loadings	Rating	Notes
How Measured		Loadings are measured based on the existing understanding between certain types of land use and related loadings.
Value	High	Aggregated loadings allow you to get to one number for all of the sources within a watershed
Scientific Validity	High	
Cost	Variable	Some states have adequate land use data while others are still in the process of gathering it.
Practicality	High	
Public Understanding	Medium to High	Public does not intuitively understand the relevance of loading, but is capable of learning (as evidenced by 40% reduced loading target for Chesapeake Bay).
Modeling Capability/ Load-Response	High	

[3]A high rating typically meant that the group felt the parameter performed well in terms of the evaluation criteria (e.g., a high rating for Scientific Validity means the parameter is well-accepted within the scientific community whereas a low rating means that it is not). The one exception was the cost criteria, in which case a low rating is preferable to a high rating.

Phosphorus	Rating	Notes
Units of Measurement (Total P)		Total phosphorus should be used because it is a more comprehensive measurement.
Value	High	Phosphorus is the start of all productivity. Total phosphorus also has the potential for being a "one shot" measurement because other variables can usually be related to it.
Scientific Validity	Medium to High	Scientific validity is high for lakes (especially for certain types of lakes); not as well established for reservoirs.
Cost	Low	Already measured as a component of most monitoring programs. Springtime phosphorus can be especially cheap because it only needs to be collected once.
Practicality	High	
Public Understanding	Medium to High	The meaning behind phosphorus concentrations has to be translated for the public, but interested citizens are aware of their significance.
Modeling Capability/ Load-Response		Modeling capability is especially good for temperate and glacial lakes.

Nitrogen	Rating	Notes
Units of Measurement (Total N)		
Value	Medium	Nitrogen used to get N:P ratio; some systems are nitrogen limited.
Scientific Validity		Nitrogen interactions have not been as extensively studied as phosphorus because there is less motivation to do so for lakes and reservoirs.
Cost	Low	Cost of sampling is similar to that for phosphorus.
Practicality	Medium	Not as practical as phosphorus because fewer lakes and reservoirs are nitrogen limited. EPA needs to promote a standard methodology for sampling.
Public Understanding	Low	Public is less aware of significance of nitrogen (as compared to phosphorus); they tend to be lumped together as "nutrients".
Modeling Capability/ Load-Response		More sophisticated models for predicting impact of nitrogen concentrations need to be developed; load-response relationship not worked out as well as for phosphorus because of the number of variables in the nitrogen cycle (e.g., nitrogen fixation).

Chlorophyll	Rating	Notes
Units of measurement (mg/l)		The group differed on the utility of using total chlorophyll vs. chlorophyll *a*. One problem with chlorophyll *a* is that measurements taken prior to 1982 can't be compared to later values because the measurement method has changed. The group suggested that EPA make a definitive statement on a standardized methodology.
Value	High	Chlorophyll is the best biomass surrogate and is the first biomass response to phosphorus. Chlorophyll is also the most interpretable parameter, but has high temporal and spatial variability.
Scientific Validity	High	Chlorophyll is less predictable than phosphorus and is not as good for trend detection because of its high variability.
Cost	Low	Sampling cost similar to phosphorus, although collection costs are higher.
Practicality	Medium	Samples need to be taken many times per year and over the course of several years to detect trends. The group recommended the sampling regime spreadsheet distributed by the North American Lake Management Society as guidance.
Public Understanding	Medium	Measurement may be linked to subjective measures of user perceptions.
Modeling Capability/ Load-Response	Variable	Chlorophyll concentrations can be modeled, but need to have phosphorus concentrations to do so.

Secchi Depth	Rating	Notes
Units of Measurement (depth)		Seems like a simple process, but the group recommended that EPA establish a consistent methods and material protocol.
Value	High	Secchi depth measurements have low temporal variance (especially compared to chlorophyll), but high silt content can distort the use of the readings for nutrient impact (i.e., may be a non-nutrient enrichment related problem).
Scientific Validity	Variable	Depends upon what it is used for—low validity if used for chlorophyll surrogate, but high if used to estimate transparency and total phosphorus.
Cost	Low	
Practicality	High	
Public Understanding	High	Measurement may be linked to subjective measures of user perceptions.
Modeling Capability/ Load-Response		Not a straight line relationship because it is difficult to distinguish Secchi depth changes once in the low visibility range. There is also a problem with natural variability when trying to detect trends: only a few models break transparency down into total suspended solids, color, etc.

Dissolved Oxygen*	Rating	Notes
Units of Measurement (mg/l)		Measurement generally accepted as end of summer dissolved oxygen in hypolimnion; could also be areal hypolimnetic oxygen depletion (AHOD) or oxygen depletion rate or days of anoxia or when the lake went anoxic.
Value	Variable	Four problems: 1) there is considerable spatial and temporal variability associated with dissolved oxygen readings; 2) southeastern reservoirs display rapid changes between anoxia and oxia during withdrawal cycles; 3) some investigators now want to measure CO_2 accumulation rate as a negative oxygen value (i.e., a reducing environment); 4) affected by non-nutrient-related factors such as temperature and morphology.
Scientific Validity		Has cross lake validity and provides information on aquatic life use support.
Cost	Low	Fairly cheap once equipment is available; have to measure at least four times per year.
Practicality	High	
Public Understanding	High	There is a high degree of public understanding because low dissolved oxygen levels are related to fish kills.
Modeling Capability/ Load-Response	High	Can be modeled at least as well as the other parameters.

*Discussion restricted to lakes that stratify.

After discussing the chemical/biomass parameters, the group talked about whether or not total suspended solids or total organic carbon should be included as potential nutrient assessment endpoints. They eventually agreed that both parameters provide additional information beyond that provided by the previously mentioned parameters and decided to add them as "secondary parameters". The point was made that carbon might prove helpful because it is related to a host of other waterbody problems that affect and are affected by eutrophication.

It was also during this discussion that the importance of emphasizing eutrophication as a public health concern first arose. In particular, the participants noted the hazards associated with consuming drinking water containing potentially carcinogenic disinfection by-products, such as trihalomethanes. (Trihalomethanes can be produced as a by-product of the disinfection process used in plants that take their water from eutrophic lakes or reservoirs.) The group agreed that this public health issue needs to be emphasized by EPA and others to gain public support for a national nutrient strategy.

Several questions were posed to the lakes breakout group by members of the plenary session. These included whether or not the group had addressed the cross-cutting issue of assessing and managing large lakes vs. small lakes and whether or not the group had evaluated how well the different species of phosphorus would serve as endpoints (as opposed to total phosphorus). The lakes group stated that these were two issues they intended to address in the next breakout session.

II. Rivers and Streams Summary

Introductions/Agenda

The rivers and streams breakout group began with participant introductions and a discussion of the agenda. The participants briefly discussed the morning plenary session and agreed that a reiteration of the breakout group's goals and objectives would not be necessary. Some of the participants suggested using the issue paper as a springboard for the discussion. While this suggestion was not adopted, participants agreed that during the course of the workgroup session attention should be given to identifying and filling in the information gaps in the issue paper.

The main issue addressed in the discussion of the agenda was whether or not the group should pursue a process versus an endpoint specific approach to addressing the problem of nutrient loading. Participants emphasized the importance of recognizing local and regional differences among rivers and streams. There was an acknowledgement that it would be very difficult to develop specific endpoints that could be applied to all river and stream systems. As a result, the group agreed to focus on developing a better understanding of the nutrient loading process in an effort to provide recommendations on appropriate indicators.

Moving from their agreement to emphasize process, the group began to discuss some of the basic biological factors associated with the nutrient loading. There was a consensus that understanding nutrient loading impacts on streams and rivers was different from understanding nutrient loading impacts on lakes. The relationship between algae growth and nutrient loading is relatively linear in lakes while in streams the relationship is more complex. In a more dynamic stream or river ecosystem, low levels of soluble nutrients can produce both small and large volumes of biomass.

Factors Contributing To Nutrient Growth

Participants agreed that given this complexity, more information was needed to improve our understanding of the various relationships and factors associated with algae growth in streams and rivers. As a result, the group felt it would be an important first step to list some of the known factors other than nutrient loading that contribute to algae growth. Some of the factors mentioned by the participants included:

- Extended low flow periods
- Grazers, fish
- Stream morphology
- Scouring flow threshold - "tumbling rocks" disrupts colonization (in periphyton dominated systems).

Emerging from this discussion was an awareness that nutrients may not always be the main contributor to algal growth. In fact, one participant noted that this might indicate a misfocusing of traditional management efforts by primarily regulating nutrient loading and not adequately addressing the other contributing factors.

As a result of this discussion, participants reconfirmed their commitment to a process oriented approach. The group agreed that it would be useful to categorize the different types of streams and rivers. Participants felt that this would facilitate the identification of some basic endpoints as well as a better understanding of the nutrient loading process.

River and Stream Categorization and Endpoint Identification

To effectively categorize rivers and streams and identify endpoints, the group began by listing some of the major factors that should be considered in a stream/river categorization. Participants identified the following factors:

- Regime/climate
- Substratum (eroding vs. deposition
- Land use
- Canopy cover/riparian vegetation
- Volume of wetted channel/bankfill
- Stream order/drainage area
- Regulated vs. unregulated
- Local geology
- Flow
- Morphology

From this brief discussion of factors, the group initially reached the conclusion that a basic way to categorize the different types of streams and rivers was to distinguish between *low gradient* and *high gradient* streams. The participants, however, soon discarded this distinction and agreed that a more appropriate characterization would be between *plankton dominated* and *periphyton dominated* streams.

Following the basic categorization, the group was then able to move on to identifying endpoints. Essentially, for both plankton dominated and periphyton dominated streams the participants identified factors that they felt were meaningful in terms of managing and understanding these systems. Participants did not focus their discussion on specific numbers or measurements for the endpoints. Table on the following page lists the basic endpoints that were identified for each system. Potential endpoints that were specifically not recommended for both systems included nitrogen:phosphorus ratios and the algal growth test.

Table 1. Potential Endpoints for Plankton and Periphyton Dominated Systems.

Plankton Dominated	*Periphyton Dominated*
Algal biomass pH (maximum and diel) Dissolved Oxygen (minimum and diel) Transparency (Secchi depth) Biointegrity (macroinvertebrate index, community composition) Total suspended solids, volatile suspended solids, ratios Dissolved organic material Autotrophic Index (AFDW/CNa) Total nitrogen Total phosphorus Ratios of summer/winter nutrient concentrations Ratios of dissolved/total nutrient concentrations Aesthetics (foam, scum)	Algal biomass (*mg/m², percent coverage*) pH (maximum and diel) Dissolved Oxygen (minimum and diel) Transparency (*Black* disk) Biointegrity (macroinvertebrate index, community composition) Total suspended solids, volatile suspended solids, ratios Dissolved organic material Autotrophic Index (AFDW/chl *a*) Total nitrogen, dissolved inorganic nitrogen Total phosphorus, soluble reactive phosphorus Ratios of summer/winter nutrient concentrations Ratios of dissolved/total nutrient concentrations Aesthetics (foam, scum)

Process Diagrams

Following the identification of endpoints, the discussion moved to how the different endpoints and factors that influence algae growth interrelate and connect. Group members decided to construct flow diagrams to facilitate the understanding of the eutrophication process in both plankton and periphyton dominated systems. The figures on the next page are the process diagrams that were created by the participants. Both of the diagrams take a "nutrient centric" view of the process. The interrelationships among the other contributing factors to algae and periphyton growth were not discussed in detail.

Figure 1. Plankton Dominated Systems.

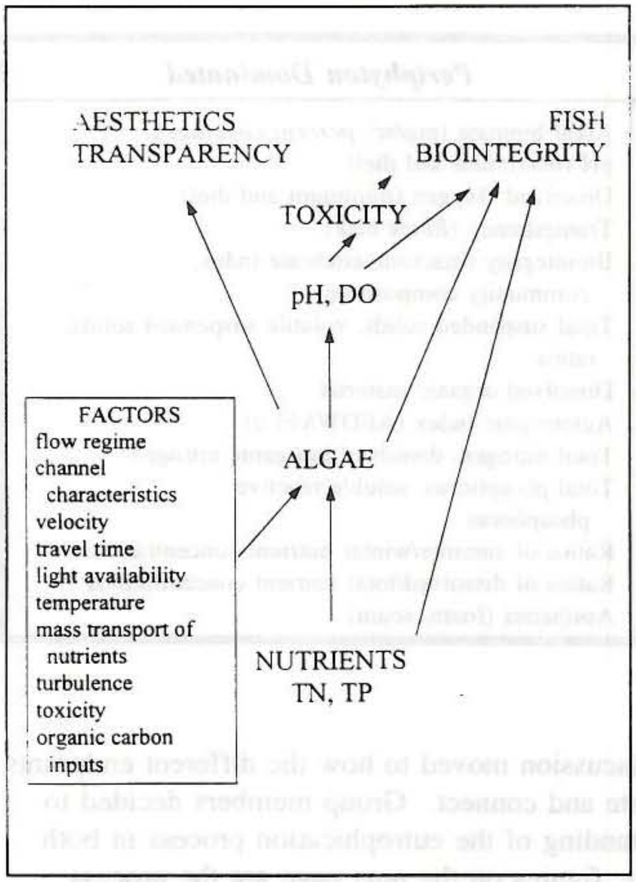
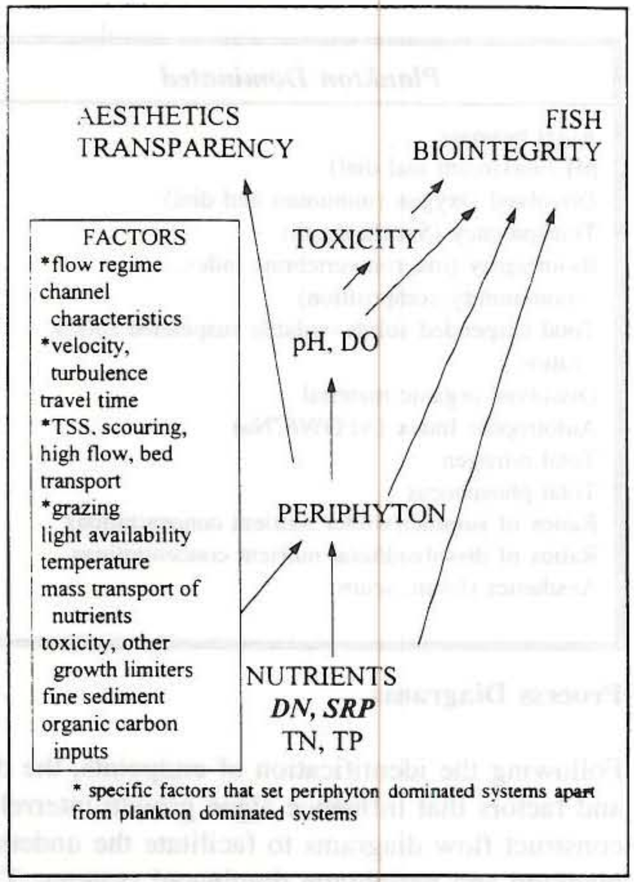


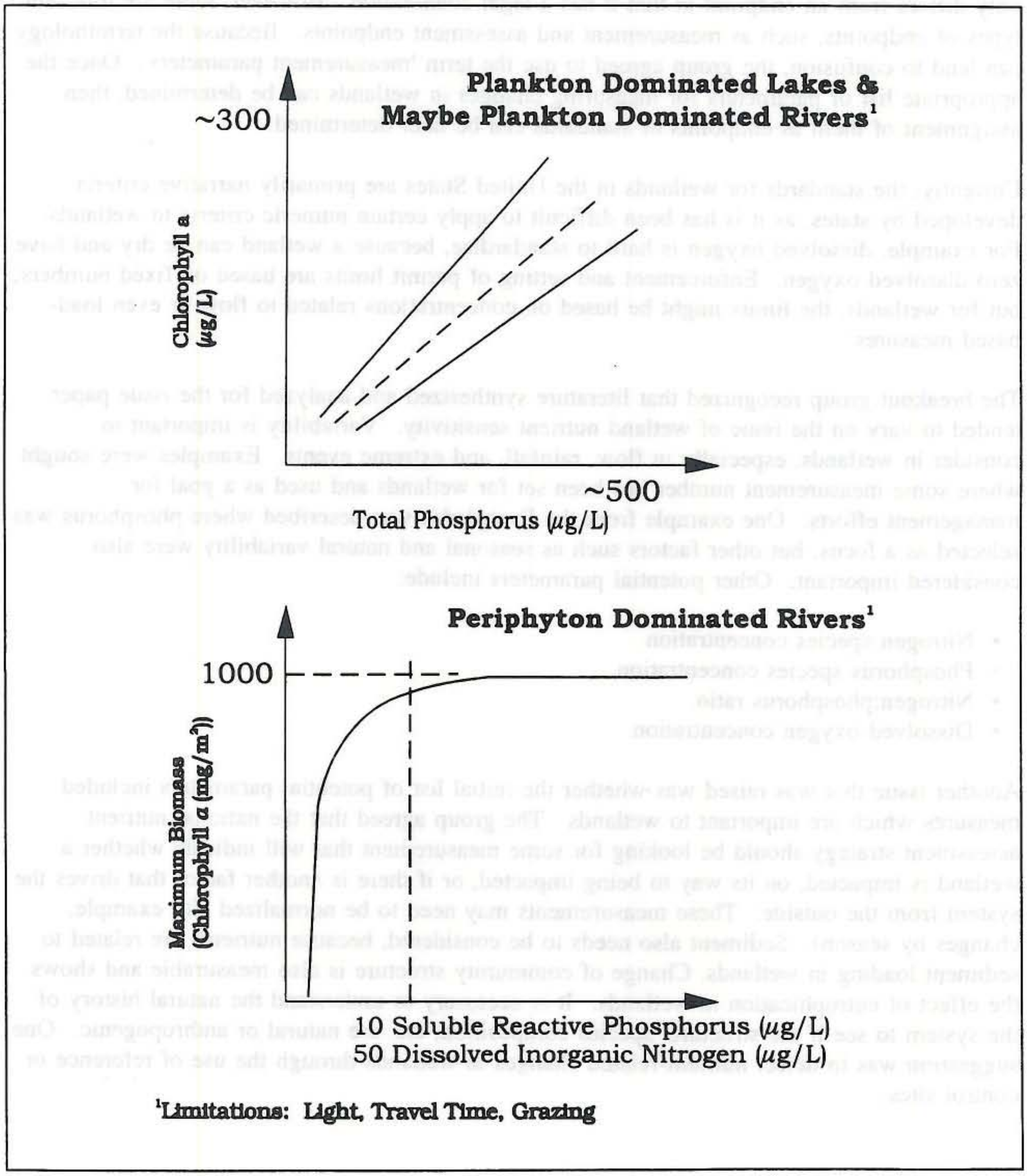
Figure 2. Periphyton Dominated Systems.



The day's discussion concluded with the group highlighting the key differences between the plankton dominated and periphyton dominated systems. Participants identified four specific factors that set periphyton dominated systems apart from plankton dominated systems: the flow regime, velocity, total suspended solids/scouring/high flow/bed transport, and grazing. The group emphasized that these factors all contribute to the one major characteristic that distinguishes these two systems. The major differentiating characteristic is that nutrients will saturate the biomass at a much lower level in the periphyton dominated system than they will in the plankton dominated system.

To help illustrate this difference, participants constructed the graphs in Figure 3. The graphs show the relationship between nutrient concentrations and biomass saturation in both plankton dominated and periphyton systems. Note that the graphs only serve to illustrate a concept. The graphs are not drawn to scale and they have not been constructed from actual data, but are based on the combined experience of the discussion group. The numbers begin to approximate the typically observed thresholds based on limited field studies.

Figure 3. Relationships between nutrient concentrations and biomass saturation in both plankton and periphyton dominated systems.



III. Wetlands

To begin the morning session, the wetlands breakout group discussed the definition and use of the term 'endpoint'. When a variable is chosen and a threshold determined, that threshold value or range can become an endpoint. Based on current operational definition, a "standard" only differs from an endpoint in that it has a legal connotation. However, there are different types of endpoints, such as measurement and assessment endpoints. Because the terminology can lend to confusion, the group agreed to use the term 'measurement parameters'. Once the appropriate list of parameters for measuring changes in wetlands can be determined, then assignment of them as endpoints or standards can be later determined.

Currently, the standards for wetlands in the United States are primarily narrative criteria developed by states, as it is has been difficult to apply certain numeric criteria to wetlands. For example, dissolved oxygen is hard to standardize, because a wetland can be dry and have zero dissolved oxygen. Enforcement and setting of permit limits are based on fixed numbers, but for wetlands, the limits might be based on concentrations related to flow or even load-based measures.

The breakout group recognized that literature synthesized and analyzed for the issue paper tended to vary on the issue of wetland nutrient sensitivity. Variability is important to consider in wetlands, especially in flow, rainfall, and extreme events. Examples were sought where some measurement number has been set for wetlands and used as a goal for management efforts. One example from the Everglades was described where phosphorus was selected as a focus, but other factors such as seasonal and natural variability were also considered important. Other potential parameters include:

- Nitrogen species concentration
- Phosphorus species concentration
- Nitrogen:phosphorus ratio
- Dissolved oxygen concentration

Another issue that was raised was whether the initial list of potential parameters included measures which are important to wetlands. The group agreed that the national nutrient assessment strategy should be looking for some measurement that will indicate whether a wetland is impacted, on its way to being impacted, or if there is another factor that drives the system from the outside. These measurements may need to be normalized (for example, changes by season). Sediment also needs to be considered, because nutrients are related to sediment loading in wetlands. Change of community structure is also measurable and shows the effect of eutrophication in wetlands. It is necessary to understand the natural history of the system to see if the structure, species composition, etc. are natural or anthropogenic. One suggestion was to detect nutrient-related changes in wetlands through the use of reference or control sites.

The group agreed to attempt to identify measurement parameters, and emphasized that there is a need to identify early warning signals, signals that may clue managers and scientists into what may occur 10 years later. The following tasks of the breakout group were identified:

1) Create a list of parameters which measure and improve our understanding of nutrient-related impacts to wetlands.
2) Rank the parameters in importance or organize them for wetlands in a national strategy.
3) Decide how to implement assessment.
4) Create a list of available models, if any, for use in nutrient assessment of wetlands.

The following two approaches were discussed on how to best address these tasks:

- Identify parameters first and then discuss application in various wetland types
- Identify wetland type first and then discuss what parameters are appropriate for specific types.

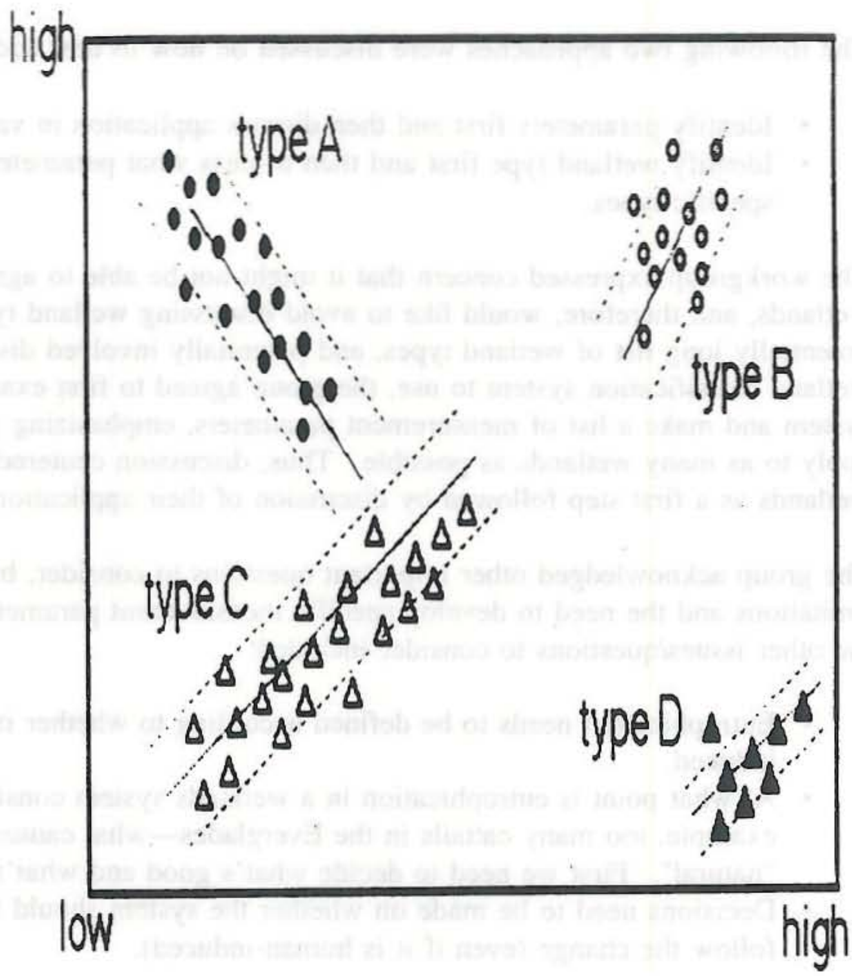
The workgroup expressed concern that it might not be able to agree on definitions of wetlands, and therefore, would like to avoid discussing wetland types initially. Given the potentially long list of wetland types, and potentially involved discussion around the type of wetland classification system to use, the group agreed to first examine a hypothetical wetland system and make a list of measurement parameters, emphasizing parameters that potentially apply to as many wetlands as possible. Thus, discussion centered around processes in wetlands as a first step followed by discussion of their application to wetlands types.

The group acknowledged other important questions to consider, but also recognized the time limitations and the need to develop specific measurement parameters for wetlands. Some of the other issues/questions to consider included:

- Eutrophication needs to be defined according to whether it is natural or human-induced.
- At what point is eutrophication in a wetlands system considered a "bad" thing? (For example, too many cattails in the Everglades—what causes these to spread and is that "natural". First we need to decide what's good and what's bad).
- Decisions need to be made on whether the system should be controlled or allowed to follow the change (even if it is human-induced).

Another suggestion was made by the breakout group that here should be an EPA database for natural wetlands (like that currently available for constructed wetlands). The database should include a wetland type and statistics that apply to each type. Figure 1 illustrates how a national database could be used to compare the measurement parameters of assessed wetlands to an established set of reference conditions. Wetland conditions found to lie outside of the normal range would indicate impacted conditions.

Figure 4. Hypothetical example of how a national database of reference wetlands could be used in assessment of nutrient impacts. Dashed lines represent a hypothetical confidence interval around a given reference set. Type A, B, C, and D refer to different wetland types. Driving functions and response variables can be single or multiple variables.



<u>Ecosystem response variable or index</u>

Community metabolism
Enzyme activity
Soil accretion rate
Sediment oxygen demand
Nutrient retention
Species indices

<u>Ecosystem driving functions</u>

Areal nutrient load index (average and variable)
Areal hydrologic load index (average and variable)
Retention time index (average and variable)
Altitude

The group next discussed the relationship between wetland processes and wetland indicators as one potential way to use indicators to assess wetlands. Parameters measuring wetland processes can be organized by:

- water quality
- soil
- hydrology
- biota
- processes

Important nutrient processes in wetlands were listed as follows:

Carbon
- respiration (microbes)
- aerobic (water, litter, soil)
- anaerobic (soil)
  - sulfate reduction
  - methanogenesis
- plant uptake

Nitrogen
- mineralization
- nitrification/denitrification (includes nitrate reduction)
- biological nitrogen fixation
- plant uptake

Phosphorus
- mineralization
- plant uptake
- soil adsorption

Traditional water quality parameters can also affect or be affected by nutrients and should not be overlooked (e.g., those parameters typically used for assessment of lakes, rivers, and streams). Surface water chemistry (pH, temperature, and cations) should be reviewed as potential parameters. Another issue discussed was groundwater chemistry. Both surface and groundwater quality should. Groundwater interactions and influence on wetlands depend on the type of wetland, but where applicable, groundwater chemistry should be considered.

Additional Issues

Important issues also discussed during the course of parameter identification included the following:

Which nutrients are limiting in a given wetland? This is the critical indicator, and a function of other variables, such as wetland type, water source, latitude, seasonality, time of sampling and other factors. Many of the selected parameters may be irrelevant unless sensitivity of the wetland's response to nutrient enrichment is known. For example, *Spartina* sp. salt marshes may exist at either low or high phosphorous or nitrogen concentrations, and loading rates may make no difference to salt marsh response sensitivity. Some bogs cannot tolerate certain nitrogen and phosphorus variations.

Signal to noise ratio was also discussed as an issue of importance for some of the chemical parameters. Multiple samples would be required to understand the variability observed during sampling within a particular wetland. For wetland assessment, one suggestion was discussed to place Feldspar (silicon dioxide) in a variety of wetlands, marking the area where deposited (e.g., with a global positioning system), and then monitoring for 5 to 10 years. This would be an inexpensive methodology and would show any sediment/organic accretion.

Finally, information gaps identified during the course of the first day's discussion included stress indicators in plant physiology and microbial community structure.

IV. Estuaries and Coastal Waters

The estuaries and coastal waters breakout group approached its review of the current status of nutrient enrichment assessment from three different perspectives: understanding causal relationships between nutrient enrichment and estuarine/coastal waters use impairments; evaluating the quantitative or qualitative nature of relationships between nutrient loadings and system responses; and identification of existing nutrient enrichment assessment tools and necessary next steps organized by several dominant estuarine/coastal communities. The group concluded that there is a real need to develop additional confidence in a select number of key empirical (hopefully causal) relationships, but that the goal of quantitative and predictive relationships for measurement "endpoints" is simply out of reach for most of the usual list of indicative measurements. In response to this, the group chose to identify use impairments (responses) and then to seek an approach to constructing empirical relationships between these and probable (generally accepted) causal agents directly or indirectly linked to nutrient loadings.

Use Impairments

The first round of breakout group discussion yielded the following listing of nutrient overenrichment related use impairments for estuarine and coastal waters:

Beaches

- Toxic and nontoxic algal blooms
- Aesthetics
- Sea Lice (Florida)

Shellfish beds

- Toxic algal blooms (both human and shellfish health concerns; habitat concerns)
- Nontoxic algal blooms (e.g., brown tides -- habitat concerns)

Aquatic habitat

- Seagrass losses (due to both loss of light penetration to leaf surfaces and direct toxic effects from exposure to elevated nitrate in the leaf zone)
- Organic fouling on settling surfaces
- Benthic community impairments (low dissolved oxygen; direct toxic effects through exposure to elevated sediment ammonia, organic carbon concentrations)
- Fish community impairments (low dissolved oxygen, loss of prey, loss of habitat, increased disease prevalence)
- Coral reef dieoffs
- Presence of nuisance species (excessive macrophytes; species shifts)
- Aesthetics (odors, discoloration of the waters)

Nutrient Loading/System Response Relationships

Breakout group members posed three questions to characterize the relative certainty of how well we understand the relationships between nutrient loading and responses of estuarine/coastal systems:

- What do we know quantitatively with regard to nutrient loading-system response?
- What do we know qualitatively with regard to nutrient loading-system response?
- Where do we think there may be a relationship, but we can't quantify or quality the relationship?

Examples of the quantitative relationships included: nutrient loading/chlorophyll *a* and primary production relationships, nutrient loading/dissolved oxygen relationship, and nutrient loading/seagrass production relationships. Two examples were given for qualitative relationships: nutrient loading/fish yields and nutrient loading/macroalgae.

System-based Tools and Needs

The breakout group next discussed nutrient assessment issues within the context of specific types of coastal systems. The first round of breakout sessions concluded with a discussion of coral reef systems.

Coral reef systems

Tools

- Regression relationship of the percent algal cover vs. nutrient concentration to provide the threshold concentrations at which the coral reefs become degraded (e.g., 1 μm dissolved inorganic nitrogen, 0.1 μm soluble reactive phosphorus from research by LaPointe and colleagues).

- Regression relationship of coral coverage by algae vs. nutrient loadings.

Next Steps

- Fund a team of investigators to conduct a critical synthesis of the studies supporting these relationships, publish the results, and get scientific/management community buy-in.

- Support synthesis of existing work and support further studies necessary to quantify the relationship between coral reef community fish diversity, algal cover, and nutrient loading.

- Develop and publish case study examples to provide for straightforward illustrations of the above relationships for the public.

Day Three Plenary Session
Summary of Reports from Breakout Groups and Discussion

On the afternoon of the second day, workshop participants addressed cross-cutting issues within their breakout groups. The focus during these discussions was on how appropriate tools can be applied to accomplish nutrient management goals, the transferability of various tools across waterbody types and geographically and ecologically diverse areas, and the prioritization of user needs.

Workshop participants then met on the morning of the third day to summarize the results of their second round of breakout group discussions. Each breakout group facilitator reported on the recommendations made by their respective breakout group and answered questions posed to the group by other members of the workshop. Important "next steps" and organizations to involve in the process were also identified during this period.

I. Lakes and Reservoirs

The lakes and reservoirs breakout group began its second round of discussion by reviewing several of the objectives of the workshop. They agreed that one component of the national nutrient assessment strategy should be to provide a guidance document to the states regarding the importance of controlling nutrient overenrichment. The guidance document should also identify the best available assessment and control tools. The group also acknowledged that an assumption throughout their discussion was that any type of guidance should be based on a regional approach due to climatological differences between regions as well as variable user perceptions between regions. The ecoregion approach used by Minnesota was brought up as an example of how this could be done. The importance of including land use changes as early warning indicators was stressed.

The next step that the group took was to evaluate the community/ecological structure parameters. A general discussion was held concerning the difference between these types of parameters and the chemical/biomass parameters. It was agreed that the biological/community structure indicators cannot be modeled as well as the chemical/biomass parameters. This is partially due to the fact that there are numerous factors (other than nutrients) that can affect them. The point was also made that biological indicators can be measured in a number of ways—tolerance, intolerance, functional groups, health conditions, diversity (dominance), numbers, threshold effects, and indicator species. It was also suggested that although the public may be most concerned with biological indicators, they are the most difficult to predict. The issues raised in connection with each of the individual community/ecological structure parameters are summarized in the tables which follow.

Algal Community	Rating	Notes
Units of Measurement (several ways to measure)		Algal community could be measured as concentration or taxa composition; need an accepted, cost-effective measurement technique.
Value	High	Value would be very high with an appropriate, accepted parameter; contains information on the state of the system.
Scientific Validity	Promising	
Cost	High	Need more rapid assessment techniques.
Practicality	Variable	Usefulness is hindered by lack of adequate state databases (i.e., need a background to compare to).
Public Understanding		
Modeling Capability/ Load-Response		Group was shown graph displaying the relationship between trophic state index and fish abundance (based on study done by Bruce Wilson, Minnesota Pollution Control Agency).

Combination of Biological Indicators*	Rating	Notes
Units of Measurement (many ways to measure)		Biological indicators can be measured in a number of ways—tolerance, intolerance, functional groups, health conditions, diversity (dominance), numbers, threshold effects, and indicator species.
Value	High	Highly visible indicators of lake biota; however, do not serve as early warning indicators and are not phosphorus specific (i.e., are influenced by a number of factors besides phosphorus.)
Scientific Validity	Variable	Validity depends upon how indicators are used (i.e., good for evaluating consequences, not as good for providing information on trophic state).
Cost	Medium to High	Individual samples cost a lot, but need to sample less often and take fewer samples.
Practicality		Complements other information; basically addresses whether or not species is there, its abundance, and the size of organisms.
Public Understanding	High	
Modeling Capability/ Load-Response	Low	We do not have accepted predictive capabilities for all of these types of endpoints; ones that are available should be used with caution.

* To facilitate the discussion, the group addressed zooplankton, benthos, and fish measurements together.

Macrophytes	Rating	Notes
Units of Measurement (many ways to measure)		Biomass of macrophytes can be measured as grams dry weight/m^2 or grams phosphorus/gram carbon; extent can be measured as areal coverage. Issue of light deprivation and effect on macrophyte growth was raised.
Value	High	
Scientific Validity	High	
Cost	Variable	
Practicality	High	
Public Understanding	High	The public can readily understand the significance of dense weedmats impeding boat traffic and swimming.
Modeling Capability/ Load-Response	Low	Limited by our inability to predict whether a system will be plankton or macrophyte dominated (depends on depth of lake, springtime conditions, substrate, etc.).

General Recommendations

1) Any nutrient strategy should be promoted and developed on a regional basis.

   The breakout group agreed that a national nutrient assessment strategy could only be done if individual regions—both EPA regions and ecoregions within states—were given the flexibility to adopt their own nutrient endpoints (i.e., no single national number should be promulgated by EPA). The regional approach is necessary because of the different types of waterbodies that exist across different climates and the fact that user perceptions of acceptable water quality also differ. Minnesota's ecoregion approach was referred to as an example.

2) The benefits of addressing nutrient overenrichment should be recognized as a public health benefit.

   To gain public support, the impact of nutrient overenrichment on public health needs to be stressed. An important example is the effect nutrient enrichment can have on drinking water supplies (e.g., trihalomethanes and toxic algal blooms).

Specific Tools

1) EPA should publish a national technical guidance document summarizing the Agency's position on (and recommended methods for) nutrient assessment and control.

   This document would serve to eliminate the ad hoc approach that currently exists from one state to another and would motivate certain states to become more proactive. The publication of such a document would also signal that addressing nutrients is a national priority.

2) States should include land use as a separate early warning indicator (e.g., if development is proposed in a watershed, some type of analysis should be done concerning potential impacts on the waterbody.)

   The group realized that this was an obvious point, but wanted to explicitly mention land use change as the most important early warning indicator of nutrient enrichment.

3) In addition, within the national guidance document, states should be encouraged to adopt a nutrient control strategy adopting at least the following set of parameters: secchi depth, dissolved oxygen, total phosphorus, and total chlorophyll).

   The group felt that this "short list" of parameters needed to be included in any state nutrient control program because of their scientific validity, cost, and modeling capability advantages in support of lake assessment.

4) EPA should provide simple, user-friendly, desktop-based software models to state and local governments to aid them in waterbody management decision making.

   This recommendation was strongly endorsed by the members of the breakout group. They noted that advances in computer technology make this easier than ever to accomplish and identified several existing models that could serve as starting points (e.g., BATHTUB, Reckhow-Simpson technique). The group stressed that state personnel should be included in the design team developing such models and that the models should correlate to the recommendations outlined in the guidance document (i.e., include the same parameters and be based on land-use information as well as in-lake processes).

Gaps that Need to Be Addressed

1) The relationship between management practices and resulting load reductions is not well understood.

   The fact that waterbody managers do not understand how much load reduction will be provided by various control practices seriously limits their ability to restore waterbodies to desired conditions. It also limits their ability to argue that the benefits of their proposed actions outweigh the costs.

2) The relationship between nutrient loading and macrophyte growth is not well understood.

   The group realized that macrophytes can be a useful parameter, but that currently there is a lack of information on the relationship between nutrient loading and macrophyte growth.

3) Simple models for addressing reservoirs and impoundments are not at the same level as lake models.

   Although impoundment and reservoir models exists, the group did not feel that they are as well-developed as some of the lake models. The group realized that this was due to the fact that impoundments and reservoirs are more complicated systems that are more difficult to model.

4) The impact of sediment loadings on nutrient enrichment is not well understood. The group felt that the lack of information on the relationship between sediment loading and related nutrient enrichment is a gap that needs to be filled with more research.

Questions and Next Steps

Several questions were posed to the lakes and reservoirs breakout group by other workshop participants. One participant wanted to know whether or not the group had addressed the issue of using particulate phosphorus versus soluble phosphorus as an endpoint. The group answered that they had for the most part only discussed total phosphorus because it is a good indicator in lakes and because of its cost-effectiveness. The group acknowledged the usefulness of measuring the different phosphorus species. Other questions that were asked concerned such issues as: how nutrient controls can lead to curbs on development, the difficulty involved with determining incremental changes in lake water quality, and how nutrient issues should be addressed in the reauthorization of the Clean Water Act.

After addressing these questions, the workshop participants identified several "next steps" that should be undertaken in regard to nutrient issues for lakes and reservoirs.

- EPA should raise the level of priority placed on nutrients in Clean Water section 319 and section 314 programs and within the regional offices.

- Other federal agencies need to be involved in this process (e.g., Corp of Engineers, Bureau of Reclamation, U.S. Geological Survey).

- The North American Lake Management Society (NALMS) could serve as a forum for the development and regional/state implementation of a national nutrient strategy specific to lakes and reservoirs.

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II. Rivers and Streams Summary

Session Review

The rivers and streams breakout group began the session by reviewing some of their conclusions from the previous day. The first topic of discussion concerned the list of potential endpoints for plankton dominated and periphyton dominated systems. The group consensus was that additional endpoints should be added to both lists. The participants considered these additional endpoints to be more of a wish list. The group recognized that the data for these endpoints might be technically difficult to obtain and the cost associated with such acquisition could be prohibitive. However, there was a recognition that, if possible, these are the types of things that really should be considered. Table 2 lists the additional endpoints that the group agreed should be added to the periphyton and plankton dominated lists.

Table 2. Additional Endpoints for Plankton and Periphyton Dominated Systems.

Plankton Dominated	*Periphyton Dominated*
Benthic community metabolism	Benthic community metabolism · *riffle/pool*
Sediment composition · organics · size fraction · nutrients · profile · sediment fluxes, O_2, nutrients	Sediment composition · organics · size fraction · nutrients · profile · sediment fluxes, O_2, nutrients
Secondary production · meiofauna, macroinvertebrates, fish	Secondary production · meiofauna, macroinvertebrates, fish
Dissolved oxygen amplitude	Dissolved oxygen amplitude
Production/respiration	Production/respiration

The discussion then moved to a review of the plankton and periphyton dominated flow diagrams and graphs. In terms of the flow diagrams, the group was generally satisfied with the diagrams and only made a few additions. In particular, the group agreed that the following factors should be added.

- Organic carbon and toxicity were added to both lists as a factors affecting algae and periphyton growth. The group felt that it was very important to consider the impact of organic carbon inputs in addition to looking at phosphorus and nitrogen loading.

- Total suspended solids, and other growth limiters were added as influential factors to the periphyton dominated system diagram. The group noted that total suspended solids is a factor that is unique to periphyton dominated streams.

In reviewing both the periphyton and plankton dominated graphs, the participants noted the following changes.

- Plankton river graph - participants noted that it was assumed that the plankton dominated rivers would behave like lakes. It was agreed that the graph should indicate some of the factors limiting algae growth such as light, travel time, and grazing. In addition, the group agreed that total phosphorus would only extend to approximately 500 µg/l.

- Periphyton river graph - participants felt that both nitrogen and phosphorus concentrations should be considered. As a result, the group agreed to add nitrogen at 50 µg/l as a suggested threshold to the Y-axis.

Modeling

The next issue discussed by the group was the role of modeling in understanding and assessing nutrient loading. The group viewed modeling as a way to explain the interrelationships between nutrient loading and the other factors affecting biomass growth. It was recognized that an improved understanding of these factors would help managers control the system. The group was in consensus that based on the current state of our scientific knowledge, biomass growth could not accurately be predicted in periphyton dominated streams and rivers. The group pointed out that our knowledge of the periphyton dominated systems is especially poor. Participants suggested that the national strategy needs to give high priority to support research and development of process oriented models to improve our level of understanding in this area.

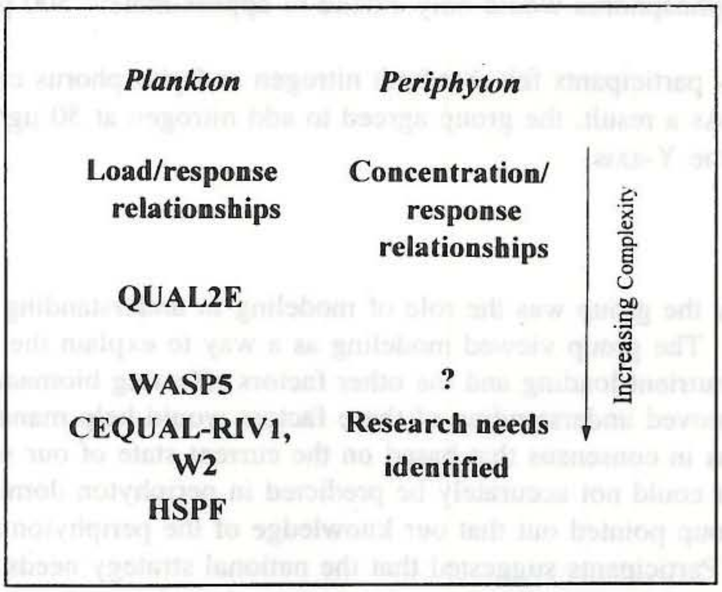
The group then moved into a discussion of some of the know instances or "case studies" where models have been used. Participants focused on how the state of Montana used modeling with a combination of other techniques in setting nutrient load targets for the Clark Fork River. Information from reference reaches, artificial stream experiments, and simplified modeling, were used to define the appropriate nutrient levels or in-stream targets. The targets developed for the Clark Fork River are as follows:

- 6 µg/L soluble reactive phosphorus
- 30 µg/L dissolved inorganic nitrogen
- 300 µg/L total nitrogen
- 20 µg/L total phosphorus
- Biomass 100 mg chlorophyll *a*/m^2 mean, 150 mg/m^2 max

A mass balance nutrient model was then used for the load allocation process to ensure that in-stream targets will be met. It was noted that at the present time only point sources are being considered, but future consideration will be given to nonpoint sources.

The next issue that the group addressed in their discussion of modeling centered on the current state of modeling as it relates to nutrient assessment. The discussion focused on identifying the existing models, characterizing their complexity, and identifying the degree to which these models help explain the processes in both plankton and periphyton dominated systems. Figure 5 is a product from this discussion. It provides information on the current state of receiving water modeling for plankton and periphyton dominated systems across a range of modeling complexity.

Figure 5. Receiving Water Modeling.



A general consensus existed among the participants that more research is needed on investigating what models are out there that can be useful for both managers and scientists. Participants also recognized that we need to know more about nutrient fate and transport (nutrient/algal cycling). In addition, there was a concern expressed over the cost of some of these modeling efforts. Do we have the resources to develop and incorporate these complex models? What is the cost to benefit ratio associated with adopting them?

Modeling tools were particularly lacking in the periphyton dominated systems—from simple mass balance or regression relationships to more complex process based models. Several potential tools for simplified analysis were identified and research needs summarized.

Research Needs

Working from both the modeling discussion and the previous days discussion of the biological aspects of nutrient assessment, the group engaged the issue of future research needs. Participants considered more of a long term time horizon and focused mostly on the scientific or biological issues that are still not well understood. Table 3 provides a list of the research needs that were identified by the participants.

Table 3. Research Needs Identified by the Rivers and Streams Breakout Group

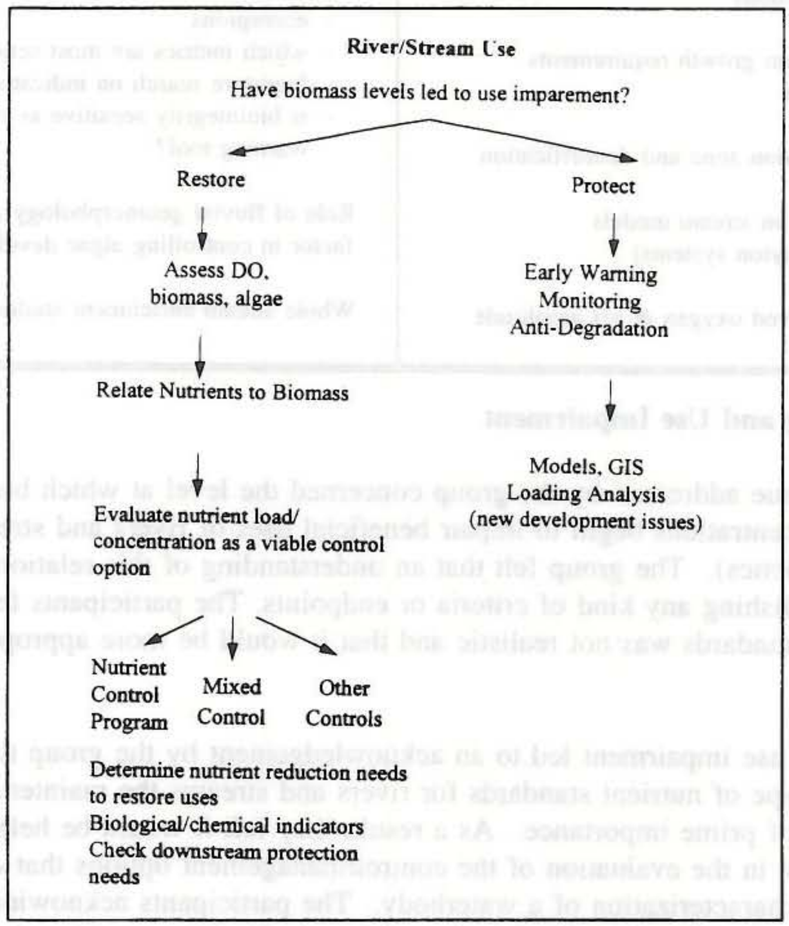
· Chlorophyll measurements (periphyton) - sampling methods · Cladophora, diatom growth requirements - field research · Stream bank riparian zone and denitrification · Literature search on stream models (especially periphyton systems) · Investigate dissolved oxygen & pH amplitude	· Investigate community metrics to characterize rivers for nutrient effects - ecoregions - which metrics are most sensitive - literature search on indicator taxa - is biointegrity sensitive as an early warning tool? · Role of fluvial geomorphology as a factor in controlling algae development · Whole stream enrichment studies

Nutrient Loading and Use Impairment

The next major issue addressed by the group concerned the level at which biomass and chlorophyll *a* concentrations begin to impair beneficial uses of rivers and streams (i.e. recreation or aesthetics). The group felt that an understanding of this relationship was important in establishing any kind of criteria or endpoints. The participants felt, however, that a set of national standards was not realistic and that it would be more appropriate on a ecoregional basis.

The discussion of use impairment led to an acknowledgement by the group that in establishing any type of nutrient standards for rivers and streams the maintenance of the designated use is of prime importance. As a result, they felt it would be helpful to develop a flow chart to assist in the evaluation of the control/management options that could be adopted based on the use characterization of a waterbody. The participants acknowledged that critical to this process was the idea that although the relationship between nutrient loading and periphyte abundance might not be fully understood, we can work from our general understanding of the relationship between nutrient loading and biomass growth. Figure 6 is the flow chart that was developed by the workgroup.

Figure 6. Use Characterization and Management Options.



Other Management Measures

The next issue addressed by the group concerned the identification of management options, other than nutrient controls, that could be implemented to maintain designated uses of streams and rivers. Group members recognized the need for these additional measures based upon our understanding of systems and that the implementation of nutrient control may not be sufficient to restore systems. A mixed or alternative control policy may be required, especially in periphyton dominated systems where nutrients are limiting only at very low concentrations. Table 4 contains a list of the other protection measures that were identified by the group.

Table 4. Other Protection Measures Identified by the Rivers and Streams Breakout Group

· Shade the stream · Riparian zone management · Grazer habitat · Sediment and erosion control · Biological controls · Cattle management	· Channel type - restoration · Hydrology, hydraulics: - stormwater management - stream regulation - flow management (scouring, freezing, minimum flow)

Short Term Needs and Tasks

The group concluded the breakout session with a discussion of what needed to be done in the short term to both improve our understanding of the eutrophication process and to enhance modeling efforts. The group agreed that at the present time there is not enough information to develop a complete guidance document. Group members noted, however, that an important first step should be to update the issue paper and provide states with the available information in an ongoing process. Information on available tools, measures, and case studies could assist in the development of nutrient control and reduction plans in the near future. The group recommended that the experts reconvene in approximately 3 years to discuss their progress. Table 5 is a list of the short term needs and tasks that should be undertaken to update the issue paper and to improve the science.

Table 5. Short Term Tasks and Needs Identified by the Rivers and Streams Breakout Group

· *Literature searches on key areas* - stream modeling techniques - community metrics - designated use and biomass relationships (public survey techniques) · *Modeling* - add temperature simulation to the WASP5 model - add periphyton to the QUAL2E, WASP5, and HSPF models - carbon-based simulations - model maintenance and support - sensitivity analysis studies - sample applications - land use connections in watershed-scale models - improved nitrogen-phosphorus cycling on different land uses (septic, forest systems) · *Investigate seasonal relationships between nutrients and biomass across streams*

Questions and Next Steps

Several questions were asked of the rivers and streams breakout group during the plenary presentation. The workshop participants were especially interested in discussing the use of biological indicators as nutrient assessment endpoints. Several of the participants felt that because previous research in this area had proven unsuccessful, it should not be recommended as a top EPA priority. Other members of the workshop argued that biological indicators are extremely valuable since they can be easily monitored and reflect waterbody changes that the public is most concerned with (i.e., loss of species). One recurring issue was the degree to which biological indicators can be used to determine the *cause* of a systemic change, as opposed to only detecting change itself.

Other points that were raised during this discussion included: the need to scale endpoints in rivers and streams (e.g., by unit substrate or depth of system); the fact that the proportion of periphyton vs. plankton dominated systems is changing (due to increased development in suburban areas); and the need to prioritize the recommended list of endpoints so that EPA can specify a minimum level of effort that is expected of the states.

The next steps that were identified by the workshop participants were:

- Involve other organizations/societies (e.g., North American Benthological Society, American Society of Limnology and Oceanography, American Society of Civil Engineers, American Geophysicists Union, and American Water Resources Association) and have them sponsor special sessions devoted to nutrient issues related to rivers and streams at their upcoming meetings.

- Need to move forward on the causal linkages observed in stream community metrics.

- Need to tap into volunteer monitoring groups as an important resource and constituency.

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III. Wetlands

On the second day, the breakout group continued with parameter identification under the five general categories.

Water Quality

Parameters added to the 'water quality' category were conductivity, dissolved organic carbon, and conservative tracers (e.g., uranium). Each of the categories was discussed and clarified as to whether it related to biological or other aspects. The issue of variability was also discussed. It was noted that the signal to noise ratio is a function of sampling frequency. Timing of sampling is also an important factor to consider. One point was that measures could be direct or indirect indicators. A direct indicator would be an actual measurement of a nutrient; an indirect indicator would be the response of vegetation to a nutrient overenrichment. A question that was raised was: how is a response measured without having a database with which to compare the samples? In order to measure a parameter, we need to understand what is being impacted.

Hydrology

Parameters that were added were surface water inflow/outflow (input/output), water source, and hydroperiod. Residence time will be an extremely variable measurement parameter. Wetland area to catchment area ratio is also important. All this background information is necessary to understand the processes potentially affecting wetlands.

The group agreed to present the following recommendations to the plenary session:

- There is a need for a nationwide database that characterizes wetlands as pristine or impacted. This would be similar to EPA's constructed wetlands database. It was suggested that tables be developed through a comprehensive literature search showing what work has already been done on nutrient-related issues in wetlands. This information could serve as the initial foundation for a wetlands database, or could be added to a wetlands database.
- The breakout group reorganized the list of parameters into five categories and discussed each parameter's relevance to monitoring.
- A short-term strategy for addressing nutrient overenrichment issues in wetlands is needed be current wetlands managers; a long-term strategy incorporating ongoing research is also critical.
- The Army Corps of Engineers is working on implementing the Hydrogeomorphic Wetland Classification system; the goal is to cover 80 percent of all wetlands, including the collection of a lot of variables on these wetlands. One issue is who will host and maintain the data. Can states access these reference data sets? Can states add parameters that are important? Impacts and functionality of wetlands are a part of this project. The scope could potentially be broadened so that it can apply to other management issues, such as nutrients.

- The 'Secchi Disk' of wetlands could be Feldspar (silicon dioxide) to show accumulation within wetlands over time as an indication of whether there has been a change in productivity. Feldspar does not react with other chemicals in the soil.

Is water quality sampling for wetlands included in STORET? No one in the group could initially answer this question. However, subsequent investigation found that some limited information is available on wetlands in STORET. In particular, Michigan and Florida have sampling stations in wetlands, but Michigan includes fish tissue data only. Florida has sampling information on wetlands, including nutrient concentrations (see Figure 7).

Figure 7. Distribution of sampling stations in the USEPA STORET database that are wetlands. The figure emphasizes the sparse national distribution on wetlands data.

In the afternoon session it was discussed whether there may be a better way to classify wetlands for understanding of nutrient impacts (i.e., other than Cowardin, et al., 1979). The hydrogeomorphic classification approach has three major themes - water source, hydrodynamics, physiographic setting (includes: slope, soils, watershed). The classification system looks at the primary source of change.

There was further discussion on whether there is a set of indicators that can potentially extend across all wetlands. The breakout group decided that the most productive course should be to list all the relevant indicators and highlight the most important indicators. Bioavailable nitrogen and phosphorus in sediments, soils, water, vegetation, hydrology, and loads are the indicators that are useful in a diagnostic study. These indicators should provide some indication of nutrient impacts. One concern is that there is no baseline data available with which to compare the collected data. Reference sites need to be chosen and baseline data determined. This can become the long-term database. From the baseline data, another wetland can be analyzed and the data could support determining if the wetland is impacted or not impacted. Because of the variability in wetland systems, the parameters cannot be prioritized or ranked. The group also cannot rank them as the "best" indicators to use.

Thus, long and short-term issues were simplified as follows:

Long-term → setting up a reference baseline database
Short-term → using a reference site next to the impacted site

Parameters for use in a short term strategy (independent of wetland type) include the following:

- Bioavailable nitrogen and phosphorus in soils and water
- Plant species composition
- Plant species richness
- Plant species structure (including recruitment)
- Plant indicator species - vascular and non vascular species
- Soil oxygen demand (indicator of microbial respiration)

Volunteer monitors would be able to measure everything on the short-term list. A suggestion was made to create several central analyzing facilities that volunteer programs could send samples to for standardized analysis. These labs would serve as the QA/QC centers to ensure a standardized sample analysis.

The long-term list of measurement parameters is included in Table 6. It was suggested that a recommendation to EPA would be to continue literature searches, especially for long-term studies.

Models

There is a model under development for understanding how wetlands function in the Everglades. Water budget, flow fields, vegetation, and biology are considered. This has been calibrated against data sets. There are some other models on the Des Plaines wetlands and Odum Creek (Florida) but they do not have as large a spatial scale as the Everglades model does. Spatial scale is an important aspect in assessing wetlands. Landscape models can address a broader spatial scale, but are much coarser in their ability to predict impacts. For example, each grid is 1 km by 1 km, so there is not as much detail. These types of models have shown some promise in South Florida. Additional literature searches on wetland models may identify other modeling approaches or applications.

The group did not address modeling efforts that compute/predict nutrient loadings to wetlands (i.e., transport models successfully used for other water bodies), and these may have important roles in helping to manage loading-related impacts to wetlands.

Table 6. Measurement Parameters Identified by Wetland Breakout Group

SURFACE WATER QUALITY

*Total Kjeldhal nitrogen
*Ammonia
Nitrate
Soluble reactive phosphorus
Total phosphorus
Total suspended solids
Chlorophyll *a*
Nitrogen:phosphorus ratio
Nutrient limitation
Dissolved oxygen
pH
Temperature
Conductivity
Dissolved organic carbon
Conservative tracer

BIOTA

*Nitrogen
*Phosphorus
*Nitrogen:phosphorus ratio
Nutrient uptake
*Changes in species composition
Vegetative structure
Plant species diversity/richness
Macroinvertebrate species diversity/richness
Indicator species
Net primary productivity
Recruitment
Leaf area/solar transmittance (remote sensing change detection)

HYDROLOGY

Wetland area
Depth
Transpiration
Precipitation
Residence time
Groundwater flow
Surface water inflow/outflow
Hydroperiod

SOILS

Total Kjeldhal nitrogen
Ammonia
Nitrate
Total phosphorus
Nitrogen:phosphorus ratio
*Soil oxygen demand
Sand/silt/clay fraction
Redox potential/pH
Organic accretion
Sediment accretion
Microbial biomass
Enzyme activity
Bioavailable nutrients

PROCESSES

Carbon
- **microbial respiration**
  - **Aerobic**
    - **Water column**
    - **Litter**
    - **Soil**
  - **Anaerobic**
    - **Soil**
    - **Sulfate reduction**
    - **Methanogenesis**

Plant uptake
*Nitrogen
- Mineralization
- Nitrification/denitrification
- Biological nitrogen fixation
- Plant uptake

*Phosphorus
- Mineralization
- Plant uptake
- Adsorption capacity

Parameters marked with an asterisk (*) are "early indicators".
Parameters in **bold type** are parameters that are specific to one category.

Some members expressed concern that there is a gap in the availability of well-accepted wetland models, but other members argued that numeric models may not be useful in predicting changes within wetlands. Rather, more simple (i.e. heuristic) models may be more helpful in understanding nutrient-related responses within wetlands.

In closing the session, the breakout group identified several information gaps in understanding nutrient-related impacts to wetlands. Based on these discussions, the breakout group submitted a series of recommendations at the closing plenary session.

1) There needs to be an accepted national wetland classification system similar to the hydrogeomorphic classification system developed by the U.S. Army Corp of Engineers.

2) A *comprehensive* literature review needs to be undertaken with the purpose of parameter and data extraction to create "threshold" table(s).

3) A nutrient database including baseline data according to wetlands types/regions needs to be developed (long-term benefit).

4) Field experimentation should be conducted to determine nutrient limitation by wetland type and to isolate the effects of nutrients from other variables, such as hydrology, climate, events, and natural variability.

5) Integrate parameter data collection with wetland indicators, or possibly other EPA wetlands program(s).

6) Volunteer monitoring needs to be encouraged, as well as the development of state capability of monitoring.

7) Need to improve upon molecular biology techniques e.g., stress proteins in plants.

8) There needs to be further development of remote sensing techniques.

Questions and Next Steps

Workshop participants had several questions for the wetlands group. They wanted to know whether or not there are any existing relationships for certain types of wetlands that could be used to develop nutrient assessment and management tools. The wetlands group responded by pointing out that this research area is in its infancy compared to work that has already been done for the other waterbody types. Workshop participants also inquired about how the hypothetical wetlands database would be organized and what information it would provide to resource managers.

The next steps that were identified were:

- Involve organizations such as the State Association of Wetlands Managers, Society of Wetlands Scientists, and private organizations (e.g., Ducks Unlimited) in the development and implementation of the wetlands component of the national nutrient assessment strategy.

- Need to synthesize the results from the workshop with a proposal for how to proceed from here.

- Need to better articulate the level of concern about nutrient overenrichment of natural wetlands and communicate to policymakers and the public.

- Need to synthesize information on the relative sensitivity of different wetlands types to nutrient enrichment and use this information to investigate regressional relationships.

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IV. Estuaries and Coastal Waters

Over the course of the workshop, several major themes and recommendations emerged from the group's detailed discussions. Perhaps the most pervading challenge in establishing tools and standards for assessment and management of nutrient inputs to aquatic systems is the fact that large, real variability exists in physical and biological structure and processes among all systems. This variability means that a given input results in a different response. While science may agree qualitatively about the expected response, quantitative prediction is rarely if ever possible with acceptable confidence limits. In addition, regional differences exist in human/societal expectations so that what condition is "acceptable" is also not uniform. Taken together, these perspectives suggest that arbitrarily defined standards of general applicability may be untenable, and strongly call for a different approach.

Most of the group's effort was initially directed toward what metrics seem meaningful, rigorous, and yet practically useful in the assessment of effects. The group set aside for the moment the issue of subjective definition of standards of what is acceptable or unacceptable as a basis for regulation. However these standards are defined, they must be based on acceptable measures of dose response.

Faced with this, the estuaries work group approached a consensus that the only practical initial approach is to construct empirical relationships between indicator responses and probable driving variables. This is essentially the approach taken by Vollenweider in documenting responses of lakes to phosphorus loading. Some examples of these empirical relationships are clearly significant statistically, but they display large variability, even for comparable estuaries. For only plankton-based systems, for example, chlorophyll stocks show a 10-fold range in response for similar inputs of total nitrogen. For other types of systems, however, such relationships have not been adequately explored. It is recommended that significant effort be devoted to a careful gathering of available information for a variety of systems, and an incisive analysis of relationships. It must be emphasized that this is not a menial task. Care and insight are necessary to evaluate each published study to assure appropriate comparisons are made.

A benefit of this approach is that the power of the method will increase with time. As more information becomes available, three significant areas of progress in the empirical approach may be anticipated. First, separate relationships may be developed for different types of systems, effectively stratifying the variance among more similar groups of sites (e.g. micro- vs. macrotidal). The group separated benthic, seagrass, and plankton based systems with this rationale. Second, schemes for parameterizing multiple factors may be developed within which tighter relationships may be possible (e.g., Vollenweider's parameter normalized phosphorus input to freshwater turnover time). Third, as time series of data for an increasing number of cases become available, tighter relationships will emerge, more strongly suggesting site specific predictive relationships.

In some cases, natural year-to-year variability may define the dose-response quite accurately for a given system. This empirical approach guarantees a number of benefits. With time, the archive of systems for which diagnostic dose-response measurements are available will increase. Thus the use of empirical relationships has immediate value, but is also perfect stimulus for adaptive management using the more predictive relationships that will emerge with time.

Seagrass Dominated Systems

Tools

- Areal survey of seagrasses (aerial photography/mapping/digitization; ground surveys via transects) to document distribution, abundance, and depth penetration of seagrass beds.

- Simple to complex nutrient loading models of the surrounding watershed and water quality models for simulating chlorophyll *a* concentrations over the seagrass beds.

- Methodologies for partitioning out the various water column light attenuators (e.g., chlorophyll *a*, inorganic suspended solids, color) to determine the relative contribution due to nutrient enrichment:

  - Multiple regressions of various fractions of total suspended solids, color, chlorophyll *a*, etc. using multi-wavelength spectral methods
  - Literature values of specific extinction coefficient (chlorophyll *a*)
  - Light transmission models (e.g., Gallegos 1994)

- The 20% incident light requirement that holds up across a range of seagrass species (recognizing one has to consider both water column and leaf surface attenuation due to epiphytes at the site/waterbody specific level).

- Field tested protocols for designing, conducting, and analyzing results from water quality monitoring along seagrass gradients (depth penetration, vegetated to non-vegetated, and water quality gradients).

- Field tested protocols for assessing relationships between seagrass depth penetration vs. light attenuation coefficient to determine waterbody specific light requirements.

- Host of field tested/possible early warning indicators of pending seagrass declines:

  - C:N:P ratios in above- and below-ground tissue, blade width, Hsat, quantum irradience, leaf chlorophyll *a*, chlorophyll *a* to *b* ratios, stable isotope.

Next Steps

- Sort out the light requirements over different regional habitats of the dominant seagrass species from the existing wealth of literature.

- Determine, synthesize and publish thresholds for nutrient concentrations, light penetration, total suspended solids, chlorophyll *a*, etc. under different environmental conditions (e.g., temperature) based on the current and soon to be published literature.

- Through a workshop/working group, develop an approach which will enable states and estuarine/coastal management programs with minimal to no existing SAV survey data (but with historical or ongoing seagrass declines) to set: 1) light and nutrient thresholds, 2) depth penetration restoration goals, 3) distribution restoration goals, 4) and seagrass resource restoration based nutrient reduction goals for the surrounding watershed.

- Assemble a series of teams of experts to actively assist these states and estuarine/coastal management programs in the design and conduct of the necessary studies and data interpretations.

- Publish a full accounting of the approaches taken in Chesapeake Bay and Florida for using seagrasses to set nutrient reduction targets in a form for ready application to other semi-tropical to temperate systems.

- Pull from the literature simplified versions of the land use mosaic/nutrient loadings relationships.

- Devote greater effort to modeling in seagrass restoration given actual or predicted water quality changes based on the reproductive biology of the waterbody's species.

- Identify species-appropriate early warning indicators, synthesize the literature supporting field validation/case study verification of the utility of these indicators, and work to build these indicators into ongoing and planned seagrass monitoring programs to build up the requisite data base.

Plankton Dominated Systems

Tools

- A range of empirical relationships and more limited set of models connecting nutrient loadings and "top-down" trophic influences interacting to control phytoplankton production.

- Techniques for separating photopigments into chlorophyll *a* and other diagnostic pigments (chlorophylls and carotenoids) with analysis by HPLC and C_{14} incorporation. These techniques can provide insight into the composition and growth rates of the algae in order to predict algae composition in response to nutrient enrichment.

- Field tested techniques and instrumentation necessary to establish monitoring stations to define a dissolved oxygen fluctuation signature for a system.

- Techniques for general source discrimination using N_{15} and C_{14} stable isotopes.
- Satellite/aerial imagery and interpretation techniques to display chlorophyll *a* and other diagnostic pigment patterns with much greater spatial resolution and estimate surficial primary productivity.
- In-depth information about the sometimes-complex life cycles of some species.
- Scanning and transmission electron microscopy to aid in identification of picoplankton (e.g., brown tide organisms) and other phytoplankton componenets (e.g., small dinoflagellates).
- Mesocosms for studying species succession and community- or ecosystem-level response to nutrient inputs.
- Molecular probes for rapid, reliable species detection, identification and enumeration.

Next Steps

- Need to further examine the role of silica relative to nitrogen and phosphorus (silica depletion in phytoplankton blooms).
- Need to take the short list of key questions that keep coming up again and again in these types of workshop but never get resolved and bring together a group drawn from key parts of the community (Estuarine Research Federation, American Society of Limnology and Oceanography, Benthic Society, NOAA, EPA, and others) to synthesize the existing information and come to a conclusion.
- Need to develop more comprehensive dissolved oxygen tolerance/effects thresholds for systems beyond Virginian Province (Cape Cod to Cape Hatteras) supporting waterbody specific determination of dissolved oxygen criteria.
- Need to tackle the question of what amount of dissolved oxygen fluctuations are due to natural vs. anthropogenic causes.
- Need technological refinements in the areas of bioindicators and sensors of nutrient enrichment. Specifically, there is a need for indicators capable of reflecting *natural* community responses.
- Need an equivalent set of relationships for plankton dominated systems for nutrient loadings to chlorophyll *a* response (estuarine version of the Vollenweider relationships).
- Continue to unravel the complex life cycles of some abundant algal taxa, and to determine nutritional controls on stage dominance.

- Further examine the role of changing N:P ratios on shifts in phytoplankton dominance from desirable to undesirable species, and the role of silica in these species shifts.

Nuisance Algal Blooms

Harmful Nontoxic Algal Blooms

Tools

- A range of empirical relationships connecting nutrient loading with species growth and abundance.

- Techniques for separating photopigments into chlorophyll *a* and other diagnostic pigments (chlorophylls and carotenoids) with analysis by HPLC and C_{14} incorporation. These techniques can provide insight into the composition and growth rates of the algae in order to predict algae composition in response to nutrient enrichment.

- Field tested techniques and instrumentation necessary to establish monitoring stations to define a dissolved oxygen fluctuation signature for a system.

- Techniques for general source discrimination using N_{15} and C_{14} stable isotopes.

- Satellite/aerial imagery and interpretation techniques to display chlorophyll *a* and other diagnostic pigment patterns with much greater spatial resolution and estimate surficial primary productivity.

- In-depth information about the sometimes-complex life cycles of some species.

- Scanning and transmission electron microscopy to aid in identification of picoplankton (e.g., brown tide organisms) and other phytoplankton componenets (e.g., small dinoflagellates).

- Mesocosms for studying species succession and community- or ecosystem-level response to nutrient inputs.

- Molecular probes for rapid, reliable species detection, identification and enumeration.

Next Steps

- Need to further examine the role of silica relative to nitrogen and phosphorus (silica depletion in phytoplankton blooms).

- Initiate long term monitoring to collect data to provide the basis for correlating brown tide blooms with nutrient enrichment.

- Test laboratory based findings in controlled field experimentation.

- Support a more comprehensive synthesis of correlative/experimental evidence for brown tide blooms responses to nutrient enrichment.

- Continue to unravel the life cycles of harmful nontoxic algal taxa, and to determine nutritional controls on stage dominance.

- Continue to develop/improve molecular probes to detect, identify and enumerate harmful algal species.

- Continue to develop isotopic methods to trace the source of nutrients to harmful algal bloom species and to determine relative importance of natural versus anthropogenic nutrient inputs in stimulating their growth.

- Improve remote-sensing algorithms for detecting harmful algal blooms and the water masses with which they are associated.

- Establish long-term monitoring programs for both bloom-forming (e.g., brown tide species) and low-abundance harmful species (e.g., "needle" forming diatoms such as *Chaetoceros concavicornis*), in order to document major trends and linkages to cultural eutrophication.

- Identify nontoxic harmful algal bloom species that can serve as "indicators" of different types or conditions of eutrophication. Develop these species as indices of ecosystem "stress."

- Determine nutrient uptake characteristics and requirements for key nontoxic harmful algal bloom species, allowing for the sensitivity of many harmful species to mixing conditions and enclosure effects.

- Determine the interactions of both inorganic and organic nutrient sources in stimulating nontoxic harmful algal bloom species.

- Develop physicological indicators of nutrient limitation in nontoxic harmful algal bloom species to assess their nutrient status under both natural and nutrient enriched conditions, and including consideration of both nutrient supplies and supply ratios.

Toxic Algal Blooms

Tools

- A range of empirical relationships connecting nutrient loading with species growth and abundance.

- Techniques for separating photopigments into chlorophyll *a* and other diagnostic pigments (chlorophylls and carotenoids) with analysis by HPLC and C_{14} incorporation. These techniques can provide insight into the composition and growth rates of the algae in order to predict algae composition in response to nutrient enrichment.

- Field tested techniques and instrumentation necessary to establish monitoring stations to define a dissolved oxygen fluctuation signature for a system.

- Techniques for general source discrimination using N_{15} and C_{14} stable isotopes.

- Satellite/aerial imagery and interpretation techniques to display chlorophyll *a* and other diagnostic pigment patterns with much greater spatial resolution and estimate surficial primary productivity.

- In-depth information about the sometimes-complex life cycles of some species.

- Scanning and transmission electron microscopy to aid in identification of picoplankton (e.g., brown tide organisms) and other phytoplankton componenets (e.g., small dinoflagellates).

- Mesocosms for studying species succession and community- or ecosystem-level response to nutrient inputs.

- Molecular probes for rapid, reliable species detection, identification and enumeration.

Next Steps

- Initiate long term local and regional monitoring to collect data to provide the basis for correlating toxic algal blooms with nutrient enrichment.

- Test laboratory based findings with controlled field experimentation.

- Utilize bioindicators (molecular, immunological) of linkage between nutrient enrichment and toxic/harmful characteristics of algal blooms.

- Support a more comprehensive synthesis of correlative/experimental evidence for toxic algal blooms responses to nutrient enrichment.

- Assessment of effects of other variables on nutrient contribution to nuisance tide generation.

- Examine linkage between newly-recognized and growing local/regional sources of nutrients and bloom dynamics in estuarine and coastal waters (i.e., atmospheric deposition and groundwater).

- Need to further examine the role of silica relative to nitrogen and phosphorus (silica depletion in phytoplankton blooms).

- Techniques for separating photopigments into chlorophyll *a* and other diagnostic pigments (chlorophylls and carotenoids) with analysis by HPLC and C_{14} incorporation. These techniques can provide insight into the composition and growth rates of the algae in order to predict algae composition in response to nutrient enrichment.

- Initiate long term monitoring to collect data to provide the basis for correlating brown tide blooms with nutrient enrichment.

- Test laboratory based findings in controlled field experimentation.

- Support a more comprehensive synthesis of correlative/experimental evidence for brown tide blooms responses to nutrient enrichment.

- Continue to unravel the life cycles of harmful toxic algal taxa, and to determine nutritional controls on stage dominance and toxin production.

- Continue to develop/improve molecular probes to detect, identify and enumerate harmful algal species, and to detect/quantify toxins.

- Continue to identify and characterize toxin bioaccumulation impacts through food webs.

- Establish research efforts to determine chronic sublethal impacts of toxic algal species on life history stages of dominant herbivores and predators (e.g., shellfish and finfish), and the influence of nutrient overenrichment in exacerbating these impacts.

- Continue to develop isotopic methods to trace the source of nutrients to harmful algal bloom species and to determine relative importance of natural versus anthropogenic nutrient inputs in stimulating their growth.

- Improve remote-sensing algorithms for detecting harmful algal blooms and the water masses with which they are associated.

- Establish long-term monitoring programs for both bloom-forming (e.g., brown tide species) and low-abundance harmful species (e.g., "needle" forming diatoms such as *Chaetoceros concavicornis*), in order to document major trends and linkages to cultural eutrophication.

- Identify nontoxic harmful algal bloom species that can serve as "indicators" of different types or conditions of eutrophication. Develop these species as indices of ecosystem "stress."

- Determine nutrient uptake characteristics and requirements for key nontoxic harmful algal bloom species, allowing for the sensitivity of many harmful species to mixing conditions and enclosure effects.

- Determine the interactions of both inorganic and organic nutrient sources in stimulating toxic harmful algal bloom species.

- Develop physicological indicators of nutrient limitation in toxic harmful algal bloom species to assess their nutrient status under both natural and nutrient enriched conditions, and including consideration of both nutrient supplies and supply ratios.

Macroalgae Dominated Systems

Tools

- N_{15} signature connecting wastewater effluent with macroalgae.

Next Steps

- Compile the various seagrass/macroalgae/phytoplankton/nutrient loadings data sets (e.g., Florida Bay, Tampa Bay, MERL, Waquiot Bay, MA, Great Bay, NH) and further confirm the paradigm shift and pull out the thresholds relating to these community shifts.

- Quantify the relationship between fish diversity, macroalgal cover, and nutrient loading.

- Quantify the macroalgae influence on dissolved oxygen concentrations, dissolved organic carbon concentrations, and lower trophic levels.

Questions

There were several questions and issues raised at the conclusion of the estuaries and coastal waters plenary session presentation. One participant recommended that endpoints be used cautiously until the basic science underlying these types systems is more well understood; he was concerned about making too many decisions based solely on empirical relationships. Other participants emphasized the importance (as well as the difficulty) of determining accurate land use loading estimates and recommended that biologists and engineers work together to confront this technical gap. Other comments that were voiced included: taking a linked management/science approach and keeping in mind the significance of spatial and temporal scales in regard to the nutrient overenrichment issue.

For questions and comments regarding the estuaries and coastal waters breakout group discussions, please contact either of the following facilitators:

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V. Workshop Wrap-up[4]

The workshop concluded with Rich Batiuk outlining EPA's plans for carrying the nutrient overenrichment issue forward. These plans include distributing the draft proceedings to workshop participants along with revised versions of the issue papers for review and comment. A draft national Nutrient Assessment Strategy will be drafted based on input from the workshop and presented to EPA Program Offices, EPA Regional Offices, ORD laboratories, federal agencies, states, and professional societies in the spring of 1996 for review and comment. A draft final strategy may then be submitted to the Science Advisory Board in the summer of 1996 and a final EPA strategy will be published in the fall of 1996.

[4] Summaries of the overheads used during the workshop wrap-up are included at the end of the proceedings.

Appendix

- **Agenda**
- **List of Workshop Participants**
- **Participant Addresses**
- **Summary of Overheads**

National Nutrient Assessment Workshop Agenda
December 4-6, 1995
Washington, D.C.

December 4 **Theme: Waterbody Type-based Endpoints, Assessment Methodologies & Models**

Plenary Session

8:30 am - 9:00 am	Registration *(1st Floor Meeting Room)*
9:00 am - 9:15 am	Welcome, Purpose & Goals of the Workshop, Introductions *(Rich Batiuk, EPA Chesapeake Bay Program Office)*
9:15 am - 9:30 am	Introductory Remarks *(Tudor Davies, Director, Office of Science and Technology, and Robert Wayland, Director, Office of Wetlands, Oceans, and Watersheds)*
9:30 am - 10:30 am	Past, Present, and Future Perspectives on the Assessment and Management of Nutrient Overenrichment Needs *(Robert Thomann, Manhattan College; Mimi Dannel USEPA HQ (invited); and Steve Heiskary, Minnesota Pollution Control Agency)*
10:30 am - 11:00 am	Purpose & Goals of the Workshop; Purpose of Breakout Groups and Charge to Breakout Groups *(Rich Batiuk, EPA Chesapeake Bay Program Office)*

Concurrent Breakout Groups - Waterbody Type-based Issues

11:00 am - 12:00 pm	Breakout Groups (Organization & Planning) For the morning and afternoon sessions, there will be 4 concurrent breakout groups, organized according to waterbody type: estuaries and coastal waters; lakes, impoundments, ponds; rivers and streams; and wetlands. Each breakout group will have a facilitator, a synthesizer, and a recorder. Each breakout group will be asked to discuss and come to consensus on a number of issues and questions. Specific questions and background materials will be provided in advance of the workshop to enable participants to prepare for workshop discussions.
12:00 pm - 1:00 pm	*LUNCH (on your own)*

Concurrent Breakout Groups - Waterbody Type-based Issues

1:00 pm - 5:30 pm	Continuation of Breakout Groups

December 5		**Theme:**	**Cross-cutting Issues - Integration of Tools Across Waterbody Types and Geographical Areas**

Concurrent Breakout Groups - Waterbody Type-based Issues

8:30 am - 10:30 am — Continuation and Closure of Breakout Group Discussions

10:30 am - 10:45 am — *BREAK*

Plenary Session

10:45 am - 11:45 am — Brief Synthesis Reports from Breakout Groups & Discussion

11:45 am - 12:00 pm — Purpose of Afternoon Breakout Group Discussions and Charge to Breakout Groups

12:00 pm - 1:00 pm — *LUNCH (on your own)*

Concurrent Breakout Groups - Cross-Cutting Issues

1:00 pm - 5:30 pm — Breakout Groups

Workshop participants will return to their breakout groups to discuss and come to consensus on a new set of issues and questions directed toward cross-cutting issues. The focus of these breakout groups will be on how appropriate tools can be applied to accomplish nutrient management goals and the transferability of various tools across waterbody types and geographically and ecologically diverse areas. Participants will also be asked to begin to identify and prioritize user needs and the steps needed to address these needs.

Breakout groups will again have a facilitator, a synthesizer, and a recorder.

December 6 | **Theme:** | **Strategy for Development & Validation of Needed Nutrient Assessment Tools (Bringing Everything Together)**

Plenary Session

8:30 am - 10:00 am Reports from Breakout Groups & Discussion

10:00 am - 12:00 pm Synthesis - Developing a Strategy

As a group, participants will identify and prioritize the available tools for addressing nutrient overenrichment that should be promoted and refined. In addition, we will try to identify and rank new tool development needs. Participants will be asked to discuss and evaluate a straw EPA strategy and create a "to do list" for EPA . Finally, the group will begin to brainstorm potential roles and responsibilities of other state, regional, federal, and academic nutrient assessment partners within the context of a larger national strategy framework.

12:00 pm - 12:30 pm Wrap-up & Next Steps
(Rich Batiuk, EPA Chesapeake Bay Program Office)

12:30 pm *ADJOURN*

Note: *Following the workshop, EPA will produce a meeting summary, including revised versions of the issue papers used at the workshop, and a draft of EPA's strategy document. This will be provided to all workshop participants for review and comment.*

National Nutrient Assessment Workshop—List of Participants by Breakout Group

Estuaries and Coastal Waters

Rich Batiuk, U.S. EPA, Chesapeake Bay Program Office
Suzanne Bricker, NOAA Office of Resources Conservation and Assessment
Tom Brosnan, NY City Harbor Monitoring Program
JoAnn Burkholder, North Carolina State University
Dominic DiToro, HydroQual Inc.
Ken Dunton, University of Texas Marine Sciences Institute
Alan Hais, U.S. EPA, Office of Science and Technology
Joe Hall, U.S. EPA, Office of Wetlands, Oceans, and Watersheds
Michael Kemp, University of Maryland
Jim Kremer, University of Connecticut
Brian LaPointe, Harbor Branch Oceanographic Institute
Virginia Lee, University of Rhode Island
George Loeb, U.S. EPA, Office of Wetlands, Oceans, and Watersheds
Chris Madden, University of Maryland
Wayne Magley, Florida Department of Environmental Regulation
Hassan Mirsajadi, Delaware Department of Natural Resources & Environmental Control
Scott Nixon, University of Rhode Island
Cynthia Nolt, U.S. EPA, Office of Science and Technology
Hans Paerl, University of North Carolina
Ruth Swanek, North Carolina Department of Environment, Health & Natural Resources
David Tomasko, Sarasota Bay National Estuary
Ivan Valiela, Boston University Marine Program
Steve Weisberg, Senior Scientist, VERSAR, Inc.

Lakes, Reservoirs, and Ponds

Robert Carlson, Kent State University
Steve Chapra, University of Colorado
Jeroen Gerritsen, Tetra Tech, Inc.
George Gibson, U.S. EPA, Office of Science and Technology
Steve Heiskary, Minnesota Pollution Control Agency
Dianne Reid, North Carolina Division of Environmental Management
Joel Salter, U.S. EPA, Office of Science and Technology
Eric Smeltzer, Vermont Department of Environmental Quality
Robert Thomann, Manhattan College

Rivers and Streams

Dennis Anderson, Colorado Department of Public Health and Environment
Dan Butler, Oklahoma Conservation Commission
Mimi Dannel, U.S. EPA, Office of Wetlands, Oceans, and Watersheds
Gary Ingman, Montana Water Quality Division
Russ Kinerson, U.S. EPA, Office of Science and Technology
Jerry LaVeck, U.S. EPA, Office of Science and Technology
Lewis Linker, U.S. EPA, Chesapeake Bay Program Office
Winston Lung, Univerity of Virginia
Dennis Newbold, Stroud Water Research Center
Patrick Ogbebor, U.S. EPA, Office of Science and Technology
Greg Searle, Wisconsin Department of Natural Resources
Leslie Shoemaker, Tetra Tech, Inc.
Amy Sosin, U.S. EPA, Office of Wetlands, Oceans, and Watersheds
Sam Stribling, Tetra Tech, Inc.
Gene Welch, University of Washington

Wetlands

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Bob Cantilli, U.S. EPA, Office of Science and Technology
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Andy Hooten, Tetra Tech, Inc.
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Ramesh Reddy, University of Florida
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Summary of Overheads Used in Day One Plenary Session

Robert Thomann, Manhattan College

I. Nutrient assessment & management—A long history: two periods

A. Observational Descriptive (Beginning c. 1930)

B. Predictive Management (Beginning c. 1965)

II. Observational descriptive period (c. 1930)

A. Juday, Birge, Hutchinson, Sawyer, Mortimer, Kethem

B. Known: Adverse Effects of High Loading: Estuaries, Lakes, Rivers

C. Known: Limiting Nutrient Concentration IN 30: IP 10 ug/L

D. Known: Nutrient Loadings; Point (Sewage), Non-Point (Farms, Urban, Atmospheric)

E. NOT KNOWN: How to Relate Loading to Water Body Response

III. Predictive management period (c. 1965)

A. Critical Loading Models for Lakes

1. Vollenweider:

a. Feb. 16 & 17th, 1966, Paris. OECD request: Gather the eutrophication literature, but with the hope that "...the report would have a 'practical' bias and not be too 'scientific'"

B. First Model to Relate Nutrient Loading to Lake Trophic Status

1. Assumptions:

a. Steady State, Completely Mixed Lake

b. Total Phosphorus: Measure of Trophic Status

2. Allow. TP Load = (0.01)*(10+H/det t)

C. Regression Models

D. Dynamic Interactive Models: All Waters

1. Di Toro (1970)

a. San Joaquin River: Effect of Nutrient Loads & flow Diversions on Sacramento-San Joaquin Delta

- Single Volume, Time Variable, CSMP/1130

E. 1970 - 1980 Rapid Expansion: Spatial Detail, State Variables

1. Great Lakes: U.S. & Canadian Phosphorus Agreement; Lake Ontario, Lake Huron, Saginaw Bay

2. Lake Erie: First Inclusion of Sediment Model for Hypolimnetic DO

3. Coastal Waters: Delta, Potomac Estuary

IV. The "Dry Years"

A. 1980 - 1990

1. A General Hiatus in National Focus on Eutrophication

a. National Eutrophication Program ends

b. "The Problem's over." Phosphorus controls in place

c. Toxics, Acid Rain

B. Exceptions: Algal bloom in Potomac; Low DO in Chesapeake & LI Sound

B. 1985 - 1990 Renewal Begins

1. Focus on Coastal Waters, National Estuary Program

2. Nitrogen Reductions: Significant Cost
3. Questions More Complex: Point vs. Non-Point Control

V. Results

A. Predictive Models are Robust: They travel well from water body to water body
B. Some Post Auditing: Predictions Generally Successful
C. Some "Failures": Potomac Estuary 198 3 bloom (sediment phosphorus release); Sacramento San Joaquin 1977 "non-bloom" (benthic filter feeders)

VI. So what?

A. Has productive management been productive?

VII. YES: An Effective Nutrient Management Program Needs:

A. Credible framework for quantitative decision making.
   1. What level of improvement to expect
   2. When will level be achieved
   3. Cost/Benefit analysis if possible
B. Support of
   1. Nutrient Dischargers
   2. Scientific Community
   3. General Public
C. Framework for Quantitative Understanding of Nutrient Response
   1. Reduce Surprises
   2. Understand surprises that do occur

VIII. Conclusions

A. We benefit now from 70 years of observation, prediction, and successful management - Can't stop now.
B. Essential that quantitative predictive frameworks be incorporated in nutrient management programs.

Mimi Dannel, U.S. EPA, Office of Wetlands, Oceans, and Watersheds

I. Is there a need for nutrient management?

A. 1994 305 (b) Report to Congress
   1. Third leading cause of impairments to rivers and streams
   2. Top cause of impairment for lakes
   3. Top cause of impairments for estuaries

B. 303(d) Lists
   1. Among the top three causes of impairments for listed waters

II. How does nutrient management mesh with current initiatives?

A. Watershed Protection Approach
B. Community-based Environmental Protection
C. Trading

III. Endpoints and Assessment Methodologies

A. How can the endpoint be tied back to exisiting State water quality standards?
B. How responsive is the endpoint to improved nutrient control?
C. Can the endpoint response be predicted?
D. Is the endpoint appropriate for volunteer monitoring?

IV. Models

A. Focus on a core set of models for "routine" applications

B. Avoid developing new models if possible
   1. Enhance existing models
      a. Enhancement can mean simplification
      b. Need improved periphyton and macrophyte capabilities

C. When evaluating a model, consider:
   1. Number of input parameters
   2. How input parameters are determined
   3. Comparability of model compexity with probable controls
   4. Cost of model application vs. cost of controls

D. Watch out for "Home Improvement" Syndrome
   1. Keep the focus on the resource, not the tool

V. Models - General Needs

A. Training and Technical Support

B. Guidance Needs
   1. Model-specific guidance
      a. Input parameter determination
         - Monitoring design for measured parameters
         - Estimation techniques for other inputs
      b. Input parameter estimation for projection models
   2. Case studies

Steve Heiskary, Minnesota Pollution Control Agency

I. Minnesota's Approach to Lake Water Quality Criteria Development

A. Steve Heiskary, Minnesota Pollution Control Agency

II. Northern Lakes and Forests

A. Land Use - Forest Dominant

1.	Forest	75%
2.	Water and March	11%
3.	Pasture and Open	7%
4.	Cultivated	5%
5.	Urban	2%

B. Soils - Sand and Silt

C. Lakes - 5,500 Over 10 Acres (46%)

1. Maximum Depth Typically 20-55 Feet
2. Surface Area Typically 100-550 Acres

III. North-Central Hardwood Forests

A. Land Use - No Single Dominant Type

1.	Forest	16%
2.	Water and March	8%
3.	Pasture and Open	20%
4.	Cultivated	50%
5.	Urban	5%

B. Soils - Sand and Silt

C. Lakes - 4,765 Over 10 Acres (40%)

1. Maximum Depth Typically 20-55 Feet
2. Surface Area Typically 150-650 Acres

IV. Western Corn Belt Plains

A. Land Use - Agriculture Dominant

1.	Forest	3%
2.	Water and March	2%
3.	Pasture and Open	10%
4.	Cultivated	83%
5.	Urban	2%

B. Soils - Silt

C. Lakes - 577 Over 10 Acres (5%)

1. Maximum Depth Typically 7-16 Feet
2. Surface Area Typically 350-850 Acres

V. Northern Glaciated Plains

A. Land Use - Agriculture Dominant

1.	Forest	<1%
2.	Water and March	3%
3.	Pasture and Open	11%
4.	Cultivated	84%

5. Urban 1%

B. Soils - Silt

C. Lakes - 855 Over 10 Acres (7%)

1. Maximum Depth Typically 7-13 Feet
2. Surface Area Typically 350-850 Acres

VI. Phosphorus Criteria for Minnesota Lakes - Factors Considered

A. Impacts on Lake Condition

1. Chlorophyll-a
2. Transparency
3. Hypolimnetic Dissolved Oxygen

B. Impacts on Lake Uses

1. Aesthetics
2. Recreation
3. Fisheries (Most Sensitive Uses)

C. Regional Factors

VII. Most Sensitive Lake Uses by Ecoregion and Corresponding Phosphorus Criteria

ECOREGION	MOST SENSITIVE USES	P CRITERIA (µg/l)
Northern Lakes and Forests	Drinking Water Supply	15
	Cold Water Fishery	15
	Primary Contact Recreation and Aesthetics (Full Support)	30
North-Central Hardwood Forests	Drinking Water Supply	30
	Primary Contact Recreation and Aesthetics (Full Support)	40
Western Corn Belt Plains	Drinking Water Supply	40
	Primary Contact Recreation and Aesthetics (Full Support)	40
	(Partial Support)	90
Northern Glaciated Plains	Primary Contact Recreation and Aesthetics (Partial Support)	90

VIII. Western Corn Belt and Northern Glaciated Plains

A. Management Considerations

1. Maintain Current Trophic Status
2. Focus on Lakes <90 ppb P (25th Percentile)

3. Realistic Goals Needed for Restoration

B. Phosphorus Goals

1. Swimmable - Full Support: <40 ppb
   - Partial Support: <90 ppb

   (Keep Frequency of Severe Nuisance Blooms , 25% of Summer)

C. Problems

1. Shallow Lakes
2. High Percent Agricultural Land Use →High P Export
3. Limited Flushing (High Evaporation)

IX. Uses and Application of Criteria

A. Prioritizing and Selecting Projects

B. Developing Water Quality Management Plans

C. Educating (Goal Setting)

D. Guide Enforcement Decisions

E. Guide Interpretations of Nondegradation

X. Approaches for Standards Criteria Development

State/Entity	Approach
VT	User Survey - Trophic Characterization & User Perception
MN	Ecoregion - Reference Lakes, User Perception, Most Sensitive Uses
British Columbia	Water Use - Trophic Characterization & Fish Requirements
ME	Nondegradation - Determine Acceptable Increase in P, Categorize for Protection
OR	Nuisance Phytoplankton - Characterization Average Chl-a, Considers Thermal Stratification
NC	Nutrient Sensitive - Chl-a Criteria for Warm and Cold Water Fisheries, Exceedance Leads to Study
VA	Nutrient-Enriched - Designation Considers chl-a, DO Fluctuation, and TP
TVA	Biological - Index of Biotic Integrity to ID Communities Out of Balance→Management Priority
IJC	Lake-Specific - Detailed Analysis of Biota, Chemical and Loading→Loading Goals, Basinwide Plans

XI. Conclusions

A. Eutrophication Standards Should Be Developed to Protect Lakes and Reservoirs From the Negative Impacts of Cultural Eutrophication.

B. Because Federal Criteria or Guidance is Lacking, State EutrophicationControl Programs May Continue to Be Discretionary in Their Approach.

C. Eutrophication Standards Should Be Developed by the States and Should Be Tailored to Local/Regional Conditions and User Expectations.

D. Lake Monitoring is an Essential Part of Eutrophication Standards Application.

E. Eutrophication Criteria and Standards Can Serve a Vareity of Purposes. Primary Purpose Is to Assist Lake Managers in the Protection and Improvement of Lake Water Quality.

Summary of Overheads Used During Workshop Wrap-up

Day Three Plenary Session Discussion Questions

- Taking into account work that has already been completed in the area of nutrient enrichment assessment, what are the priority nutrient assessment tools EPA should high-light within its national strategy for development, validation, and implementation?
- Of these high priority nutrient assessment tools, which ones have the added benefit of being applicable across waterbodies and/or different geographical areas?
- What should be the relative timing for development, validation, and field testing these high-priority assessment tools?
- What organizations--state, federal, local, consulting, academic, nonprofit, etc.--and which key individuals should be involved in the development, validation, and implementation of these high-priority assessment tools?
- What opportunities are there to leverage ongoing or planned work in the short (1-3 years) and long (3-10 years) timeframes?
- What national and regional programmatic shifts should be encouraged and fostered?
- How feasible are these recommendations taking into consideration the realities of resources and capabilities?
- How do we keep the momentum for getting nutrient enrichment assessment back on the national, regional, state, and local watershed management agendas? What existing networks should we tap into to get the word out and leverage the necessary resources (both people and dollars)?

National Nutrient Assessment Strategy: Next Steps

- Preliminary draft strategy (based on the workshop) and draft workshop proceedings distributed to workshop participants with a request for review (December)
- Revised versions of the issue papers distributed to relevant breakout group members with request for review (January)
- Final workshop proceedings summary published (end of January)
- Final issue papers published (end of February)
- Draft strategy ready for presentation to senior EPA managers (February)
- Presentation of draft strategy to EPA Program Offices, EPA Regional Offices, ORD Laboratories, federal agencies, states, professional societies (February - May)
- Draft final strategy for Science Advisory Board review (May)
- Science Advisory Board review (Summer 1996)
- Publication of subsets of the issue papers in peer reviewed literature (Fall 1996)
- Final EPA Strategy (Fall 1996)
- Continued work with other federal agencies, states, professional societies on a more encompassing strategy--similar substance, more definitive commitments to take responsibility for specific actions, policy changes, resource allocations, etc. (Summer/Fall 1996)

National Nutrient Assessment Strategy: An Overview of Available Endpoints and Assessment Tools

Introduction

Nutrient overenrichment continues to be one of the leading causes of water quality impairment in the United States. The *National Water Quality Inventory 1994 Report to Congress* cites nutrients (nitrogen and phosphorus) as one of the leading causes of water quality impairment in our nation's rivers, lakes, and estuaries. Although nutrients are essential to the health of aquatic ecosystems, excessive nutrient loadings can result in the growth of aquatic weeds and algae, leading to oxygen depletion, increased fish and macroinvertebrate mortality, and other water quality and habitat impairments.

Over the years, the Environmental Protection Agency's Office of Water has issued a number of technical guidance documents and supported the development of water quality simulation models and load estimating models to assess the impacts of urban, rural, and mixed land use activities on receiving waters. In addition, some states currently have water quality standards that incorporate criteria, primarily narrative, aimed at controlling problems associated with overenrichment. However, for state and local agencies to better understand and manage nutrient impacts to surface waters, additional work is required.

EPA established a Nutrient Task Force in July 1993 to gather existing data on nutrient problems and currently available tools. The Task Force recommended that EPA provide additional assistance to states in developing and implementing appropriate nutrient endpoints, assessment methodologies, and models. The first step in carrying out the recommendations of the Task Force was a nutrient assessment workshop, which was held in Washington, D.C., on December 4-6, 1995. The workshop was organized around discussions on four major waterbody types: estuarine and coastal waters; lakes, impoundments/reservoirs, and ponds; rivers and streams; and wetlands. Working papers describing the state of the science, gaps, and user needs in terms of nutrient assessment tools and methodologies for each waterbody type were developed and used as foundations for group discussion.

These issue papers have since been revised and condensed into this one document by incorporating the discussions that were held at the workshop and reviewing additional literature suggested by workshop participants. This condensed paper is intended to represent a brief yet comprehensive overview of the nutrient overenrichment endpoints, assessment methodologies, and modeling tools that have been or could be used in assessing nutrient overenrichment. It should be viewed as a working document that provides an initial synthesis of the available literature, not as an attempt to address all of the detailed issues associated with managing nutrient overenrichment.

This introductory section includes a discussion of issues that overlap across the four waterbody types. These overlapping issues include defining the term *endpoint*, outlining characteristics of effective endpoints, and reviewing simulation loading models that can be applied across multiple types of waterbodies. The following papers then address specific endpoints and models that can be used for each of the four waterbody types.

Defining Endpoints

Key to the development of a national nutrient assessment strategy will be identifying endpoints that can be used at an ecosystem or regional level to describe nutrient enrichment conditions. In the context of this paper, the term *endpoint* is used broadly to encompass any type of target set for nutrient protection. As a simple example, Minnesota has adopted a phosphorus endpoint of 30 µg/L for drinking water supply lakes in its North Central Hardwood Forests ecoregion. The endpoint 30 µg/L therefore separates an "impacted" lake from a "nonimpacted" lake. Because natural conditions, like trophic state, actually represent a continuum of conditions, endpoints are only artificially exacting. Yet they serve a necessary role in management activities because crossing an endpoint typically yields consequences. Once endpoints have been established, actual or predicted measurements can be compared to them and conclusions drawn about the status of nutrient enrichment.

Although water quality endpoints are similar, they are not the same as water quality standards, and the two terms should not be confused. Water quality standards consist of a designated use for a waterbody (such as for swimming, fishing, or protection of aquatic life), as well as either numeric or narrative criteria to protect that use (e.g., a minimum dissolved oxygen concentration of 5 mg/L) (USEPA, 1994). Similarly, nutrient endpoints are used to determine whether a waterbody has become eutrophied to the point that certain uses are impaired. The key distinction is that water quality standards have a legal and regulatory connotation because of their inclusion in state or federal regulations, whereas endpoints do not necessarily have a regulatory connotation.

Using Endpoints

The selection of universal nutrient endpoints is complicated by the fact that given nutrient concentrations can cause different effects in different aquatic systems (Ingman, 1992). Many states approach this problem by using narrative criteria to address the effects of overenrichment (e.g., nutrients should be kept at concentrations that prevent nuisance growths of aquatic plants). A primary advantage of having quantitative nutrient overenrichment endpoints in place would be that violations would prompt and focus management action and remedies could be instituted when available (NALMS, 1992). Quantitative endpoints would allow water resource managers to determine the status of their waterbody and would also provide the justification for taking steps toward improving the resource. For example, if a certain river segment were violating a soluble reactive phosphorus endpoint of 6 µg/L, the regulatory agency would be obligated to initiate some type of management action. This could be as simple as checking to make sure the local wastewater treatment plant is in compliance with its permit or it could be as far-reaching as developing a total maximum daily load (TMDL) that targets point and nonpoint source phosphorus controls.

Given the wide variability in the types of waterbodies that can be affected by nutrient overenrichment, in addition to the climatological differences that might affect waters in different areas, it is generally accepted that endpoints should be regional in nature as opposed to nationwide (Heiskary and Wilson, 1989; NALMS, 1992; Porcella, 1989). This reflects the fact that some waters will not be able to reach the same level of quality as others and also acknowledges that different parts of the country have different uses. In some cases, nutrient endpoints might even need to be site-specific or developed within the context of a total maximum daily load. In general, a balance should be struck between choosing endpoints that reflect specific ecoregion conditions and choosing those which allow practical implementation.

Endpoints can also be used to identify incremental changes as a waterbody moves toward eutrophication-related degradation, thus affording water quality managers the opportunity to take actions that might prevent a system from degrading to a point where beneficial uses are affected. An example of this would be a nutrient concentration endpoint that signals that a lake is approaching the point where plankton blooms and surface scums have historically been a problem. This "early warning" function can be especially beneficial in high-profile, nondegraded systems where the costs of nutrient overenrichment might be especially high.

Quantitative endpoints can be used as indices to determine waterbody status at a point in time, as well as status over a number of years. This use may be important in biennial Clean Water Act section 305(b) reporting, in identifying which lakes, rivers, estuaries, and other waterbodies should be prioritized for restoration, and in evaluating the success of existing programs and watershed management plans.

Endpoints can provide focus for watershed management activities and can serve an important role in communicating waterbody goals to the public. A Secchi depth endpoint of 3 feet, for example, directly reflects something that is valued by the public and is more easily understood than a goal of having "no undesirable or nuisance aquatic life." Since waterbody users will often be called on to voluntarily alter their actions in an attempt to improve the resource, it is important that they understand and support the reasons for doing so. Endpoints can play a key role in this capacity. Additionally, endpoints can play a role in communicating to the public what level of quality can be reasonably expected of a resource given background conditions (Heiskary, 1989).

Characteristics of Effective Endpoints

Effective nutrient overenrichment endpoints should accurately reflect the water quality that is desired in the waterbody. They should be directly related to the designated beneficial uses of the water and should be strict enough to keep a comfortable margin of safety between actual conditions and conditions that have been determined to impair use (whether those uses are swimming, boating, fishing, drinking water supply, or others) (NALMS, 1992). Borrowing from concepts applied in lakes and reservoirs, a number of authors have suggested that impaired use conditions can be estimated by surveying users to determine at what levels they feel their enjoyment of a waterbody is diminished (Heiskary and Wilson, 1989; Heiskary, 1989; Smeltzer and Heiskary, 1990). Survey results can be correlated with simultaneous water quality measurements to establish endpoints at the border between acceptable and unacceptable conditions. If 90 percent of those surveyed agree that their aesthetic enjoyment of a lake is impaired at chlorophyll *a* concentrations exceeding 30 μg/L, this value represents a possible biomass endpoint. The survey approach recognizes that the overall water quality of a waterbody is highly subjective and is best determined by the end user.

Nutrient enrichment endpoints should also be linked in some manner to potential management options. In other words, endpoints should provide managers with some understanding of the nutrient load reductions that will be necessary to correct violations. Most likely this will involve the use of water quality models that can predict water quality parameters based on existing and potential nutrient loads, as well as other model inputs (such as streamflow, residence time, and temperature).

A final consideration in selecting endpoints is the cost associated with monitoring. Water quality parameters that can be measured easily and accurately will obviously be superior to those which require more time and resources to obtain. Issues to keep in mind in this regard include how often the measurements must be taken, who (e.g., experts vs. volunteers) can take the measurements, and the cost of monitoring equipment and laboratory analysis.

Watershed Loading Models

Simulation models have been and will continue to be integral components of water quality planning and pollution control activities, including evaluating the degree of nutrient overenrichment. They serve an essential role in assessing and predicting nutrient overenrichment effects and in evaluating management alternatives. Model selection criteria should include factors such as prediction uncertainty, cost of calibration and testing, meaningful endpoints, appropriate spatial and temporal detail, and simplicity in application and understanding.

Most simulation models can be classified according to whether they focus on pollutant export (loading models) or waterbody response (receiving water models). Because the focus of loading models is on the surrounding watershed and not the waterbody itself, these types of models can be applied to estimate loadings to multiple types of waterbodies A summary of loadings models is therefore presented here, while a discussion of receiving water models specific to the various waterbody types is included in the individual waterbody sections. Information on these models has been summarized from the revised *Compendium of Tools for Watershed Assessment and TMDL Development* (USEPA, in review).

Loading models are used to predict key factors such as flow, nutrient type, concentration, and load delivered to the selected receiving waterbody. Models vary in their capabilities to simulate these processes. For discussion purposes, loading models are divided into categories based on their complexity, operation, time step, and simulation technique. They can be grouped into three categories—simple methods, mid-range models, and detailed models (USEPA, 1992). The three categories of models and types of available models in each category are discussed below.

Simple Loading Methods

The major advantage of simple methods is that they can provide a rapid means of identifying critical loading areas with minimal effort and data requirements. Simple methods are compilations of expert judgment and empirical relationships between physiographic characteristics of the watershed and pollutant export. They can often be applied by using a spreadsheet program or hand-held calculator. Simple methods are often used when data limitations and budget and time constraints preclude the use of complex models. They are used to diagnose nonpoint source pollution problems where limited information is available. Default values provided for these methods are derived from empirical relationships that are evaluated based on regional or site-specific data.

Simple methods provide only rough estimates of sediment and pollutant loadings and have very limited predictive capability. The empiricism contained in the models limits their transferability to regions other than those for which they were developed. Because they often neglect seasonal variability, simple methods might not be adequate to model water quality problems for which loadings of shorter duration are important. They might be sufficient for modeling nutrient loadings to and eutrophication of long-residence-time waterbodies (e.g., lakes, reservoirs). Summaries of the simple methods' capabilities, model components, and input and output data are provided below.

EPA Screening Procedures. The EPA Screening Procedures, developed by the EPA Environmental Research Laboratory in Athens, Georgia, (McElroy et al., 1976; Mills, 1985) include methodologies to calculate nutrient loads from point and nonpoint sources, including atmospheric deposition, for preliminary assessment of water quality. The procedures consist of loading functions and simple empirical expressions relating nonpoint pollutant loads to other readily available parameters. Data required generally include information on land use/land cover, management practices, soils, and topography. An advantage of this approach is the possibility of using readily available data as default values when site-specific information is lacking. Application of these procedures requires minimum personnel training and practically no calibration. However, application to large complex watersheds should be limited to preplanning activities.

The Simple Method. The Simple Method is an empirical approach developed for estimating pollutant export from urban development sites in the Washington, D.C., area (Schueler, 1987). It is used at the site-planning level to predict pollutant loadings under a variety of development scenarios. Its application is limited to small drainage areas of less than a square mile. Pollutant concentrations of phosphorus, nitrogen, chemical oxygen demand, biological oxygen demand (BOD), and metals are calculated from flow-weighted concentration values for new suburban areas, older urban areas, central business districts, hardwood forests, and urban highways. The method relies on the National Urban Runoff Program (NURP) data for default values (USEPA, 1983). A graphical relationship is used to determine the event mean sediment concentration based on readily available information. This method is not coded into a computer program but can be easily implemented with a hand-held calculator.

USGS Regression Approach. The regression approach developed by USGS researchers is based on a statistical description of historic records of storm runoff responses on a watershed level (Tasker and Driver, 1988). This method may be used for rough preliminary calculations of annual pollutant loads when data and time are limiting. Input data include drainage area, percent imperviousness, mean annual rainfall, general land use pattern, and mean minimum monthly temperature. Application of this method provides storm-mean pollutant loads and corresponding confidence intervals. The use of this method as a planning tool at a regional or watershed level might require preliminary calibration and verification with additional, more recent monitoring data.

Simplified Pollutant Yield Approach (SLOSS-PHOSPH). This method uses two simplified loading algorithms to evaluate soil erosion, sedimentation, and phosphorus transport from distributed watershed areas. The SLOSS algorithm provides estimates of sediment yield, whereas the PHOSPH algorithm uses a loading function to evaluate the amount of sediment-bound phosphorus. Application to watershed and subwatershed levels was developed by Tim et al. (1991) based on an integrated approach coupling these algorithms with the Virginia Geographical Information System (VirGIS). The approach was applied to the Nomini Creek Watershed, Westmoreland County, Virginia, to target critical areas of nonpoint source pollution at the subwatershed level (USEPA, 1992c). In this application, analysis was limited to phosphorus loading; however, other pollutants for which input data or default values are available can be modeled in a similar fashion. The approach requires full-scale GIS capability and trained personnel.

Watershed. Watershed is a spreadsheet model developed at the University of Wisconsin to calculate phosphorus loading from point sources, combined sewer overflows (CSOs), septic tanks, rural croplands, and other urban and rural sources. The Watershed program can be used to evaluate the trade-offs between control of point and nonpoint sources (Walker et al, 1989). It uses an annual time step to calculate total nutrient loads and to evaluate the cost-effectiveness of pollution control practices in term of cost per unit load reduction. The program uses a series of worksheets to summarize watershed characteristics and to estimate pollutant loadings for uncontrolled and controlled conditions. Because of the simple formulation describing the various pollutant loading processes, the model can be applied using available default values with minimum calibration effort.

The Federal Highway Administration (FHWA) Model. The FHWA's Office of Engineering and Highway Operations has developed a simple statistical spreadsheet procedure to estimate pollutant loading (including nutrients) and impacts to streams and lakes that receive highway stormwater runoff (Federal Highway Administration, 1990). The procedure uses several worksheets to tabulate site characteristics and other input parameters, as well as to calculate runoff volumes, pollutant loads, and the magnitude and frequency of occurrence of instream pollutant concentrations. The FHWA model uses a set of default values for pollutant event-mean concentrations that depend on traffic volume and the rural or urban setting of the highway's pathway. The Federal Highway Administration uses this method to identify and quantify the constituents of highway runoff and their potential effects on receiving waters and to identify areas that might require controls.

Watershed Management Model (WMM). The Watershed Management Model was developed for the Florida Department of Environmental Regulation for watershed management planning and estimation of watershed pollutant loads (Camp, Dresser, and McKee, 1992). Nitrogen and phosphorus from point and nonpoint sources can be estimated. The model is implemented in the Lotus 1-2-3 spreadsheet environment and can thus calculate standard statistics and produce plots and bar charts of results. The model includes computational components for stream and lake water quality analysis using simple transport and transformation formulations based on travel time. The WMM has been applied to several watershed management projects including the development of a master plan for Jacksonville, Florida, and the Part II estimation of watershed loadings for the NPDES permitting process. It has also been applied in Norfolk County, Virginia; to a Watershed Management Plan for North Carolina; to a wasteload allocation study for Lake Tohopekaliga, near Orlando, Florida; and for water quality planning in Austin, Texas (Pantalion et al., 1995).

Mid-range Loading Models

Mid-range models attempt a compromise between the empiricism of the simple methods and the complexity of detailed mechanistic models. The advantage of mid-range watershed-scale models is that they evaluate nutrient sources and impacts over broad geographic scales and therefore can assist in defining target areas for mitigation programs on a watershed basis. Several mid-range models are designed to interface with geographic information systems, which greatly facilitate parameter estimation (e.g., AGNPS). Greater reliance on site-specific data gives mid-range models a relatively broad range of regional applicability. However, the use of simplifying assumptions can limit the accuracy of their predictions to within about an order of magnitude (Dillaha, 1992) and can restrict their analysis to relative comparisons.

Unlike the simple methods, which are restricted to predictions of annual or storm loads, mid-range tools can be used to assess the seasonal or inter-annual variability of nonpoint source pollutant loadings and to assess long-term water quality trends. Also, they can be used to address land use patterns and landscape configurations in actual watersheds. In addition, they typically require some site-specific data and calibration. Some mid-range models simplify the description of transport processes while emphasizing possible reductions available with controls; others simplify the description of control options and emphasize changes in concentrations as pollutants move through the watershed.

It should be noted that neither the simple nor the mid-range models consider degradation and transformation processes, and few incorporate adequate representation of pollutant transport within and from the watershed. Although their applications might be limited to relative comparisons, however, they can often provide water quality managers with useful information for watershed-scale planning level decisions.

Stormwater Intercept and Treatment Evaluation Model for Analysis and Planning (SITEMAP). SITEMAP, previously distributed under the name NPSMAP, is a dynamic simulation program that predicts daily runoff, nutrient loadings, infiltration, soil moisture, evapotranspiration, and drainage to groundwater. (Omicron Associates, 1990). The model can be used to evaluate user-specified alternative control strategies, and it simulates stream segment load capacities (LCs) in an attempt to develop point source wasteload allocations (WLAs) and nonpoint source load allocations (LAs). Probability distributions for runoff and nutrient loadings can be calculated by the model based on either single-event or continuous simulations. The model can be applied in urban, agricultural, or complex watershed simulations. Although this model requires a minimum calibration effort, it requires moderate effort to prepare input data files. The current version of the program considers only nutrient loading; sediment and other pollutants are not yet incorporated into the program. The model is easily interfaced with GIS (ARC/INFO) to facilitate preparation of land use files. Two examples of where SITEMAP has been applied as a component of a full watershed model are the Tualatin River basin for the Oregon Department of Environmental Quality, and the Fairview Creek watershed for the Metropolitan Service District in Portland, Oregon.

Generalized Watershed Loading Functions (GWLF) Model. The GWLF model was developed at Cornell University to assess the point and nonpoint loadings of nitrogen and phosphorus from urban and agricultural watersheds, including septic systems, and to evaluate the effectiveness of certain land use management practices (Haith et al., 1992). One advantage of this model is that it was written with the express purpose of requiring no calibration, making extensive use of default parameters. The GWLF model includes rainfall/runoff and erosion and sediment generation components, as well as total and dissolved nitrogen and phosphorus loadings. The GWLF model uses daily time steps and allows analysis of annual and seasonal time series. The model also uses simple transport routing, based on the delivery ratio concept. In addition, simulation results can be used to identify and rank pollution sources and evaluate basinwide management programs and land use changes. The most recent update of the model incorporates a septic (on-site wastewater disposal) system component. The model also includes several reporting and graphical representations of simulation output to aid in interpretation of the results. This model was successfully tested on a medium-size watershed in New York (Haith and Shoemaker, 1987). A version of the model with an enhanced user interface and linkages to national databases, WSM (Watershed Screening Model), has recently become available and is distributed with EPA-OWOW's Watershed Screening and Targeting Tool (WSTT).

Urban Catchment Model (P8-UCM). The P8-UCM program was developed for the Narragansett Bay Project to simulate the generation and transport of stormwater runoff pollutants in small urban catchments and to assess impacts of development on water quality, with minimum site-specific data. It includes several routines for evaluating the expected removal efficiency for particular site plans, selecting or siting best management practices (BMPs) necessary to achieve a specified level of pollutant removal, and comparing the relative changes in pollutant loads as a watershed develops (Palmstrom and Walker, 1990). Default input parameters can be derived from NURP data and are available as a function of land use, land cover, and soil properties. However, without calibration, the use of model results should be limited to relative comparisons. Spreadsheet-like menus and on-line help documentation make extensive user interface possible. On-screen graphical representations of output are developed for a better interpretation of simulation results. The model also includes components for performing monthly or cumulative frequency distributions for flows and pollutant loadings.

Automated Q-ILLUDAS (AUTO-QI). AUTO-QI is a watershed model developed by the Illinois State Water Survey to perform continuous simulations of stormwater runoff from pervious and impervious urban lands (Terstriep et al., 1990). It also allows the examination of storm events or storm sequence impacts on receiving water. Critical events are also identified by the model. However, hourly weather input data are required. Several pollutants, including nutrients, chemical oxygen demand, metals, and bacteria, can be analyzed simultaneously. This model also includes a component to evaluate the relative effectiveness of best management practices. An updated version of AUTO-QI, with an improved user interface and linkage to a geographic information system (ARC/INFO on PRIME computer), has been completed by the Illinois State Water Survey. This interface is provided to generate the necessary input files related to land use, soils, and control measures. AUTO-QI was verified on the Boneyard Creek in Champaign, Illinois, and applied to the Calumet and Little Rivers to determine annual pollutant loadings.

Agricultural Nonpoint Source Pollution Model (AGNPS). Developed by the U.S. Department of Agriculture's (USDA) Agricultural Research Service, AGNPS addresses concerns related to the potential impacts of point and nonpoint source pollution on water quality (Young et al., 1989). It was designed to quantitatively estimate pollution loads from agricultural watersheds and to assess the relative effects of alternative management programs. The model simulates surface water runoff, as well as nutrient and sediment constituents associated with agricultural nonpoint sources and point sources. It also accounts for point sources like treatment plants, and stream bank or gully erosion. The available version of AGNPS is event-based; however, a continuous version is under development (Needham and Young, 1993). The AGNPS grid system allows the model to be connected to other software such as geographic information systems (GIS) and digital elevation models (DEM). This connectivity can facilitate the development of a number of the model's input parameters. Two new terrain-enhanced versions of the model— AGNPS-C, a contour-based version, and AGNPS-G, a grid-based version—have been developed to generate the grid network and the required topographic parameters (Panuska et al., 1991). Vieux and Needham (1993) describe a GIS-based analysis of the sensitivity of AGNPS predictions to grid-cell size. Engel et al. (1993) present GRASS-based tools to assist with the preparation of model inputs and visualization and analysis of model results. Tim and Jolly (1994) used AGNPS with ARC/INFO to evaluate the effectiveness of several alternative management strategies in reducing sediment pollution in a 417-ha watershed in southern Iowa. The model also includes enhanced graphical representations of input and output information.

Detailed Loading Models

Detailed models best represent the current understanding of watershed processes affecting pollution generation. If properly applied and calibrated, detailed models can provide relatively accurate predictions of variable flows and water quality at numerous points within a watershed. The additional precision they provide, however, comes at the expense of considerable time and resource expenditure.

Detailed models incorporate the manner in which watershed processes change over time in a continuous fashion rather than relying on simplified terms for rates of change (Addiscott and Wagenet, 1985). They tend to require rate parameters for flow velocities and nutrient accumulation, settling, and decay instead of capacity terms. The length of time steps is variable and depends on the stability of numerical solutions as well as the response time for the system (Nix, 1991). Algorithms in detailed models more closely simulate the physical processes of infiltration, runoff, pollutant accumulation, instream effects, and groundwater/surface water interaction. The input and output of detailed models also have greater spatial and temporal resolution. Moreover, the manner in which physical characteristics and processes differ over space is incorporated within the governing equations (Nix, 1991). Linkage to biological modeling is possible because of the comprehensive nature of continuous simulation models. In addition, detailed hydrologic simulations can be used to design potential control actions.

Detailed models use small time steps to allow for continuous and storm event simulations. However, input data file preparation and calibration require professional training and adequate resources. Some of these models (e.g., STORM, SWMM, ANSWERS) were developed not only to support planning-level evaluations but also to provide design criteria for pollution control practices. If appropriately applied, state-of-the-art models like HSPF and SWMM can provide accurate estimations of pollutant loads and the expected impacts on water quality. New interfaces developed for HSPF and SWMM, and links with GISs, can facilitate the use of complex models for environmental decision-making. However, their added accuracy might not always justify the amount of effort and resources they require. Application of such detailed models is more cost-effective when used to address complex situations or objectives.

Storage, Treatment, Overflow Runoff Model (STORM). STORM is a U.S. Army Corps of Engineers (COE) model developed for continuous simulation of runoff quantity and quality, including sediments and several conservative pollutants. STORM has been widely used for planning and evaluation of the tradeoff between treatment and storage control options for CSOs and was primarily designed for modeling stormwater runoff from urban areas. It requires relatively moderate to high calibration and input data. STORM was initially developed for mainframe computer usage; however, several versions have been adapted by various individual consultants for use on microcomputers. The model has been applied recently to water quality planning in the City of Austin, Texas (Pantalion et al., 1995).

Areal Nonpoint Source Watershed Environment Response Simulation Model (ANSWERS). ANSWERS is a comprehensive model developed at the University of Georgia to evaluate the effects of land use, management schemes, and conservation practices or structures on the quantity and quality of water from both agricultural and nonagricultural watersheds (Beasley, 1986). The distributed structure of this model allows for a better analysis of the spatial as well as temporal variability of pollution sources and loads. It was initially developed on a storm event basis to enhance the physical description of erosion and sediment transport processes. Data file preparation for the ANSWERS program is rather complex and requires mainframe capabilities, especially when dealing with large watersheds. The output routines are quite flexible; results may be obtained in several tabular and graphical forms. The program has been used to evaluate management practices for agricultural watersheds and construction sites in Indiana. It has been combined with extensive monitoring programs to evaluate the relative importance of point and nonpoint source contributions to Saginaw Bay. This application involved the computation of unit area loadings under different land use scenarios for evaluation of the trade-offs between LAs and WLAs. Recent model revisions include improvements to the nutrient transport and transformation subroutines (Dillaha et al., 1988). Bouraoui et al. (1993) describe the development of a continuous version of the model.

Multi-event urban runoff quality model (DR3M-QUAL). DR3M is a watershed model for routing storm runoff through a branched system of pipes and/or natural channels using rainfall as input. The model provides detailed simulation of storm-runoff periods selected by the user and a daily soil-moisture accounting between storms. Kinematic wave theory is used for routing flows over contributing overland-flow areas and through the channel network. Storm hydrographs may be saved for input to DR3M-QUAL, which simulates the quality of surface runoff from urban watersheds. The model simulates impervious areas, pervious area, and precipitation contributions to runoff quality, as well as the effects of street sweeping and/or detention storage. Variations of runoff quality are simulated for user-specified storm-runoff periods. Between these storms, a daily accounting of the accumulation and washoff of water-quality constituents on effective impervious areas is maintained. Input to the model includes the storm hydrographs, usually from DR3M. The program has been extensively reviewed within the USGS and applied to several urban modeling studies (Brabets, 1986; Lindner-Lunsford and Ellis, 1987; Guay, 1990).

Simulation for Water Resources in Rural Basins - Water Quality (SWRRBWQ). The SWRRBWQ model was adapted from the field-scale CREAMS model by USDA to simulate hydrologic, sedimentation, nutrient, and pesticide movement in large, complex rural watersheds (Arnold et al., 1989). SWRRBWQ uses a daily time step to evaluate the effect of management decisions on water, sediment yields, and pollutant loadings. The model is useful for estimation of the order of magnitude of pollutant loadings from relatively small watersheds or watersheds with fairly uniform properties. Input requirements are relatively high, and experienced personnel are required for successful simulations. SWRRBWQ was used by the National Oceanic and Atmospheric Administration (NOAA) to evaluate pollutant loadings to coastal estuaries and embayments as part of its national Coastal Pollution Discharge Inventory. The model has been run for all major estuaries on the east coast, west coast, and Gulf coast for a wide range of pollutants (Donigian and Huber, 1991). Although SWRRBWQ is no longer under active development, the technology is being incorporated into the Soil and Water Assessment Tool (SWAT) as part of the Hydrologic Unit Model for the United States (HUMUS) project at Temple, Texas (Arnold et al., 1993; Srinivasan and Arnold, 1994). EPA's Office of Science and Technology (OST) has recently developed a Microsoft Windows-based interface for SWRRBWQ to allow convenient access to temperature, precipitation, and soil data files.

Storm Water Management Model (SWMM). SWMM is a comprehensive watershed-scale model developed by EPA (Huber and Dickinson, 1988). It was initially developed to address urban stormwater and assist in storm-event analysis and derivation of design criteria for structural control of urban stormwater pollution, but it was later upgraded to allow continuous simulation and application to complex watersheds and land uses. SWMM can be used to model several types of pollutants provided that input data are available. Recent versions of the model can be used for either continuous or storm event simulation with user-specified variable time steps. The model is relatively data-intensive and requires special effort for validation and calibration. Its application in detailed studies of complex watersheds might require a team effort and highly trained personnel. In addition to developing comprehensive watershed-scale planning, typical uses of SWMM include predicting combined sewer overflows, assessing the effectiveness of BMPs, providing input to short-time-increment dynamic receiving water quality models, and interpreting receiving water quality monitoring data (Donigian and Huber, 1991). EPA's Office of Science and Technology distributes a Microsoft Windows interface for SWMM that makes the model more accessible. A postprocessor allows tabular and graphical display of model results and has a special section to help in model calibration.

The Hydrological Simulation Program - FORTRAN (HSPF). HSPF is a comprehensive package developed by EPA for simulating water quantity and quality for a wide range of organic and inorganic pollutants from agricultural watersheds (Bicknell et al., 1993). The model uses continuous simulations of water balance and pollutant generation, transformation, and transport. Time series of the runoff flow rate, sediment yield, and user-specified pollutant concentrations can be generated at any point in the watershed. The model also includes instream quality components for nutrient fate and transport, biochemical oxygen demand (BOD), dissolved oxygen (DO), pH, phytoplankton, zooplankton, and benthic algae. Statistical features are incorporated into the model to allow for frequency-duration analysis of specific output parameters. Data requirements for HSPF are extensive, and calibration and verification are recommended. The program is maintained on IBM microcomputers and DEC/VAX systems. Because of its comprehensive nature, the HSPF model requires highly trained personnel. It is recommended that its application to real case studies be carried out as a team effort. The model has been extensively used for both screening-level and detailed analyses. The Chesapeake Bay Program is using HSPF to model total watershed contributions of flow, sediment, nutrients, and associated constituents to the tidal region of the Bay (Donigian et al., 1990; Donigian and Patwardhan, 1992). Moore et al. (1992) describe an application to model BMP effects on a Tennessee watershed. Scheckenberger and Kennedy (1994) discuss how HSPF can be used in subwatershed planning. Ball et al. (1993) describe an application of HSPF in Australia. Lumb et al. (1990) describe an interactive program for data management and analyses that can be effectively used with HS and Lumb and Kittle (1993) present an expert system that can be used for calibration and application of HSPF.

Lakes and Reservoirs

Lakes and reservoirs are among the water resources identified as part of EPA's National Nutrient Assessment Strategy. They are especially vulnerable to the effects of nutrient overenrichment because of their long residence time. Eutrophication in lakes and reservoirs can manifest itself in many ways, including as plankton surface scums, excessive macrophyte growths, seasonal blue-green algal blooms, and decreased dissolved oxygen concentrations. Treated drinking water taken from eutrophic lakes and reservoirs can also potentially be contaminated with disinfection by-products such as trihalomethanes, posing a threat to human health. Through the biennial *National Water Quality Inventory Report to Congress*, state officials consistently identify nutrients as the cause of more lake and reservoir impairments than any other single pollutant (USEPA, 1992, 1994a).

In some respects, the body of science regarding the causes and effects of eutrophication in lakes and reservoirs is more mature than it is for other waterbodies. A great deal of research has been conducted in this area dating back to the work of Vollenweider in the mid- to late-1960s, and the linkage between nutrient loading and the eutrophication of lakes and reservoirs has been strongly established in the literature (e.g., Vollenweider, 1968; Hutchinson, 1973; Wetzel, 1983). Many lake and reservoir managers have taken advantage of this by applying the existing base of knowledge to the management of their individual lakes. Nevertheless, there is no widely accepted approach to managing nutrient overenrichment and many communities continue to suffer the ill effects of human-induced eutrophication.

The relationship between nutrient loading and eutrophication is complicated by a variety of physical, chemical, and biological factors that make each lake, reservoir, and watershed unique. In some waterbodies, for example, increased sedimentation (and corresponding loss of volume) of nutrient-rich soils and partially decomposed plant material is an important part of the eutrophication process (Cooke and Carlson, 1989). Internal loading of nutrients from the sediment may also drive biological production cycles in many lakes (e.g., Cooke et al., 1993; van der Molen and Boers, 1994), and the growth of nuisance vegetation may often be controlled by factors such as the amount of available light, grazing populations, or water temperature. Resource managers must keep all of these factors in mind as they attempt to formulate a strategy to address nutrient overenrichment.

State of the Science—Endpoints

A review of the available literature suggests that a number of water quality parameters have been used to characterize the nutrient enrichment of lakes and reservoirs. In some instances these parameters have been used on a statewide basis; in other instances they have been specifically designed for individual lakes. Still other parameters have simply been proposed in the literature.

Carlson (1984) points out the importance of distinguishing between causal factor parameters (e.g., phosphorus) and biological response parameters (e.g., algal biomass). Endpoints of both types of factors have been used to evaluate, classify, and set standards for lakes and reservoirs. Other parameters include hypolimnetic oxygen depletion rate and odor and taste indicators. Presented below are short descriptions of these parameters.

Causal Factor Parameters

Total Phosphorus. Perhaps the most common current nutrient enrichment endpoint in lakes and reservoirs is total phosphorus concentration. Total phosphorus standards have been used to address eutrophication in the Great Lakes, Lake Champlain, Lake Okeechobee, and a number of smaller watersheds (NALMS, 1992; Smeltzer, 1992; Goldstein and Ritter, 1993; Heiskary and Walker, 1988). These standards are typically based on summer ambient levels and can range anywhere from 7 to 200 µg/L (MWCG, 1982). The use of spring total phosphorus has also been shown to be an effective early warning predictor of summer algal levels (Jones et al., 1979) and might prove to be a highly cost-effective endpoint. Several studies have shown it to be an effective predictor of summer mean and summer maximum chlorophyll *a* in both temperate and subarctic lakes (Ostrofsky and Rigler, 1987).

Phosphorus has been commonly used as an enrichment endpoint because of its direct correlation with the associated negative effects of eutrophication. It is most often the limiting nutrient that controls the growth of nuisance algae and macrophytes and can therefore be used to predict their growth. Phosphorus is also relatively inexpensive to monitor and is already incorporated into most monitoring programs. Most simulation models incorporate phosphorus concentrations either as an input variable or as part of model output.

One concern with using total phosphorus as an endpoint is that the nutrient itself is not what hinders water use (Walker, 1979). There are those who would argue that only use-related parameters (such as algal biomass or fish kills) should serve as endpoints. Another concern arises in situations where the limiting nutrient is unclear or might change by season. A phosphorus endpoint in a nitrogen limited lake, for instance, would be a misdirected goal.

Total Nitrogen Concentration. Several states reported using total nitrogen concentration as a eutrophication-based water quality standard in 1987 (NALMS, 1988). However, since freshwater systems are more often phosphorus, rather than nitrogen-limited, relatively little attention has been focused on using this parameter as a eutrophication endpoint. This is reflected by that fact that it has not been as extensively studied as has phosphorus and by the lack of sophisticated models that adequately account for its effect on the eutrophication process. Nitrogen might be an appropriate endpoint only for those highly productive lakes and reservoirs that are moving toward nitrogen limitation.

Nitrogen:Phosphorus Ratio. For those lakes and reservoirs which might be approaching nitrogen limitation, the nitrogen-to-phosphorus ratio, both of loadings and of standing concentrations, can provide important insight regarding which nutrient should be controlled. Nitrogen and phosphorus are taken up by algae in an approximately constant ratio of 16 atoms of nitrogen per 1 atom of phosphorus, or 7.2:1 by weight. The practical range around this number is 10-20 (Chiaudani and Vighi, 1974; Boynton et al., 1995; Thomann and Mueller, 1987). Waters with relative concentrations, then, of <10:1 nitrogen to phosphorus might have insufficient nitrogen for algal uptake relative to the available phosphorus. Nitrogen availability limits plant growth in these situations. When nitrogen:phosphorus is >20, the converse may be true: phosphorus becomes the limiting nutrient for plant production. An assessment of potential limitation at intermediate values is problematic. Complicating factors involve other limitations (e.g., light), the presence of nitrogen-fixing algae, and the availability of the forms of nitrogen and phosphorus present. Bioassays have been used to assess nutrient limitation; for fresh waters, a standardized test has been developed.

Biological Response Parameters

Although macrophytes play a dominant role in many lakes and reservoirs, algal biomass in the open water is traditionally the parameter used to characterize nutrient enrichment. Due to the difficulty in analyzing biomass, limnologists have turned to two popular biomass surrogates, chlorophyll *a* concentration and Secchi disk depth.

Chlorophyll *a*. Chlorophyll *a* is the dominant pigment in algal cells and is fairly easy to measure. Even though chlorophyll concentration varies from species to species and fluctuates according to light intensity, it still remains a valuable surrogate for algal biomass (Carlson, 1980; Watson, et al., 1992). Several states have adopted chlorophyll *a* concentrations as standards for lake quality. Oregon has set an endpoint of 10 μg/L for natural lakes that thermally stratify and 15 μg/L for natural lakes that do not thermally stratify (NALMS, 1992). Similarly, North Carolina uses a standard of 40 μg/L for warm waters and 15 μg/L for cold waters (NALMS, 1992). On the regional level, Raschke (1994) has proposed a mean growing season limit of 15 μg/L for water supply impoundments in the southeastern United States and a value of 25 μg/L for waterbodies primarily used for other purposes (e.g., viewing pleasure, safe swimming, fishing, boating).

Chlorophyll *a* is desirable as an endpoint because it closely reflects use impairment (often it *is* the impairment) and because it can usually be closely correlated to loading conditions. Both seasonal mean and instantaneous maximum concentrations can be used to determine violations, and most monitoring programs already include measurements for chlorophyll.

Chlorophyll might not be an appropriate endpoint in instances where use impairment is more closely related to excessive macrophyte or attached algae growth. There also might be instances where the relationship between nutrient concentrations and chlorphyll response is highly variable and difficult to predict. Laws and Chalup (1990), for example, have shown that the correlation between growth rate and chlorophyll *a*:carbon is positive under nutrient-limited conditions and negative under nutrient-saturated (light-limited) conditions. This might complicate the ability to make management decisions based on targeted chlorophyll concentrations. Other potential drawbacks associated with using chlorophyll include the lack of a standardized testing methodology and its high spatial and temporal variability.

Secchi Depth. Due to its simplicity and low cost, Secchi depth is the most widely used surrogate for estimating algal biomass and, subsequently, trophic state (Michaud, 1991). It is, however, a much more indirect measure and subject to interferences from a variety of sources. Secchi depth has been shown to be highly correlated to chlorophyll *a* concentrations (Rast and Lee, 1978) and is a particularly important measure because water clarity is easily perceived by the public (Thomann and Mueller, 1977). However, lake and reservoir managers must still be able to relate desired Secchi depths with nutrient loads. Secchi depth might be a reliable indicator of the trophic state of a waterbody provided that water clarity is primarily dependent on algal biomass (i.e., the amounts of inorganic turbidity and color present in the water column are low).

Six states reported using Secchi depth as a eutrophication-based water quality standard in 1988 (NALMS, 1988), and Raschke (1993) reports on using a mean growing season Secchi depth endpoint of >1.5 meters for water supply impoundments in the southeastern Piedmont. For impoundments used for fishing and swimming, >1 meter is considered acceptable. Secchi depth was also included as one of the parameters to be used in the surface waters component of the Environmental Monitoring and Assessment Program (EMAP) (Whittier and Paulsen, 1992).

Other biological response factors that have potential for use as water quality endpoints include the following:

Macrophyte Coverage or Density. Newbry et al. (1981) recommend using percent macrophyte coverage as a potential nutrient enrichment endpoint. Specifically, the endpoint would be the percentage of the waterbody having a depth of 2 meters or less that is impaired by macrophyte growth during peak recreation use. Other researchers have suggested a similar approach, but a somewhat deeper cutoff point (Porcella et al., 1980). This potential endpoint might be particularly appropriate for those shallow lakes and reservoirs where users are aware of and sensitive to changes in the abundance and distribution of macrophytes (e.g., the southeastern United States). One confounding issue related to macrophyte density is the positive relationship between increasing water clarity and the extent of macrophyte beds (Quinn, 1991). A limitation involved with using macrophyte measurements as endpoints is the difficulty involved with predicting whether a system will be dominated by macrophytes or plankton.

Fish Yields/Quality. Lee and Jones (1991) discuss the effects of eutrophication on fisheries and also introduce an approach for estimating the fish yield that can be associated with changing phosphorus loadings. Fish yields were also included as indicators in the surface waters component of the Environmental Monitoring and Assessment Program (Whittier and Paulsen, 1992). This endpoint is obviously attractive for lakes or reservoirs where fishing is a primary use.

Biological Indicators. A number of states have used biological indicators in setting stream and river standards, although it does not appear that any have developed indices specifically designed for lakes or reservoirs (Hughes et al., 1992). Proposed biological indicators have attempted to incorporate information on fish, benthic invertebrates, zooplankton assemblages, algae, macrophytes, etc. An advantage of using biological indicators is that they are not as subject to temporal variability as are chemical pollutants (NALMS, 1992). However, a disadvantage associated with biological indicators is that they might reflect impairments not necessarily caused by cultural eutrophication (i.e., they are better at indicating change itself than they are at providing information on the cause of change (Hughes et al., 1992).

Sedimentary Diatom Assemblages. The emerging field of paleolimnology has shown that sedimentary diatom assemblages can be used to determine baseline trophic conditions and natural variability in lakes and reservoirs (Dixit et al., 1992). In addition to providing information on historical conditions, these algal microfossils can also be used to detect water quality trends occurring during more recent years; they could therefore serve as potential early warning indicators of eutrophication. Sedimentary diatom assemblages were one of the parameters measured as part of the Northeast Lakes Pilot of the Environmental Monitoring and Assessment Program (Larsen et al., 1991; Hughes et al., 1992).

Other Factors

Dissolved Oxygen Concentration. Dissolved oxygen concentration could be used as a eutrophication endpoint in waterbodies where a primary concern is support of aquatic life. It could be measured as end of summer concentration in the hypolimnion, the date at which a lake or reservoir becomes anoxic, or the length of time a system is anoxic. Problems associated with using dissolved oxygen as an endpoint are related to the high spatial and temporal variability in its measurement and the fact that some reservoirs (especially southeastern ones) can display rapid changes between anoxia and oxia.

Others. Other water quality parameters that have at times been mentioned as potential eutrophication endpoints include extent of submerged aquatic vegetation (USEPA, 1994b), odor and taste indicators (Hughes and Paulsen, 1990; Seligman et al., 1992), and treatment plant chlorine demand (Lee et al., 1995a). These potential endpoints might prove to be most appropriate where they relate to specific considerations in individual lakes and reservoirs. They might be less applicable across state or ecoregion boundaries than the endpoints discussed previously.

State of the Science—Models

A discussion of the issues involved with assessing nutrient overenrichment in lakes and reservoirs would be incomplete without addressing the use of water quality models. As mentioned in the introduction to the issue papers, numerous watershed loading models are available for the assessment of nutrient overenrichment, many of which can be applied across multiple waterbody types. A summary is provided here of some of the receiving water models that are especially useful for predicting and assessing the eutrophication process in lakes and reservoirs. These summaries were prepared based on information from the revised *Compendium of Tools for Watershed Assessment and TMDL Development* (USEPA, in review). References pertaining to each of these models are listed in a separate section at the end of this paper.

EUTROMOD. EUTROMOD is a spreadsheet-based watershed and lake modeling procedure developed for eutrophication management, with an emphasis on uncertainty analysis. The model estimates nutrient loading, various trophic state parameters, and trihalomethane concentration using data on land use, pollutant concentrations, and lake characteristics. The model was developed using empirical data from the USEPA's national eutrophication survey, with trophic state models utilized to relate phosphorus and nitrogen loading to in-lake nutrient concentrations. The phosphorus and nitrogen concentrations were then related to maximum chlorophyll level, Secchi depth, dominant algal species, hypolimnetic dissolved oxygen status, and trihalomethane concentration. EUTROMOD allows for uncertainty analysis by considering the error in regression equations employed, and utilizing an annual mean precipitation and coefficient of variation to account for hydrologic variability. EUTROMOD is limited in its application because it is designed for watersheds in the Southeast and because it provides only predictions of growing season averages. EUTROMOD is distributed by the North American Lake Management Society (NALMS).

PHOSMOD. PHOSMOD uses a modeling framework described by Chapra and Canale (1991) for assessing the impact of phosphorus loading on stratified lakes. A total phosphorus budget for the water layer is developed with inputs from external loading, recycling from the sediments, and considering losses due to flushing and settling. The sediment-to-water recycling is dependent on the levels of sediment total phosphorus and hypolimnetic oxygen, with the concentration of the latter estimated with a semi-empirical model. PHOSMOD can be used to make daily or seasonal analyses and was developed to assess long-term dynamic trends; output includes tabular and graphical output of lake total phosphorus, percentage of total phosphorus in sediment, hypolimnetic dissolved oxygen concentrations, and days of anoxia. It also is distributed by NALMS.

BATHTUB. BATHTUB applies a series of empirical eutrophication models to morphologically complex lakes and reservoirs. The program performs steady-state water and nutrient balance calculations in a spatially segmented hydraulic network that accounts for advective and diffusive transport, and nutrient sedimentation. Eutrophication-related water quality conditions (total phosphorus, total nitrogen, chlorophyll *a*, transparency, and hypolimnetic oxygen depletion) are predicted using empirical relationships derived from assessment of reservoir data (Walker, 1985; 1986). Applications of BATHTUB are limited to steady-state evaluation of relationships between nutrient-loading, transparency and hydrology, and eutrophication responses. BATHTUB has been cited as an effective tool for lake and reservoir water quality assessment and management, particularly where data are limited (Ernst et al., 1994).

WASP5. WASP5 is a general-purpose modeling system for assessing the fate and transport of conventional and toxic pollutants in surface waterbodies. Its EUTRO5 submodel is designed to address eutrophication processes and has been used in a wide range of regulatory and water quality management applications. The model may be applied to most waterbodies in one, two, or three dimensions and can be used to predict time-varying concentrations of water quality constituents. It might have some limited applications in lakes due to a lack of internal temperature simulation. The model reports a set of parameters, including dissolved oxygen concentration, carbonaceous biochemical oxygen demand (BOD), ultimate BOD, phytoplankton, carbon, chlorophyll *a*, total nitrogen, total inorganic nitrogen, ammonia, nitrate, organic nitrogen, total inorganic nitrogen, organic phosphorus, and inorganic phosphorus. Although zooplankton dynamics are not simulated in EUTRO5, their effect can be described by user-specified forcing functions.

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

Several limitations are associated with using CE-QUAL-W2 to model nutrient overenrichment in lakes and reservoirs. Because the model assumes lateral homogeneity, it is best suited for relatively long and narrow waterbodies that exhibit strong longitudinal and vertical water quality gradients. It might be inappropriate for large waterbodies. The model also has only one algal compartment, and algal succession, zooplankton, and macrophytes cannot be modeled.

CE-QUAL-ICM. CE-QUAL-ICM incorporates detailed algorithms for water quality kinetics and can be applied to most waterbodies in one, two, or three dimensions. Interactions among input variables are described in 80 partial differential equations that employ more than 40 parameters (Cerco and Cole, 1993). Model outputs include temperature; inorganic suspended solids; diatoms; blue-green algae (and other phytoplankton); dissolved, labile and refractory components of particulate organic carbon; organic nitrogen; organic phosphorus; ammonium; nitrate and nitrite; total phosphate; and dissolved oxygen. Although the model has full capabilities to simulate state-of-the-art water quality kinetics, it is potentially limited by available data for calibration and verification. In addition, the model might require significant technical expertise in aquatic biology and chemistry to be used appropriately.

Rivers and Streams

The beneficial uses supported by the Nation's rivers and streams continue to be threatened by the damaging effects of nutrient overenrichment. Nitrogen and phosphorus loadings originating from wastewater treatment plants, agricultural fields, industrial discharges and other sources are contributing to excessive plant growths in rivers and streams of all sizes and in all regions. These plant growths can clog water intake pipes, use up precious dissolved oxygen when they decay, and interfere with fishing and other recreational uses. As an indication of the extent of this problem, the most recent *National Water Quality Inventory Report to Congress* stated that nutrients were the second leading cause of impairment in assessed rivers and streams, behind only siltation (USEPA, 1994a). State officials reported that nutrients had negatively impacted 37 percent of all the assessed river and stream miles in the country.

Although the damaging effects of nutrient overenrichment have been recognized for some time, many states and local communities continue to struggle with developing an effective strategy for addressing the issue. Rivers and streams are complicated systems, and it is difficult to accurately assess the nature of nutrient loadings as well as their impact on the growth of nuisance vegetation. Managers are also handicapped by their need to consider the role rivers and streams play as conduits through which nutrients are carried into and impact other waterbodies, such as lakes, reservoirs, and estuaries.

Background

In rivers and streams, eutrophication can result from an increase in the supply of inorganic nutrients such as nitrates, ammonia, and soluble reactive phosphorus. For the purposes of this paper, the focus will be on nutrient overenrichment that occurs as a consequence of human activity and results in the impairment of river and stream beneficial uses (e.g., fishing, swimming, and drinking water supply). To better understand the linkage between nutrient loading and impaired uses, one must first realize there are various types of rivers and streams that can be affected by eutrophication.

Rivers and streams represent a diverse set of waterbody systems, ranging from the deep, slow-flowing lower Mississippi to the cascading, coldwater mountain streams of the Rockies. A number of different concepts have been used to describe this diversity (Johnson et al., 1995). For example, the river-continuum concept specifies that rivers and streams can be categorized according to stream order. A first-order stream has no tributaries, whereas a second-order stream is formed when two first order streams join. When two second-order streams meet, they form a third-order stream and so on (Johnson et al., 1995). Within this concept, stream orders of first to third are considered small streams, fourth to sixth are considered medium-sized rivers, and those greater than sixth order are considered large rivers.

The significance of the stream order concept is that the structure and function of small stream biotic communities are very much different from those of large rivers (Johnson et al., 1995). For example, in small forest streams the existence of a canopy reduces the amount of light available for photosynthesis. This means that the primary energy source is likely to be terrestrial leaf litter and the dominant invertebrate species will be shredders and collectors. Conversely, there is typically more light available in mid-size rivers; the main energy source is autochthonous production[1], and the dominant invertebrate groups are grazers that feed by scraping algae and microbes off rocks (Johnson et al., 1995). These grazers can have a profound impact on whether nutrient-enriched benthic algae will grow to nuisance levels (Walton et al., 1995).

In addition to the continuum concept, rivers and streams can be described according to their flow characteristics. A high-gradient stream is one that is located along relatively steep slopes, displays relatively high velocities, and is more likely to have rapids, whitewater, and similar features; low-gradient streams, on the other hand, follow more gradual terrain and therefore have slower currents and longer residence times. The distinct physical conditions that exist in these two types of systems have a significant impact on eutrophication conditions. For example, research has shown that the growth of benthic algae can be highly dependent on stream current velocity (Horner et al., 1990; Welch, 1989; see discussion below).

Another way to distinguish different types of rivers and streams, and the one that is perhaps of most significance for the purposes of this paper, is by their dominant plant types. Plankton-dominated systems are those with large populations of freely floating microscopic plants such as diatoms, blue-green algae, and flagellates (Quinn, 1991). Periphyton (or benthic algae)-dominated systems, on the other hand, are impaired by plants that grow attached to the streambed or on solid objects like logs. One of the most common groups of benthic algae are the filamentous greens, such as *Cladophora*. The nuisance effects of both plankton and periphyton are described in Tables 1 and 2 (adapted from Quinn, 1991).

The processes governing the growth of nuisance plants (especially periphyton) in rivers and streams are not nearly as straightforward as they are for other waterbodies. In lakes, for example, the biomass of plankton per unit volume of water is proportional to the initial mass of limiting nutrient per unit volume (Welch et al., 1989). This relationship, which has been shown in the laboratory and

[1]Autochthonous production refers to production originating in the river itself (as opposed to being from terrestrial sources).

Table 1. Potential nuisance effects of excessive phytoplankton growth with possible nutrient enrichment endpoints (from Quinn, 1991).

Water Uses	Nuisance Effects	Possible Endpoints
Water Supply	• Taste and odor problems • Production of toxins (e.g., by blue-green algae) • Blockage of intake screens and filters • Disruption of flocculation and chlorination processes in water treatment plants.	• Odor and taste indicators • Cost of treatment
Aesthetic appeal	• Reduced clarity and altered color • Surface scums (including "red tides") and floating mats	• Secchi depth
Recreation	• Boating, swimming, water skiing and other water-based recreation restricted or degraded.	• Secchi depth • Chlorophyll a
Aquaculture	• Fish kills (via toxicity) • Shellfish contamination resulting in human poisoning	• Fish advisories
Ecosystem Protection	• Diurnal fluctuations in pH and dissolved oxygen that can stress or eliminate sensitive species • Oxygen depletion in bottom water through decay of organic material eliminates sensitive species and releases sediment phosphorus • Reduced light penetration may cause macrophyte decline	• pH • Dissolved oxygen • Secchi depth

Table 2. Potential nuisance effects of excessive periphyton growth (from Quinn, 1991).

Water Uses	Nuisance Effects	Possible Endpoints
Water Supply	• Blockage of intake screens and filters	• Cost of treatment
Aesthetic appeal	• Reduced clarity and altered color due to sloughed material • Floating mats • Strandings on river margins during flow recessions cause odor	• Black disk. • Biomass • Odor indicators
Recreation	• Swimming and other water based recreation restricted or degraded due to aesthetic degradation • Slippery bed makes wading dangerous • Sloughed material fouls fisher' lines and nets • Dense algal mats restrict invertebrates preferred as food by sports fish	• Biomass • Black disk
Ecosystem Protection	• Diurnal fluctuations in pH and dissolved oxygen that can stress or eliminate sensitive species • Dense mats covering the bed reduce intergravel flow and habitat quality for benthic invertebrates and fish spawning	• pH • Dissolved oxygen • Biomass

frequently observed in the field (e.g., Vollenweider, 1968; Lee et al., 1995) makes the selection of nutrient endpoints for lakes and other lentic systems somewhat clear (i.e., a certain algal biomass is expected given a certain nutrient concentration). Such a linear relationship between ambient nutrient concentrations and periphyton biomass has not been observed in rivers and streams (Jones et al., 1984) and may not exist (Welch, 1994). A number of factors other than nutrient concentration govern periphyton biomass growth in rivers and streams, and the quantitative relationship between nutrient supplies and algal biomass in lotic systems has not yet been well characterized (Dodds and Smith, 1995). Nutrients receive the most attention only because the other factors are nearly impossible to control (Ingman, 1992).

One of the most important factors controlling periphyton growth in rivers and streams is velocity. Horner et al. (1990) and others have shown in laboratory streams that biomass growth is positively related to stream velocity in the range of 0 to 60 cm/s (after which increases in velocity decrease biomass growth by increasing erosion due to abrasion by suspended sediments.) Researchers believe that higher stream velocities enhance algal growth "by reducing the thickness of the . . . nutrient-depleted laminar boundary layer adjacent to the growing surface" (Dodds and Smith, 1995).

As mentioned previously, grazing pressure might also play a significant, non-nutrient-dependent role in periphyton growth. Grazing by aquatic insects has been shown to affect periphyton populations by decreasing biomass, affecting primary productivity, and changing the taxonomic composition and community structure (Walton et al., 1995). Other important factors that affect nuisance plant growth include frequency of scouring floods, quantity of suspended sediment, degree of shading, and types of substrata (Quinn, 1991). Water resource managers must recognize that the eutrophication process in rivers and streams is governed by the interactions between all of these various factors. They make the adoption of an effective, comprehensive nutrient enrichment strategy for rivers a complex process.

State of the Science—Endpoints

Selecting Endpoints

A review of the available literature suggests that there are several examples of rivers and streams with different nutrient enrichment endpoints already in place. These are described below along with other water quality parameters that could serve as endpoints in certain situations. The complete list of endpoints identified by participants in the National Nutrient Assessment Workshop is shown in the Table 3.

Table 3. Potential Endpoints for Plankton- and Periphyton- Dominated Systems.

Plankton Dominated	*Periphyton Dominated*
Algal biomass	Algal biomass (*mg/m², percent coverage*)
pH (maximum and diel)	pH (maximum and diel)
Dissolved Oxygen (minimum and diel)	Dissolved Oxygen (minimum and diel)
Transparency (Secchi depth)	Transparency (*Black* disk)
Biointegrity (macroinvertebrate index, community composition)	Biointegrity (macroinvertebrate index, community composition)
Total suspended solids, volatile suspended solids, ratios	Total suspended solids, volatile suspended solids, ratios
Dissolved organic material	Dissolved organic material
Autotrophic Index (AFDW/CNa)	Autotrophic Index (AFDW/chl *a*)
Total nitrogen	Total nitrogen, dissolved inorganic nitrogen
Total phosphorus	Total phosphorus, soluble reactive phosphorus
Ratios of summer/winter nutrient concentrations	Ratios of summer/winter nutrient concentrations
Ratios of dissolved/total nutrient concentrations	Ratios of dissolved/total nutrient concentrations
Aesthetics (foam, scum)	Aesthetics (foam, scum)

Periphyton Biomass. Perhaps the most obvious example of an endpoint for periphyton-dominated rivers and streams would be periphyton biomass. Periphyton biomass can be measured either qualitatively (e.g., no visible growths on hand-held stones) or quantitatively (e.g., ash-free dry weight or milligrams of chlorophyll *a* per square meter) (Quinn, 1991). The primary advantage of this measurement is that it directly reflects the water quality characteristic that impairs use. In this way biomass endpoints force managers to focus on all of the factors that contribute to periphyton growth, instead of relying only on nutrient controls to provide relief. (For example, repairing the canopy in a small river could potentially cause light to become a limiting factor.) The primary disadvantage associated with using periphyton biomass as an endpoint is the cost and difficulty associated with its monitoring.

An example of the adoption of algal biomass as a target is in the Clark Fork River in the states of Washington, Idaho, and Montana. The target is a mean value of 100 mg chlorophyll a/m^2 and a maximum value of 150 mg chlorophyll a/m^2. These values were chosen based on a literature review performed to determine what levels of algae interfere with beneficial water uses (Ingman, 1992), and they resemble levels proposed by Horner et al. (1983) and Welch et al. (1988). The targets have been adopted as part of the Clark Fork - Pend Oreille Basin Management Plan (USEPA, 1993). A variety of actions have been planned to achieve these targets, including instituting a basinwide phosphorus detergent ban and making improvements at municipal wastewater treatment facilities.

Dissolved Oxygen Concentration. Dissolved oxygen concentrations are an ideal endpoint where the primary beneficial use to be protected is support of aquatic life.

Such an endpoint could be stated as a minimum daily value or as an average value over a certain period (e.g., weekly, monthly, or seasonally). The advantage of using a daily minimum value is that this reflects what organisms (especially fish) respond to. However, practical limitations involved with daily monitoring of dissolved oxygen have often meant that the standard applies to a longer period of time. The spatial variability associated with dissolved oxygen readings also makes a longer monitoring period attractive. Current criteria for dissolved oxygen in ambient fresh waters have been established primarily on the basis of lethal conditions for freshwater fish (USEPA, 1986.) The criteria stipulate 30-day means of 6.5 and 5.5 mg/L and an acute, one-day (instantaneous) minima of 4.0 mg/Land 3.0 mg/L for cold waters and warm waters, respectively.

pH. pH measurements might be another parameter that is useful in instances where protection of aquatic life is of utmost importance. pH levels in gravel-bottom rivers with large periphyton biomass can be as high as 10, severely restricting the ability of stream organisms to function normally. A difficulty associated with this parameter is that factors other than eutrophication might be affecting water acidity. One advantage of using pH as an endpoint is that pH can be inexpensively monitored by nontechnical personnel.

Transparency. Transparency, as measured by Secchi depth, would be a very effective nutrient overenrichment endpoint in those rivers and streams which are plankton-dominated. Its advantages include its simplicity, low cost, and acceptability (Michaud, 1991). Its chief disadvantage in terms of assessing nutrient overenrichment is that it is only an indirect indicator of eutrophication. One cannot tell strictly from Secchi readings whether a river is impaired by high algal growths or by some other conditions, such as elevated levels of suspended sediments or color. Transparency would not be an appropriate indicator in high-gradient, periphyton-dominated systems.

Biointegrity/Macroinvertebrate Index. A number of states have used biological indicators in setting stream and river standards (Hughes et al., 1992). The Ohio EPA uses the index of biotic integrity (IBI), for example, to assess the aquatic life in its rivers and streams (Hughes et al., 1992). Biological indicators are advantageous in that they relate broadly to ecological condition, which is a primary concern of the public. However, it may be difficult to use these indicators in situations where a number of factors (in addition to nutrient enrichment) are affecting biotic integrity.

Nitrogen. Nitrogen concentrations can serve as useful endpoints in those systems where nitrogen is potentially the limiting factor. They can be measured as either total nitrogen or dissolved inorganic nitrogen (nitrate, nitrite, and ammonia). However, limiting concentrations in severely enriched waters might often be so low as to preclude serving as practical targets. This might be especially true for rivers affected by the filamentous green species *Cladophora*, which is able to survive in relatively low nitrogen environments (Ingman, 1992). The lack of understanding concerning the level of nuisance vegetative growth associated with nitrogen concentrations may also pose a challenge to using this nutrient as an endpoint. As mentioned above, other factors such as stream velocity, grazing populations, and light availability are likely to play an important role in periphyton growth.

Nevertheless, nitrogen concentration endpoints might still provide a variety of useful purposes. Where the limiting concentration is known, for instance, it could serve as the long-term goal of a nutrient management plan. Somewhat higher values could then be adopted as intermediate goals. Nitrogen endpoints could also be used to control the extent of nuisance growth, if not the total yield. Data suggest, for example, that nutrient additions beyond the range of 40 to 100 µg/L dissolved inorganic nitrogen will not the increase periphyton *yield* immediately downstream of a discharge, but might increase the downstream *extent* of periphyton proliferations (Quinn, 1991). A nitrogen endpoint that aims to limit the distance downstream at which periphyton growth reaches nuisance levels could therefore be instituted.

The nitrogen targets in the Clark Fork management plan mentioned earlier are 300 µg/L total nitrogen and 30 µg/L dissolved inorganic nitrogen (Ingman, 1992). These levels were based on the nitrogen concentrations found in reaches of the Clark Fork where algae are not a frequent problem.

Phosphorus. As with nitrogen, phosphorus endpoints (measured as either total phosphorus or soluble reactive phosphorus) are not as easy to implement in rivers and streams as they are in lakes and reservoirs. Again, limiting concentrations might be so low as to be difficult to achieve and an adequate relationship between phosphorus and algal growth might not be available. For the Spokane River, Welch et al. (1989) report that biomass levels exceeding 200 mg chlorophyll a/m^2 can persist for more than 10 km downstream from a point source unless soluble reactive phosphorus concentrations are held below 10 µg/L. Bothwell (1985, 1988) reports that streams can be phosphorus-saturated at concentrations as low as < 1 to 4 µg/L.

Summer phosphorus targets in the Clark Fork River were set at 6 µg/L soluble reactive phosphorus and 20 µg/L total phosphorus. These were based on ambient levels in relatively unimpaired reaches of the river (Ingman, 1992).

Ratio of Dissolved to Total Nutrient Concentrations and Ratio of Summer/Winter Nutrient Concentrations. In addition to using strict nutrient concentrations as endpoints, another possible methodology is to base management decisions on the ratios of either dissolved to total nutrients or summer to winter concentrations. The nitrogen-to-phosphorus ratio, both of loadings and of standing concentrations, can provide important insight regarding which nutrient should be controlled. Nitrogen and phosphorus are taken up by algae in an approximately constant ratio of 16 atoms of nitrogen per 1 atom of phosphorus, or 7.2:1 by weight. The practical range around this number is 10 to 20 (Chiaudani and Vighi 1974; Boynton et al., 1995; Thomann and Mueller, 1987). Waters with relative concentrations, then, of <10:1 nitrogen to phosphorus might have insufficient nitrogen for algal uptake relative to the available phosphorus. Nitrogen availability limits plant growth in these situations. When nitrogen:phosphorus is >20, the converse might be true: phosphorus becomes the limiting nutrient for plant production. An assessment of potential limitation at intermediate values is problematic. Complicating factors involve other limitations (e.g., light), the presence of nitrogen-fixing algae, and the availability of the forms of nitrogen and phosphorus present. Bioassays have been used to assess nutrient limitation; for fresh waters, a standardized test has been developed.

Others. Numerous other water quality measurements could potentially serve as nutrient overenrichment endpoints. Some of these might be appropriate for certain regions or for specific rivers with unique considerations. Many of the following measurements are not ideal overenrichment endpoints because they describe conditions that might be unrelated to nutrient loading. They might best be used in combination with some of the other measurements described previously.

- Total and volatile suspended solids concentrations
- Dissolved organic material
- Benthic community metabolism
- Sediment composition (organics, size fraction, nutrients, profile, sediment fluxes)
- Secondary production (meiofauna, macroinvertebrates, fish)
- Production/respiration
- Aesthetics (foam, scum)

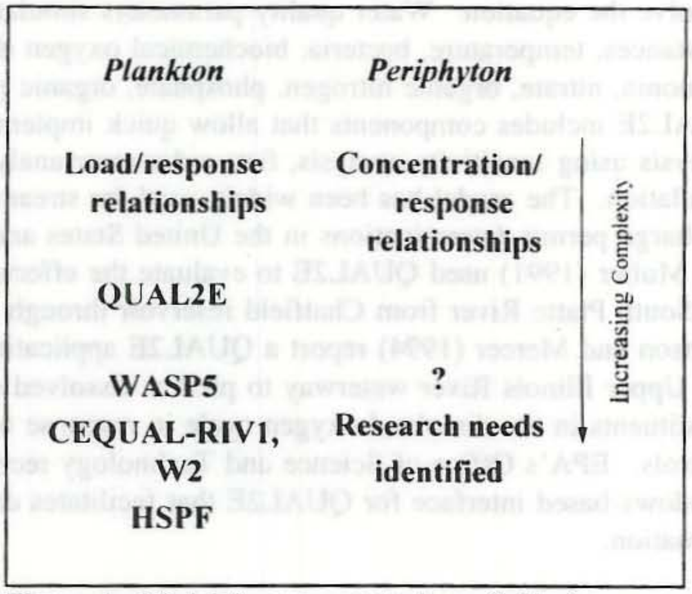

Figure 1. Models/assessment tools available for plankton and periphyton dominated systems.

State of the Science—Receiving Water Models

Several widely available simulation models can be used to assess aspects of the nutrient overenrichment process in rivers and streams (see Figure 1, for example). Plankton-dominated systems can be well described by currently available models that incorporate the dynamics of algal growth. Modeling of periphyton-dominant systems is limited by our understanding of the growth processes. Because the underlying chemical, physical, and biological processes are so complex, the accuracy that can be attained from using these models varies. Nevertheless, the ability to predict nutrient and dissolved oxygen concentrations, as well as algae and macrophyte levels, is essential to the water quality management process. The following summaries describe the state of the science regarding the available models. The descriptions are based on information from the revised *Compendium of Tools for Watershed Assessment and TMDL Development* (USEPA, in review).

QUAL2E. The Enhanced Stream Water Quality Model (QUAL2E), originally developed in the early 1970s, is a one-dimensional water quality model that assumes steady-state flow but allows simulation of diurnal variations in temperature or algal photosynthesis and respiration (Brown and Barnwell, 1987.) QUAL2E represents the stream as a system of reaches of variable length, each of which is subdivided into computational elements that have the same length in all reaches. Withdrawals, branches, and tributaries can be incorporated into the prototype representation of the stream system. The basic equation used in QUAL2E is the one-dimensional advection-dispersion mass transport equation. An implicit, backward difference scheme, averaged over time and space, is employed to solve the equation. Water quality parameters simulated include conservative substances, temperature, bacteria, biochemical oxygen demand, dissolved oxygen, ammonia, nitrate, organic nitrogen, phosphate, organic phosphorus, and algae. QUAL2E includes components that allow quick implementation of uncertainty analysis using sensitivity analysis, first-order error analysis, or Monte Carlo simulation. The model has been widely used for stream waste load allocations and discharge permit determinations in the United States and other countries. Paschal and Muller (1991) used QUAL2E to evaluate the effects of wastewater effluent on the South Platte River from Chatfield reservoir through Denver, Colorado, and Johnson and Mercer (1994) report a QUAL2E application to the Chicago waterway and Upper Illinois River waterway to predict dissolved oxygen and other constituents in the dissolved oxygen cycle in response to various water pollution controls. EPA's Office of Science and Technology recently developed a Microsoft Windows-based interface for QUAL2E that facilitates data input and output evaluation.

WASP5. WASP5 is a general-purpose modeling system for assessing the fate and transport of conventional and toxic pollutants in surface waterbodies. Its EUTRO5 submodel is designed to address eutrophication processes and has been used in a wide range of regulatory and water quality management applications. The model may be applied to most waterbodies in one, two, or three dimensions and can be used to predict time-varying concentrations of water quality constituents. The model reports a set of parameters, including dissolved oxygen concentration, carbonaceous biochemical oxygen demand (CBOD), ultimate BOD, phytoplankton carbon and chlorophyll *a*, total nitrogen, total inorganic nitrogen, ammonia, nitrate, organic nitrogen, total inorganic nitrogen, organic phosphorus, and inorganic phosphorus. Although zooplankton dynamics are not simulated in EUTRO5, their effect can be described by user-specified forcing functions. Lung and Larson (1995) used EUTRO5 to evaluate phosphorus loading reduction scenarios for the Upper Mississippi River and Lake Pepin, while Cockrum and Warwick (1994) used WASP to characterize the impact of agricultural activities on instream water quality in a periphyton-dominated stream. Stoddard et al. (1995) describe a fully three-dimensional application of EUTRO5 in conjunction with the EFDC hydrodynamic model to assess the effectiveness of options for the removal of total nitrogen from a wastewater treatment plant.

CE-QUAL-RIV1. The Hydrodynamic and Water Quality Model for Streams (CE-QUAL-RIV1) was developed to simulate transient water quality conditions associated with the highly unsteady flows that can occur in regulated rivers. The model has two submodels for hydrodynamics (RIV1H) and water quality (RIV1Q). Output from the hydrodyanmic solution is used to drive the water quality model. Water quality constituents modeled include temperature, dissolved oxygen, carbonaceous biochemical oxygen demand, organic nitrogen, ammonia nitrogen, nitrate nitrogen, and soluble reactive phosphorus. The effects of algae and macrophytes can also be included as external forcing functions specified by the user.

CE-QUAL-W2. CE-QUAL-W2 is a two-dimensional, longitudinal/vertical water quality model that can be applied to most waterbodies (Cole and Buchak, 1994). It includes both a hydrodynamic component and a water quality component. CE-QUAL-W2 is best applied to stratified waterbodies like reservoirs and narrow estuaries where large variations in lateral velocities and constituents do not occur. It might not be appropriate for high-flow streams and rivers. The model simulates the interaction of physical factors (such as flow and temperature), chemical factors (such as nutrients), and algal interaction. The water quality and hydrodynamic routines are directly coupled; however, the water quality routines can be updated less frequently than the hydrodynamic time step thereby reducing the computation burden for complex systems. A limitation associated with using CE-QUAL-W2 to address nutrient overenrichment issues is that it has only one algal compartment and algal succession, zooplankton, and macrophytes cannot be modeled.

CE-QUAL-ICM. CE-QUAL-ICM was developed as the integrated-compartment eutrophication model component of the Chesapeake Bay model package (Cerco and Cole, 1993). The model incorporates detailed algorithms for water quality kinetics. Interactions among variables are described in 80 partial differential equations that employ more than 140 parameters (Cerco and Cole, 1993). The state variables can be categorized into a group and five cycles—the physical group, and the carbon, nitrogen, phosphorus, silica, and dissolved oxygen cycles. An improved finite-difference formulation is used to solve the mass conservation equation for each grid cell and for each state variable. Although the model has been designed for application in Chesapeake Bay, it can be applied to other waterbodies. One limitation is the difficulty associated with amassing an adequate amount of data for calibration and verification. In addition, the model might require significant technical expertise in aquatic biology and chemistry to be used appropriately.

Estuaries and Coastal Waters

During the National Nutrient Assessment Workshop, several major themes and recommendations emerged from the detailed discussions of the estuaries and coastal water group. Perhaps the most pervading challenge in establishing tools and standards for assessment and management of nutrient inputs to aquatic systems is the fact that large, real variability exists in physical and biological structure and processes among all systems. This variability means that a given input might result in different responses. While science might agree qualitatively about the expected response, quantitative prediction is rarely if ever possible within acceptable confidence limits. In addition, regional differences in the human/societal expectations exist so that what condition is considered "acceptable" is not uniform. Taken together, these perspectives suggest that arbitrarily defined standards of general applicability might be untenable and strongly call for a different approach.

Most of the group's effort was initially directed toward what metrics seem meaningful, rigorous, and yet practically useful in the assessment of effects. The group set aside for the moment the issue of subjective definition of standards of what is acceptable or unacceptable as a basis for regulation. However these standards are defined, they must be based on acceptable measures of dose response.

Faced with this, the estuaries work group approached a consensus that the only practical initial approach is to construct empirical relationships between indicator responses and probable driving variables. This is essentially the approach taken by Vollenweider in documenting responses of lakes to phosphorus loading. Some examples of these empirical relationships are clearly significant statistically, but they display large variability, even for comparable estuaries. For only plankton-based systems, for example, chlorophyll stocks show a 10-fold range in response for similar inputs of total nitrogen. For other types of systems, however, such relationships have not been adequately explored. The group recommended that significant effort be devoted to a careful gathering of available information for a variety of systems, as well as an incisive analysis of relationships. It must be emphasized that is not a menial task. Care and insight are necessary to evaluate each published study to ensure appropriate comparisons are made.

A benefit of this approach is that the power of the method will increase with time. As more information becomes available, three significant areas of progress in the empirical approach can be anticipated. First, separate relationships might be developed for different types of systems, effectively stratifying the variance among more similar groups of sites (e.g., micro- vs. macrotidal). The group separated coral reefs, seagrass, and plankton-based systems with this rationale. Second, schemes for parameterizing multiple factors might be developed, within which tighter relationships might be possible. For example, Vollenweider's parameter normalized phosphorus input to freshwater turnover time. Third, as time series of data for an increasing number of cases become available, tighter relationships will emerge, more strongly suggesting site-specific predictive relationships.

In some cases, natural year-to-year variability might define the dose response quite accurately for a given system. This empirical approach guarantees a number of benefits. With time, the archive of systems for which diagnostic dose-response measurements are available will increase. Thus, the use of empirical relationships has immediate value, but is also a perfect stimulus for adaptive management using the more predictive relationships that will emerge with time.

The following discussion identifies several measurement indices for which empirical relationships can be determined in estuaries and coastal waters. It also addresses several tools to assess and identify nutrient limitation and potential indicators of historic trends.

State of the Science—Endpoints

Clarity. Clarity is a function of the quantity and spectral quality of light in the aquatic environment. Nutrient enrichment affects clarity through the promotion of phytoplankton growth and biomass accumulation. Phytoplankton of various groups absorb and attenuate light in the water column (Kirk, 1983). They shade each other and the bottom. At high biomass, conditions of low light selectively favor algal species that are adapted to low light or have motility, morphological, and/or buoyancy mechanisms that sustain them near the water surface. By preventing sufficient light from reaching the bottom, they eliminate most benthic primary production, leading to the demise of benthic algal and vascular plant populations. Often these algae are not edible or easily grazed on by zooplankton and fish planktivores, or they lead to gill-clogging in fish. The result is a shift in trophic status toward a monocultural, algal-dominated system. The fate of this organic carbon is microbial breakdown, with subsequent hypoxia/anoxia and organic accumulation in the sediments.

However, clarity also has contributing factors that are not the result of nutrient enrichment. These factors include colored dissolved organic matter, such as humic and fulvic acids, which are the degradation products of organic matter from terrestrial and aquatic plants and animals. In some regions, rivers deliver large quantities of dissolved organic matter into estuarine and coastal waters. Dissolved organic matter is a strong absorber of light in the blue region of the spectrum, near the blue absorption peak of chlorophyll (Kirk, 1983). Of course, water itself is a strong absorber in the red region of the spectrum. The light that penetrates waters high in dissolved organic matter has a yellow quality and is of little use to plants regardless of how much is present. Another factor in clarity not directly related to nutrient enrichment is the suspended inorganic load, which predominately scatters light, again most strongly in the blue region of the spectrum (Kirk, 1983).

It is clear, then, that with respect to clarity and nutrient enrichment, only certain measurement endpoints are applicable. Secchi disk depth, a quick, inexpensive, and ubiquitous measure of light penetration, is indiscriminant with respect to the factors affecting clarity. Also, Secchi disk determinations are subject to the response of the human eye, which is more sensitive to light in the middle of the visible spectrum, while plants absorb, and therefore require, light in the blue and red regions of visible light. Therefore, although Secchi depth is directly a measure of water clarity, it is not the optimal measure of light penetration pertinent to photosynthesis. The diffuse downwelling light attenuation coefficient (K_d) for photosynthetically active radiation (PAR) is an endpoint that can be related to water quality parameters and is relevant for assessment of potential photosynthesis. It also has the advantage of connecting traditional water quality monitoring and modeling parameters with a key indicator of habitat quality for plant production, especially for benthic flora. Measurements of K_d are made with quantum light sensors. Supporting measurements should include water column chlorophyll *a*, total suspended matter, and if warranted by the water body, color or dissolved organic matter. Advancement of the science of near-shore aquatic optics should involve spectral characteristics of light penetration and the optically active constituents in estuarine and coastal waters (e.g., Gallegos, 1994; McPherson and Miller, 1987) and the relationships between how light conditions are assessed and the natural (temporal) signals and availability of light in the aquatic environment (Morris and Tomasko, 1993; Miller and McPherson 1995).

Hypoxia/Anoxia. Dissolved oxygen concentration in natural waters is a function of physical and chemical factors that determine solubility and the transport of oxygen across the air-water interface, as well as the mixing of surface and deep waters. Within a water mass, a mix of biological processes determine oxygen concentration. Phytoplankton in surface waters and, given sufficient light penetration, benthic algae and vascular plants produce oxygen via photosynthesis. Plant, animal, and bacterial respiration consumes oxygen, as do bacterially mediated transformations of nitrogen during nitrification (oxidizing ammonium to nitrite and nitrate). Other bacterial processes leading to the reduction of sulfate and carbon dioxide produce sulfide and methane, which react with and provide additional demands on dissolved oxygen. Considering especially these biological processes, nutrient enrichment is often signaled by excessive oxygen production in surface waters, leading to supersaturation in some cases, and by hypoxia or anoxia in deep waters when excessive plant production is consumed.

Thus, dissolved oxygen concentration is one of the major endpoints for assessment of eutrophication in estuaries and coastal waters. Criteria for dissolved oxygen in ambient (fresh) waters have been established, principally on the basis of lethal low dissolved oxygen conditions for freshwater fish (USEPA 1986). The criteria stipulate 30 day means of 6.5 and 5.5 mg/L and acute, 1 day (instantaneous) minima of 4.0 and 3.0 mg/L for cold water and warm water, respectively.

In coastal waters where natural variability is high, long-term records of dissolved oxygen concentrations in both surface and bottom waters might be necessary for the evaluation of nutrient enrichment effects (e.g., Justic et al., 1987). In estuarine and coastal waters, hypoxia, generally where dissolved oxygen concentration drops below 2.0 mg l^{-1} (Harding et al. 1992), is apparent with alterations in the benthic macrofaunal community, both in structure and loss of biomass (Smith and Dauer, 1994; Schaffner et al., 1992).

Submerged Aquatic Vegetation (SAV) Losses. SAV is composed of flowering vascular plants (angiosperms) that live completely underwater; a few species have flowers that protrude through the surface (Hurley, 1990). SAV species are found from freshwater to marine habitats. While several hundred SAV species are known from freshwater and brackish habitats, the truly marine species number about 60 (den Hartog, 1970; Dennison et al., 1993). SAV species are relatively sensitive to the available underwater light; they require approximately 20 percent of daily incident light for their survival in comparison with other plants (ranging from planktonic and benthic algae to forest floor dwellers), which require ≤1 percent of daily incident light (Dennison et al., 1993; Kenworthy and Haunert, 1991).

In temperate, nutrient-enriched estuaries such as the Chesapeake Bay, this light requirement restricts the maximum depth of occurrence to 1 to 2 meters (MLW). Decadal scale changes in water transparency are reflected in the changes in the historic distributions of SAV (Orth and Moore, 1983, 1986). Simulation analysis driven by light attenuation coefficients from sites both with and without present SAV beds also suggest sensitivity to available light as a determinant in the survival and depth of occurrence of SAV (Wetzel and Meyers, 1994). Nutrient enrichment represents a large factor in water transparency by supporting increased phytoplankton biomass. It also enhances epiphytic growth on SAV leaf surfaces themselves, further limiting the light available to SAV for photosynthesis (Neckles et al., 1993). This later relationship, however, can be mitigated by extensive grazing on epiphytes (Neckles et al., 1993).

SAV are an important source of habitat, providing refuge to juvenile fish and shellfish and providing a food source for fish and waterfowl. Consequently, the assessment of SAV provides a direct link between water quality (nutrients, chlorophyll *a*, and suspended sediments) and ecologically and economically important species. In addition, the use of SAV as an water quality endpoint is not confounded by direct human uses, unlike similar assessments for fisheries stocks (Dennison et al., 1993). Lastly, the sensitive relationship between SAV, water transparency, and nutrients makes SAV a good indicator of habitat quality.

The sensitive nature of SAV relative to available underwater light makes the maximum depth of individual SAV beds an indicator of habitat quality. Assessment criteria should be related to the species present or potentially able to inhabit the site, with targets based on species-specific light requirements for growth applied to the maximum depth observed for the species in pristine sites similar to the region of interest. As a guide, a target of not less than 20 percent of daily irradiance just below the water surface available at the bottom during the growing season of the plant may be set. At a depth of 2.0 m, this would correspond to a downwelling diffuse attenuation coefficient of $-\ln(0.20)/2.0 = 0.80$ m^{-1}. Required knowledge includes key SAV species and their light compensation points under local conditions, and the desired depth distribution, based on historic data obtained prior to nutrient enrichment or data from a comparable location.

Geographic (areal) distribution, in concert with maximum depth of distribution, defines the area of SAV habitat. The availability of this habitat is important for fish, shellfish, and waterfowl. Monitoring, typically on a large scale with remote imagery (aerial photography, satellite imagery), in combination with a GIS database approach, can provide regional areal estimates and the basis for spatial and temporal trend analysis. NOAA's C-CAP program is evaluating methods and building a database of land (including submerged lands or estuaries) use and biotic change using remote sensing techniques (Dobson et al., 1995).

Monospecific Algal Blooms and Trophic Alterations. Alterations in either the primary production or the consumption of that production can change the state of a waterbody. Nutrient enrichment can cause such changes. One such alteration is an ultimate shift from diatom-dominant production, which is easily grazed on and ultimately beneficial to desirable fish stocks, to other phytoplankton groups that are less heavily grazed on. Additions of nitrogen and/or phosphorus lead to short-term increases in diatom production, followed by depletion of dissolved silica (DSi) and eventually, the replacement of diatoms with green and blue-green algae as DSi concentrations fall into limiting ranges for diatom growth (Schelske and Stoermer, 1971, 1972). The implications of this for coastal and marine environments were made clear by Officer and Ryther (Officer and Ryther, 1980, 1981): food webs and fates of phytoplankton carbon are distinctly different depending on whether primary production is dominated by diatoms or by other algae (including greens, flagellates, and blue-greens). Enrichment-induced shifts from a diatom-based food web lead to changes in food web structure, away from production of desirable fish species toward the production of excessive, ungrazed algae, which sink and promote bottom water hypoxia/anoxia. Recent reviews of the global implications of alterations in the Si:N:P ratio, including increased potentials for toxic and nuisance algal blooms, food web alterations, and changes in bottom water oxygen dynamics, further suggest that coastal waters that once had excesses of silicon relative to nitrogen and phosphorus are now approaching nutrient balance or are experiencing seasonal silicon limitation (Justic et al., 1995; Conley et al., 1993). This suggests that coastal areas dominated by direct river discharge will be subject to increasing alterations in plankton composition, increased occurrences of nuisance algal blooms, and subsequent changes in trophic structure.

Although not directly impacted by nutrient enrichment, suspension-feeding populations, both benthic and planktonic, in some cases can exert a primary control on algal populations and alter suspended particle concentrations. Losses of these populations by harvesting (shellfish), disease, increases in suspended load, enrichment-induced hypoxia, or shifts in algal populations represent positive feedback on eutrophication effects. That is, a control of algal populations that under other circumstances would counter the enrichment effect, fails, eliciting greater negative impacts on the ecosystem. Furthermore, with respect to restoration goals, in the absence of these "top-down" grazer controls, it may be more difficult or take longe to restore a system using moderate nutrient load reductions.

Dissolved Concentration Potential. Similar to Vollenweider relationships for lakes, Quinn and others (1989) have proposed the use of the dissolved concentration potential (DCP) as an index of estuarine susceptibility to eutrophication. In a comparison mode, the DCP uses a constant loading, L=10,000 tons/year so that intrinsic physical differences (i.e., flushing ability) among estuaries can be assessed. For evaluation of an individual estuary's status and classification relative to the impact of its nutrient loadings, the actual loading value is used for W.

$$DCP = (W / Q_{fw})\ (V_{fw} / V_{tot})$$

where:

W = nutrient loading rate (kg d^{-1}),

V_{fw} = volume of freshwater in the estuary (m^{-3}),

Q_{fw} = rate of freshwater inflow ($m^3\ d^{-1}$), and

V_{tot} = total volume of estuary (m^{-3}).

Annual averages are used for inflows and volumes (NOAA, 1985; Klein and Orlando, 1989). Essentially, DCP is a loading factor weighted for total volume and freshwater turnover. Orders of magnitude for potential enrichment impact have been established: low (0.01-0.10 mg/L, med (0.1-1.0 mg/L), high (1.0-10 mg/L). Limitations of this index are (1) it assumes conservative behavior of bioactive nutrient elements; (2) the oceanic nutrient source is neglected; (3) it assumes a vertically homogeneous estuarine water mass; (4) it ignores variability (seasonal, aperiodic) in discharges; and (5) to be applicable, it requires a gradient in salinity relative to the seawater source (e.g., DCP would be a poor predictor of susceptibility in estuaries dominated by seawater inputs such as Cape Cod Bay).

NOAA's National Estuarine Inventory (NEI) has tabulated estimates of nitrogen and phosphorus concentrations for given estuaries. Note that the estimates are based on loadings and are subject to error resulting from the relative degree of recycling within an estuary, which might be very high (Boynton et al., 1982; Kemp et al., 1982). Based on this and on observations of eutrophication impacts within estuaries and the DCP, classifications of nutrient concentrations were developed:

Nitrogen: low (<0.1 mg/L), medium (0.1-1.0 mg /L), high (>1.0 mg/L).
Phosphorus: low (<0.01 mg/L), medium (0.01-0.10 mg/L), high (>0.10 mg/L)

The utility of this classification scheme is problematic. The loadings from which concentrations were estimated are based on total inputs, some portion of which (particularly in the case of phosphorus) might not be available to plants because of its chemical form. Further, estimates based on loadings might not reflect actual concentrations because of the intensity of recycling within a particular estuary or subestuary (Boynton et al., 1982). They also do not take into account the relative recycling rates of nitrogen versus phosphorus or the denitrification potential, which is a significant loss of nitrogen in estuarine systems (Seitzinger, 1988).

Nitrogen:Phosphorus Ratio. Another proposed indicator of potential enrichment problems is the nitrogen-to-phosphorus ratio, both of loadings and of standing concentrations. Nitrogen and phosphorus are taken up by algae in an approximately constant ratio of 16 atoms of nitrogen per 1 atom of phosphorus, or 7.2:1 by weight. The practical range around this number is 10-20 (Chiaudani and Vighi, 1974; Boynton et al, 1995; USEPA, 1985). Waters with relative concentrations, then, of <10:1 nitrogen to phosphorus might have insufficient nitrogen for algal uptake relative to the available phosphorus. Nitrogen availability limits plant growth in these situations. When nitrogen:phosphorus is >20, the converse might be true: phosphorus becomes the limiting nutrient for plant production. An assessment of potential limitation at intermediate values is problematic. Complicating factors involve other limitations (e.g., light), the presence of nitrogen-fixing algae, and the availability of the forms of nitrogen and phosphorus present. Bioassays have been used to assess nutrient limitation; for fresh waters, a standardized test has been developed.

Biological Indicators. Detection of incipient change resulting from nutrient enrichment relies on consistent and continuous monitoring programs. The natural variability inherent in measurement endpoints from estuarine and coastal waters makes the interpretation of trends difficult unless a sufficient multiyear record, i.e., 10 years or more, is available, especially if biological variables are to be used as indicators.

Benthic fauna biomass and composition, larval fish abundance, and algal and zooplankton species composition are both affected by and may be good substitutes for primary endpoints like dissolved oxygen and nutrient concentrations. The use of these biological indicators as measurement endpoints provides a fundamental link between water quality and habitat quality, as well as stated assessment endpoint objectives, i.e., the maintenance of beneficial algal populations, a healthy benthic fauna, and desired abundances of commercially or recreationally important fish stocks, for restoration plans.

Tools to Assess and Identify Nutrient Limitation

Algal Growth Potential Test. The Algal Growth Potential Test (AGPT), reviewed by Raschke and Schultz (1987), provides a means to assess and identify potential nutrient limitation and the potential impacts of increased concentrations of a given nutrient on the resultant algal production. Bottle assays of *Selenastrum capriconutum*, a freshwater alga, using various test waters (fresh waters only) based on the premise that yield is proportional to the bioavailable nutrient in least supply (Liebig's Law of the Minimum) can be used to define nutrient limitation. The advantage of this test as a standard method is its simplicity and low cost. Yields respond to bioavailable portions of total nitrogen and total phosphorus pools. There is no need to assume which nutrient element is limiting, nor assume a maximum potential growth rate. The test can give maximum potential growth rate estimates and identify the limiting nutrient. The disadvantages are the questionable applicability of a standard organism to site-specific, species-specific responses; i.e., the test does not consider the ecological or trophic relationships of native species and their grazers. There appears to be a need to establish relationship between the test maximum allowable biomass of *Selenastrum capriconutum* and that for the native suite of species present in the waterbody of interest.

Enzyme Assays. In estuarine and coastal waters, as well as in fresh waters, enzyme assays have been used to assess phosphorus limitation. Vadstein et al. (1988) and Vargo and Shaney (1985) have shown that phytoplankton alkaline phosphatase (AP)[2] activity might be related to phosphorus limitation. In addition, bacterial AP as well as 5′-nucleotidase (5PN), which cleaves the phosphate moiety off nucleotides and other organic phosphorus sources, might be indicative of phosphorus-limited systems or might increase the pool of bioavailable phosphorus (Ammerman and Azam, 1991). Problems of spatial scale, in terms of relating this activity to ambient concentrations of organic phosphorus, exist. It has been suggested that recycling of organic phosphorus to inorganic phosphorus might be tightly coupled to organic matter excretion within clusters of bacteria and phytoplankton such that recycling appears to be independent of bulk concentrations of either soluble reactive phosphorus or organic phosphorus (Ammerman and Azam, 1991). Further use of such an assay requires more scientific investigation and a larger database in relation to changes in nutrient enrichment status of a given waterbody.

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[2] Alkaline phosphate is an enzyme present on the outer cell membrane that cleaves the phosphate group off dissolved organic matter.

Other elements, notably silica, have an impact on the production of specific algal groups, i.e., diatoms. The impact of nutrient enrichment and environmental change on the relative contribution of diatoms to total phytoplankton production and its implications for trophic alterations are discussed above.

Indicators of Historic Trends

Patterns observed in monitoring programs in estuaries, coastal waters, and their tributaries might be biased by the relatively brief period of coverage. Changes that might or might not be attributed to nutrient enrichment because of their ambiguity need to be placed into a context that extends back prior to the intense coastal development of the 20th century and, if possible, prior to the intense agricultural development of the 19th century. Such long-term records of biotic and physical conditions within estuarine and coastal waters place nutrient enrichment processes in context with natural long term trends and meterological cycles, which might be biasing our present view of the situation. Historical records are also useful for calibrating simulation models prior to the simulation of nutrient enrichment effects and testing management scenarios for nutrient reduction (Chapra, 1977).

In a review of historic trends in Chesapeake Bay nutrients, D'Elia suggests that a time series of data should encompass 10 cycles in order to distinguish cyclical patterns from long-term trends; for a 6-year climatic rainfall cycle, a 60-year nutrient loading record would be required (D'Elia, 1982). Though few, there are nutrient, suspended sediment and oxygen records that extend over many decades through the 20th century and can at least lend weight to the fact that nutrient enrichment in coastal waters has exerted real impacts on a global scale (Heinle et al., 1980; Justic et al., 1987; Walsh et al., 1981; Turner and Rabalais, 1991).

Additional perspective, extending our view centuries into the past, can be obtained from the sediment record. Paleoecological indicators have been analyzed for a 2,500-year history in Chesapeake Bay (Cooper and Brush, 1993). These indicators include pollen, diatom assemblages, total organic carbon, nitrogen, sulfur, and the degree of pyritization. Pollen and charcoal can be used to date deforestation/agricultural events and natural fires/early industrial activity, respectively (Brush, 1989). Preserved frustules (silicious, species-specific cell walls) of diatoms can be used to indicate loss of diversity and shifts in the centric-to-pennate ratio (c:p). Centric diatom forms are cylindrical and typify planktonic species, whereas pennate forms are oblate and typify benthic species. Increased predominance of centric forms indicates shifts from benthic to pelagic primary production as would accompany increased nutrient loading and turbidity (Cooper and Brush, 1993). In Chesapeake Bay, there is a dramatic increase in organic carbon and in sulfur deposition since the mid 1700s, continuing to the present (Cooper and Brush, 1991, 1993). Similarly, the degree of pyritization, which is the ratio of pyritic iron to pyritic iron plus acid-soluble iron, which varies with the oxygen content of bottom waters during deposition (Raiswell et al, 1988), has also increased in the last two centuries of coastal habitation (Cooper and Brush, 1991, 1993). Since there is no natural signal during this time frame that can compare with the increase in human activity, the historic record presents a strong case for the impacts that humans can have on the coastal environment.

The value of these long-term records is that they corroborate long model runs prior to management scenario testing. Additionally, and perhaps most importantly, they do show a discernible human signal in the environmental record which puts a magnitude on the impacts.

State of the Science—Models

A discussion of the issues associated with assessing nutrient overenrichment in lakes and reservoirs would be incomplete without addressing the use of water quality models. As mentioned in the introduction to the issue papers, numerous watershed loading models are available for the assessment of nutrient overenrichment, many of which can be applied across multiple waterbody types. A summary is provided here of some of the receiving water models that are especially useful for predicting and assessing the eutrophication process in lakes and reservoirs. These summaries were prepared based on information from the revised *Compendium of Tools for Watershed Assessment and TMDL Development* (USEPA, in review). References pertaining to each of these models are listed in a separate section at the end of this document.

WASP5. WASP5 is a general-purpose modeling system for assessing the fate and transport of conventional and toxic pollutants in surface waterbodies. Its EUTRO5 submodel is designed to address eutrophication processes and has been used in a wide range of regulatory and water quality management applications. The model can be applied to most waterbodies in one, two, or three dimensions and can be used to predict time-varying concentrations of water quality constituents. The model reports a set of parameters, including dissolved oxygen concentration, carbonaceous biochemical oxygen demand (CBOD), ultimate BOD, phytoplankton carbon and chlorophyll *a*, total nitrogen, total inorganic nitrogen, ammonia, nitrate, organic nitrogen, total inorganic nitrogen, organic phosphorus, and inorganic phosphorus. Although zooplankton dynamics are not simulated in EUTRO5, their effect can be described by user-specified forcing functions. Stoddard et al. (1995) describe a fully three-dimensional application of EUTRO5 in conjunction with the EFDC hydrodynamic model to assess the effectiveness options for the removal of total nitrogen from a wastewater treatment plan.

Tidal Prism Model (TPM). TPM was originally developed as a tool for water quality management of small coastal basins (Kuo and Neilson,1988). Physical transport processes are simulated in terms of the concept of tidal flushing. The numerical solution scheme implemented for solving the tidal flushing equations is well suited to application in small coastal basins, including those with a high degree of branching. The model allows consideration of shallow embayments connected to the primary branches in the basin. The basic assumptions in the model are that the tide rises and falls simultaneously throughout the waterbody and that the system is in hydrodynamic equilibrium. Kinetic formulations in TPM are similar to those in CE-QUAL-ICM (Cerco and Cole, 1993), and 23 state variables, including total active metal, fecal coliform bacteria, and temperature, can be simulated. TPM includes a sediment submodel, also based on the sediment process model in CE-QUAL-ICM, that considers the depositional flux of particulate organic matter, its diagenesis, and the resulting sediment flux. TPM has been applied to a number of tidal creeks and coastal embayments in Virginia (Kuo and Neilson, 1988).

CE-QUAL-ICM. CE-QUAL-ICM was developed as the integrated-compartment eutrophication model component of the Chesapeake Bay model package (Cerco and Cole, 1993). The model incorporates detailed algorithms for water quality kinetics. Interactions among variables are described in 80 partial differential equations that employ more than 140 parameters (Cerco and Cole, 1993). The state variables can be categorized into a group and five cycles—the physical group, and the carbon, nitrogen, phosphorus, silica, and dissolved oxygen cycles. An improved finite-difference formulation is used to solve the mass conservation equation for each grid cell and for each state variable. Although the model has been designed for application in Chesapeake Bay, it may be applied to other waterbodies. One limitation is the difficulty of amassing an adequate amount of data for calibration and verification. In addition, the model might require significant technical expertise in aquatic biology and chemistry to be used appropriately.

Wetlands

Wetlands are consistently cited for their value in the control of nonpoint source (NPS) pollution through water storage, flood attenuation, and water quality enhancement through uptake and processing of nutrients, sediments, and toxic chemicals (see Olson, 1993). Texts have been written on the design, construction, and use of wetlands specifically for treatment of wastewater and enhancement of water quality (Hammer, 1989; Moshiri, 1993). Because the purifying capacity of wetlands is so often identified as fundamental in their value to society, the question might arise: "Is wetland eutrophication a national problem?" After all, wetlands are most often the first line of defense in protecting receiving waters from the effects of pollution such as excessive nutrients.

Many authors indicate that potential impacts to natural wetlands from nutrient enrichment and pollutant overloads are of significant concern to researchers and resource managers (Olson, 1993; Walbridge, 1993; van der Valk and Jolly, 1992; Bowden et al., 1991; Landers and Knuth, 1991; Hammer, 1989; Johnston, 1989; Cooper and Gilliam., 1987; Kadlec, 1983). However, scientific understanding of wetlands, wetland management, and even wetland classification schemes is still evolving, and synthesizing common information about such variable ecosystems on a nationwide scale remains an onerous task (see Nixon and Lee, 1986, Knight, 1992). As examples, researchers such as Brinson (1993) Adamus et al. (1987); and Kent et al. (1992) discuss classification and monitoring strategies for wetlands, and others, such as Howard-Williams (1985) and Johnston (1991) discuss nutrient dynamics in various wetlands. However, in the literature reviewed, no examples of a standardized strategy to assess or manage nutrient overenrichment in the interest of wetland "health" were found.

The purpose of this working document is to stimulate thought and discussion regarding how to assess and manage nutrient overenrichment in wetlands. The literature reviewed and cited does not reflect a comprehensive examination of relevant research in wetlands, but has been selected to highlight particular issues.

This discussion restricts the definition of "wetlands" to those lands which are transitional between terrestrial and aquatic systems (for example, shallow, vegetated tidal and nontidal wetland communities, either separate from or adjacent to stream, river, lake, and coastal waterbodies). Benthic wetland communities, such as submerged aquatic vegetation located within streams or rivers, or seagrass beds in nearshore coastal environs, have been implicitly excluded because they are part of receiving waters addressed in other sections of this overview.

Background

Nationally, natural wetlands continue to experience reduction in abundance and distribution. Approximately half of the wetlands in the continental United States have been lost since colonial settlement (Frayer et al., 1983). Although wetland modification has slowed significantly since the 1950s, wetland losses from dredging, filling and land development have continued. National policy instituting a "no net loss" of wetlands was adopted in the mid-1980s in an attempt to reverse the negative trend.

In July 1990 the USEPA developed national guidance for wetland water quality standards to be established during fiscal years 1991-1993. The guidance was developed to ensure that provisions of the Clean Water Act (CWA) applied to other surface waters were also applied to wetlands. States were required to include wetlands in their definition of "State Waters," as well as to adopt narrative and numeric criteria and antidegradation policies for wetland protection (USEPA, 1990). Constructed wetlands, for the specific purpose of wastewater treatment, are excluded from protection; however, the knowledge gained from the processes within these wetlands might contribute to developing endpoints or standards elsewhere. Currently, the standards for wetlands are primarily narrative criteria developed by states, because it has been difficult to apply certain numeric criteria to wetlands. For example, dissolved oxygen is hard to standardize because a wetland can be dry and have zero dissolved oxygen. Enforcement and setting of permit limits are based on fixed numbers, but for wetlands the limits might be based on concentrations related to flow or perhaps load-based measures.

On June 10-11, 1991, EPA sponsored a workshop to define scientific and policy issues concerning wetland quality and nonpoint source (NPS) pollution, and to evaluate the role of created and natural wetlands in control of rural NPS pollution (Olson, 1993). General findings of the workshop identified several key points. Since eutrophication can be considered a subset of NPS pollution, the following findings and issues provide a basis for further consideration in development of a national nutrient assessment strategy:

Natural wetlands should not be used as wastewater treatment systems. In most cases, natural wetlands are considered waters of the United States and thus are entitled to protection under the CWA. Natural wetlands, while providing NPS pollution control service, should not have NPS pollutants intentionally diverted into them. Furthermore, natural wetlands that may be in threat of significant community changes should be protected from NPS sources through the use of BMPs or vegetated buffer strips. Several researchers echo this concern. Hammer (1989) and van der Valk and Jolly (1992) also recommend that natural wetlands be avoided when trying to address excessive nutrient loadings. Natural wetlands are not particularly effective as nutrient sinks, but clearly perform a valuable function in attenuating diffuse runoff (Howard-Williams, 1985). Johnston (1991) identified wetlands designed for pollutant filtration as having a much greater assimilative capacity and range compared to natural wetlands.

Wetlands must be part of an integral landscape approach to NPS control. Created, restored or natural wetlands can contribute to this strategy, but must be sited appropriately, so they are not overloaded. Attempts have been made in developing guidelines for certain aspects of wetlands as pollutant filters, including wetland siting within a watershed or landscape unit. However, certain technical issues, such as fate and effects of contaminants and development of design and site selection criteria, are in need of resolution before siting and using restored wetlands for NPS pollution control (van der Valk and Jolly, 1992).

Knowledge of technical issues is uneven. This point was also made by Nixon and Lee (1986). Having performed an extensive review of wetlands and water quality, they state that information on wetlands nationwide is unevenly distributed, and there is a need to learn a great deal more about wetland processes and functions before the scientific community can generalize the overall role of wetlands in water quality. Current knowledge of nutrient dynamics (and other parameters) varies by wetland type and region. These factors are important considerations since they might indicate that a regional or ecoregional strategy might be more effective for nutrient assessment than a single, national strategy.

Technical and scientific issues will not be as difficult to resolve compared to social and economic ones, largely because administrative boundaries rarely coincide with the logical management units of watersheds. This issue is particularly challenging for environmental managers in developing cooperative agreements with other administrations sharing the same watershed(s). The changes observed in wetlands from nutrient enrichment are likely to be more subtle and over a longer term compared to receiving waters, such as lakes and reservoirs. Public use values for receiving waters, such as drinking water, fishing, and swimming, have been well developed for most areas. Conversely, social, and in particular, economic use values might not be as well developed or expressed for specific wetlands in some regions. In general, large uncertainties remain in the economic evaluation of wetlands (Costanza et al., 1989).

For this discussion, additional policy issues/questions to consider include the following:

- Eutrophication needs to be defined according to whether it is natural or human-induced.
- At what point is eutrophication in a wetland system considered a "bad" thing, and what standards will be used for comparison?
- Decisions and value judgments are needed to determine whether a given system should be controlled or allowed to follow the change (even if that change is human-induced).

State of the Science—Endpoints

Knowledge of wetlands has increased significantly over the last 20 years. The scientific literature describes studies of nutrient dynamics, uptake capacity of soils and wetland types, examples of using wetlands to ameliorate nutrients from agricultural or nonpoint source pollution, and discussions of wetland values (for example, Gilliam, 1994; Detenbeck et al., 1993; Phillips et al., 1993; Baker, 1992; Knight, 1992; Masscheleyn et al., 1992; Mitsch et al., 1991; Costanza et al., 1989; German, 1989; Kuenzler, 1989; Reddy et al., 1989; Kemp et al., 1985; Rhodes et al., 1985; Johnston et al., 1984). A common finding in much of the literature reviewed is that natural wetlands have many dynamic variables. For nutrients, various input sources, such as surface, atmosphere, runoff, and groundwater make it difficult to generalize about their effect on *wetland* quality (Johnston, 1991; Leibowitz and Brown, 1990; Nixon and Lee, 1986).

The use of restored or constructed wetlands specifically for nutrient retention (i.e., sewage treatment) and water quality enhancement has been extensively studied and has emerged as a discipline within wetland science (see Moshiri, 1993; Mitsch, 1992b; Hammer, 1989). Consequently, findings from work with engineered wetlands are likely to be useful in determining some measurement parameters and appropriate endpoints. However, compared to many natural wetlands, constructed wetlands are often much less diverse in biological and physical characteristics, compared to many natural wetlands.

A potentially important consideration in assessing nutrient conditions in wetlands on a broad scale, such as a watershed or ecoregional level, might involve revisiting wetland classification with nutrient dynamics in mind. The Cowardin et al. (1979) hierarchical system of wetland classification is based largely on plant species composition. Although this classification has been useful for inventorying wetlands nationally, uptake by plants is only one pathway for nutrients. Alternatively, Brinson (1993) developed a wetland classification system that grouped characteristics more by hydrogeomorphic processes. Figure 2 depicts nutrient availability as a core factor with hydrologic energy and hydroperiod. Biological attributes, such as wetland species composition, are then used to refine wetland classification. Such a general classification approach might be useful in organizing wetlands specifically for nutrient management.

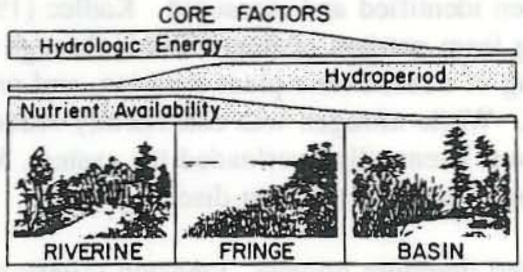

Figure 2. Relationship of core hydrogeomorphic factors (note nutrient availability) to charactrize various wetland types. Modified from Brinson (1993).

Nutrients in Wetlands

Nutrient retention and release kinetics are highly site-specific, and uptake mechanisms and differ among wetland types, soils, water levels, seasons, and other parameters (Phillips et al., 1993; Johnston, 1991; Kemp et al., 1985). Factors contributing to variable nutrient uptake in wetlands include:

- Source. Wetlands can receive nutrients from agricultural, rural or urbanizing environments, which deliver varying concentrations. Sources might be specific points or a cumulative contribution from nonpoint sources, or both. Nutrients are carried by inputs from streams, rivers, tidal exchanges, groundwater, precipitation, or atmospheric deposition.

- *Physical Factors.* Latitude, climate, hydrology, and soil type are among the physical attributes that affect wetland size, structure, and, consequently, nutrient dynamics.

- *Chemical Factors.* pH, organic matter, substrate characteristics, nitrification/denitrification, and reduction/oxidation potential all affect nutrient cycling and availability.

- *Biological Factors.* Plant, animal, and microbial ecology all contribute to nutrient dynamics and pathways within wetlands. Aside from sediment often serving as a sink for phosphorus, biological components serve as significant nutrient storages and processors in wetlands. The availability of biological processes depends greatly on source, physical, and chemical factors.

Several other authors have identified hydrology within wetlands as critical, but understanding of nutrient dynamics is lacking (Gilliam, 1994; Brinson, 1993; Johnston, 1991; Clausen and Johnson, 1990). The highly transient hydrology of many wetlands makes evaluating long-term nutrient and sediment removal difficult. However, in spite of the range of variables and currently unknown factors of nutrient dynamics in wetlands, some ecological consequences of eutrophication have been identified and measured. Kadlec (1983) identified certain conditions resulting from nutrient overenrichment through effluent discharge: the death and uprooting of trees, exotic plant invasion, and presence of specific fungi and bacteria. While nitrogen was consistently reduced over 8 years of study, phosphorus eventually overloaded the system. Wetland recovery was documented after cessation of the effluent discharge.

In detailed literature reviews, Johnston (1989) and Howard-Williams (1985) identify several factors in which nutrient overenrichment likely affects wetland quality:

- **Changes in plant species composition.** Eutrophication may alter plant species dominance in a wetland, affecting factors such as long-term nutrient assimilative capacity or foraging habits by wildlife. Changes in trophic status may favor invasion by exotic plants, which might outcompete native species for space and resources (see Kadlec, 1983). Also, changes in species composition might ultimately affect a wetland's ability to take up other pollutants.

- **Reduced photosynthesis.** A reduction in photosynthesis is a reduction in primary productivity and, consequently, nutrient uptake and processing.

- **Interference with feeding and nutrition of aquatic plants and animals.** In shallow-water wetland systems with long residence times, eutrophic conditions might cause phytoplankton and macroalgal blooms or development of epiphytes, which might interfere with the growth and productivity of wetland plants and microbes.

The majority of research specifically addressing nutrient dynamics in wetlands measures concentrations and pathways of either nitrogen or phosphorus, and deposition of sediment, litter, or other input. Specific techniques for measuring nutrient flows or storage (e.g., soils, plants, detritus) in wetlands appear to be well developed (see van der Valk et al., 1991; Johnston, 1991; Chambers et al., 1992). However, no examples were found that specifically recommend limits or thresholds of nutrients or sediment for a particular type of wetland.

Of the literature reviewed, Johnston (1991) comes closest in designing a table-type matrix to consider a range of nutrient parameters affecting wetland quality (Table 4). Johnston's review focused on wetland effects on the quality of surface water entering ecosystems downstream, and not directly on wetland "health." Nevertheless, the approach to examining the literature for various means and ranges of nutrients in wetlands is useful, and an updated literature review targeted specifically for wetland quality could provide insight into organization of wetlands by region or type.

Table 4. **Matrix considering a range of nutrient parameters affecting wetland quality.**

Wetlands Name	Wetland Type	Enriched with	Location	St	Total N		Total P	
Reedy Cr. Wetland, 1979	Swamp	Sewage	Orlando	FL	53.2	71%	-19.4	-103%
Reedy Cr. Wetland, 1980	Swamp	Sewage	Orlando	FL	75.5	84%	-19.2	-80%
Reedy Cr. Wetland, 1981	Swamp	Sewage	Orlando	FL	74.9	95%	5.72	23%
Reedy Cr. Wetland, 1982	Swamp	Sewage	Orlando	FL	71.0	87%	-11.6	-52%
Reedy Cr. Wetland, 1983	Swamp	Sewage	Orlando	FL	72.6	77%	-17.0	-171%
Reedy Cr. Wetland, 1984	Swamp	Sewage	Orlando	FL	57.5	76%	-2.71	-30%
Reedy Cr. Wetland, 1985	Swamp	Sewage	Orlando	FL	61.7	77%	-2.35	-28%
Thuja peatland, 1976	Bog	Sewage	Bellaire	MI	1.84	75%	1.07	91%
Thuja peatland, 1977	Bog	Sewage	Bellaire	MI	6.75	80%	3.01	88%
Thuja peatland, 1978	Bog	Sewage	Bellaire	MI	9.63	80%	1.58	72%
Thuja peatland, 1979	Bog	Sewage	Bellaire	MI	6.21	77%	1.47	64%
Thuja peatland, 1980	Bog	Sewage	Bellaire	MI	9.07	75%	1.46	65%
Thuja peatland, 1981	Bog	Sewage	Bellaire	MI	4.46	81%	-0.33	-27%
Thuja peatland, 1982	Bog	-	Bellaire	MI	0.76	61%	-0.04	-2%
Thuja peatland, 1983	Bog	-	Bellaire	MI	-	-	-0.03	-1%
Fresh marsh, enriched	Marsh	Sewage	Clermont	FL	-	-	37.1	98%
Water hyacinth marsh	Marsh	Sewage	Gnsville.	FL	-	-	7.7	16%
Boggy Gut Wetland	Marsh	Sewage	H. Head	SC	44.9	83%	12.1	62%
Cattail march	Marsh	Sewage	Brillion	WI	-	-	4.8	68%
Nevin Wetland	Marsh	Hatchery	Madison	WI	15.0	21%	0.11	7%
Mixed hardwood swamp	Swamp	Sewage	Widlwood	FL	-	-	0.79	87%
Cypress Dome	Swamp	Sewage	Gnsville.	FL	11.1	74%	10.4	92%
Cypress-Tupelo Swamp	Swamp	Drainage	Barataria	LA	3.8	26%	1.69	41%
Tupelo Swamp	Swamp	Nutrients	Tar River	NC	-	-	25.1	57%
Arithmetic Mean					10.34	64%	-	-

Currently, the more common techniques for assessing nutrient conditions in wetlands come from development of evaluation indicators of ecological stress and design and engineering of wetlands for water quality enhancement. In particular, Leibowitz and Brown (1990), Mitsch (1992b), Mendelssohn and McKee (1992), and Kent et al. (1992) have identified indicators specifically for wetlands. The techniques identified are designed to evaluate multiple stressors, but include nutrients.

Measurement parameters to be considered for wetlands include the following:

Nitrogen Species Concentrations. Ammonium-N, nitrite-N, and nitrate-N can all be sampled using standardized collection and analytical methods. Sources of N are the water column, wetland detritus, litter and peat, and standing stock of plant biomass. A challenge in considering any nutrient concentration in the context of developing endpoints is the uptake range and variability among wetlands. Examples of the variability found in nitrogen uptake in wetlands include German (1989), who found a 36 percent removal rate of N from an undeveloped Florida wetland, and Rhodes et al. (1985), who documented a 99 percent removal efficiency of riparian forests.

Phosphorous Species Concentrations. Total inorganic phosphorous and total organic phosphorous. Sources are sediment, litter/detritus, microbes, and standing stock biomass. Ranges of P concentrations and uptake are also large depending on wetland type. As examples, Mitsch (1992a) found phosphorous retention capacity of natural wetlands to be 4 to 10 percent efficient, compared to 63 to 96 percent in constructed wetlands. Masscheleyn et al. (1992) compared phosphorous assimilation and release of a bottomland hardwood swamp soil with that of a freshwater marsh and found that the swamp forest soil displayed greater sorption capability. Cooper and Gilliam (1987) estimated that 50 percent of P was removed by runoff from agricultural fields in North Carolina. Chescheir et al. (1991) found that small pumping events of agricultural drainage water into wetlands retained substantial phosphorus concentrations, but that large events documented increased phosphorus concentrations leaving the wetland.

Nitrogen:Phosphorus Ratio. The N:P ratio indicates relative concentration of N and P and aids in determining nutrient availability or limitation within a system. N:P ratios are used in assessing marine and aquatic systems. Given the variability of nutrient limitation within various wetlands, the N:P ratio could be a useful endpoint for comparing impacted wetlands to reference conditions.

Dissolved Oxygen Concentration. The dissolved oxygen (DO) concentration could be used as a eutrophication indicator in wetlands with extended hydroperiods.

Processes as Measurement Parameters

Hydrology and Hydroperiod. Hydrology within wetlands is a critical determinant of wetland condition and nutrient dynamics, but more research is needed to understand hydrological mechanisms in wetlands (Gilliam, 1994; Johnston, 1991; Clausen and Johnson, 1990). The highly transient hydrology of many wetlands makes evaluating long-term nutrient and sediment removal difficult. Clausen and Johnson (1990) determined that lake-level hydrology significantly influenced the fluctuation of nutrients and total suspended solids into and out of an adjacent wetland. Dye tracer studies used to examine water velocity through a wetland found rates to be highly variable; lower velocities resulted in greater residence time and greater pollutant removal (Chescheir et al., 1991b). Phillips et al. (1993) determined that the effect of forested wetlands on water quality depends largely on hydrologic conditions. Also, changes in hydroperiod can have significant effects on nutrient pathways and cycling rates, as well as plant species composition (Leibowitz and Brown, 1990; Mitsch, 1992; Brinson, 1993).

Organic Matter and Sediment Accretion. Sediment accretion refers to the accumulation of both mineral and organic matter in wetlands and provides a good indication of trophic status (Leibowitz and Brown, 1990). Cooper and Gilliam (1987) demonstrated that areas between field edges and perennial streams served as sinks for sediment and phosphorus from the continual runoff deposition.

Nutrient Uptake and Mass Balance. Mass balance measures the difference between nutrient input and output of a system and assigns the difference as uptake by biologiccal, chemical, or physical processes. There are many examples of measuring nutrient uptake in applied wetland research. Kemp et al. (1985) conducted field and laboratory studies to determine decomposition rates and nutrient flux in a Louisiana swamp forest. Twenty-six percent N and 40 percent P were retained by the swamp, mainly from the settlement of particulate matter, and these percentages indicate that the swamp is a long-term sink for N and P via burial of decomposed organic matter and denitrification. However, direct measurement of nutrient cycling is difficult in systems that lack the clearly defined input/output points necessary for mass balance equations or nutrient fluxes between storage compartments (Leibowitz and Brown, 1990; Johnston, 1991). As a potential assessment endpoint, Kent et al. (1992) present nutrient uptake and metabolism as a component in a functional value index for monitoring wetlands. In combination with other metrics of the index, net loss of N or P from a monitored wetland is compared to reference wetland conditions.

Nitrification/Denitrification. Nitrification is a biological process by which atmospheric N is fixed into inorganic N compounds available for biological assimilation. Denitrification is a reverse process by which nitrate is chemically converted into gas and dissipated into atmospheric N. Since many wetlands are nitrogen-limited, this might be a critical measurement parameter to determine a wetland's capacity to handle nitrogen (Hansen et al, 1994). Reddy et al. (1989) have successfully quantified nitrification-denitrification at the plant root-sediment interface in wetland plants.

Biological/Ecological Parameters

Use of Indicator Species to Determine a Nutrient-related Condition. The presence or absence of certain plants, animals, or microbes might be useful in flagging a potentially changing wetland trophic condition. Often, invasive exotic plants might serve as early indicators of elevated nutrient concentrations.

Abundance, Diversity and Species Composition. Wetland plants are reliable indicators of stress, such as eutrophication, and hydrologic condition, and sampling methods are well developed (Leibowitz and Brown, 1990). Marked changes in species richness and composition of plants might indicate an accelerated change resulting from nutrient enrichment. A change in successional status of a wetland might be included as part of measuring changes in species richness. However, plant community succession might occur in combination with other factors, such as sedimentation and changes in hydrology or competition for space (Odum, 1971).

Peak Biomass or Net Primary Productivity. Overall net primary productivity of a wetland can indicate "health" and a relative stage of succession (Mitsch, 1992). Although nutrients often stimulate plant growth and production, this factor, used in conjunction with plant species diversity, might indicate a change in trophic status resulting from nutrient enrichment.

Leaf Area, Solar Transmittance and Greenness. Changes in characteristics of wetland canopy plant species might also be an indicator of eutrophic conditions (Leibowitz and Brown,, 1990). If positively correlated, then remote sensing techniques might be useful in assessing large wetland areas. Nutrient flux has been examined at the landscape level using remote sensing and GIS tools (Correll et al., 1992).

Aquatic Microbial Community Structure. Microbial communities are linked to nutrient cycling and litter decomposition (Leibowitz and Brown, 1990).

State of the Science—Wetland Monitoring and Assessment

Of the literature reviewed, several authors provide detailed discussions of monitoring wetlands (Kent et al, 1992; Adamus, 1992; Leibowitz and Brown, 1990). Assessment of nutrients is generally included as a component in monitoring programs, but is often combined with other metrics to assess general conditions. Consequently, the focus is often qualitative and therefore lacks the precision needed to clearly assess nutrient impacts. Landers and Knuth (1991) found that wetland monitoring to determine the impact of nutrients on lakes in USEPA Region 5 was lacking. Nixon and Lee (1986) recommend that a combination of carefully selected field studies, with establishment of microcosm studies, serve as the basis for monitoring and increased understanding of wetland functions.

Various approaches to field monitoring were identified by:

- Geographic region
- Wetland type
- Accessibility and ease of study (for cost-effectiveness)
- Size and visibility (enlisting public interest and support)
- Availability of supporting information (building on previous research)

The use of reference wetlands or conditions is an important consideration in assessing impacts. Establishing a series of reference wetlands to serve as a database within a region would allow comparison for a range of parameters (Figure 3). For example, a parameter (such as nutrient load) for wetlands undergoing assessment could be compared to those reference conditions having a "normal" range. Assessed wetlands falling outside the normal range may be considered impaired for that specific parameter.

Within EPA, the STORET database has some limited information on wetlands. In particular, Michigan and Florida have wetland sampling stations, but Michigan contains fish tissue data only. Florida has sampling information on wetlands, including nutrient concentrations (Figure 4). However, the distribution of sampling stations emphasizes the uneven spatial coverage on a national scale.

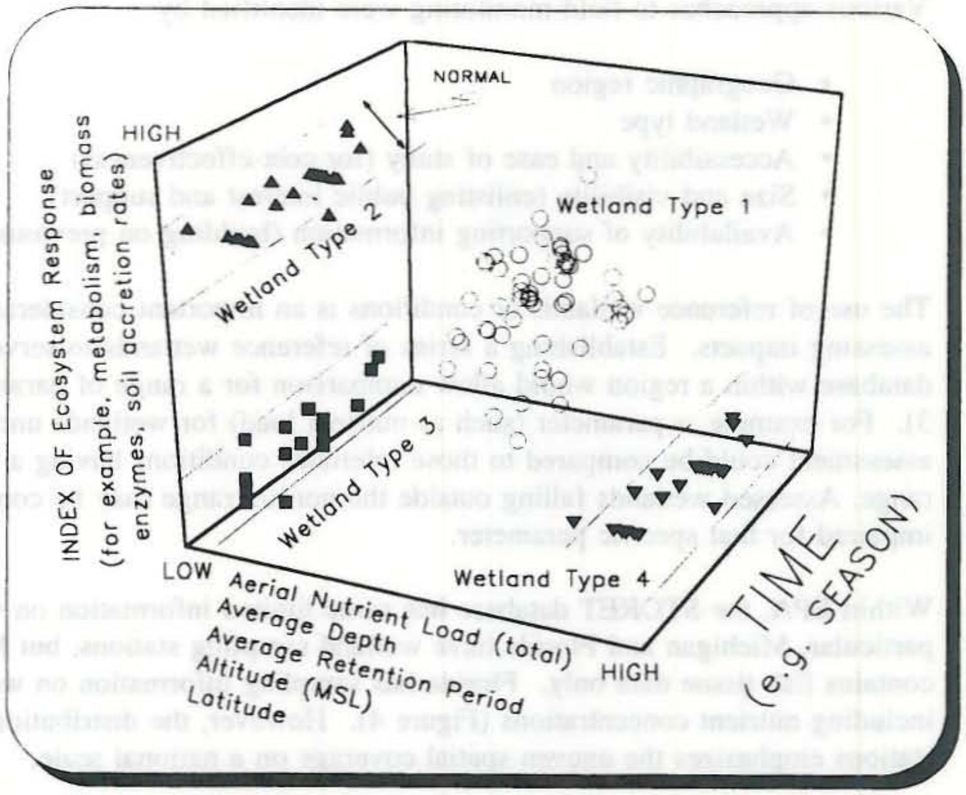

Figure 3. Hypothetical example of how a national or regional database of reference wetlands could be used to assess nutrient impacts. "Wetland Type 1, 2, 3 & 4" refers to different hypothetical wetland classifications that display a range of "normal" characteristics for a given parameter. The parallel lines bordering the wetland types represent hypothetical confidence intervals for a "normal" response. Driving functions (X axis) and response variables (Y axis) can represent single or multiple variables. Assessed wetlands for a given variable falling outside a "normal" range would indicate a potentially impaired condition.

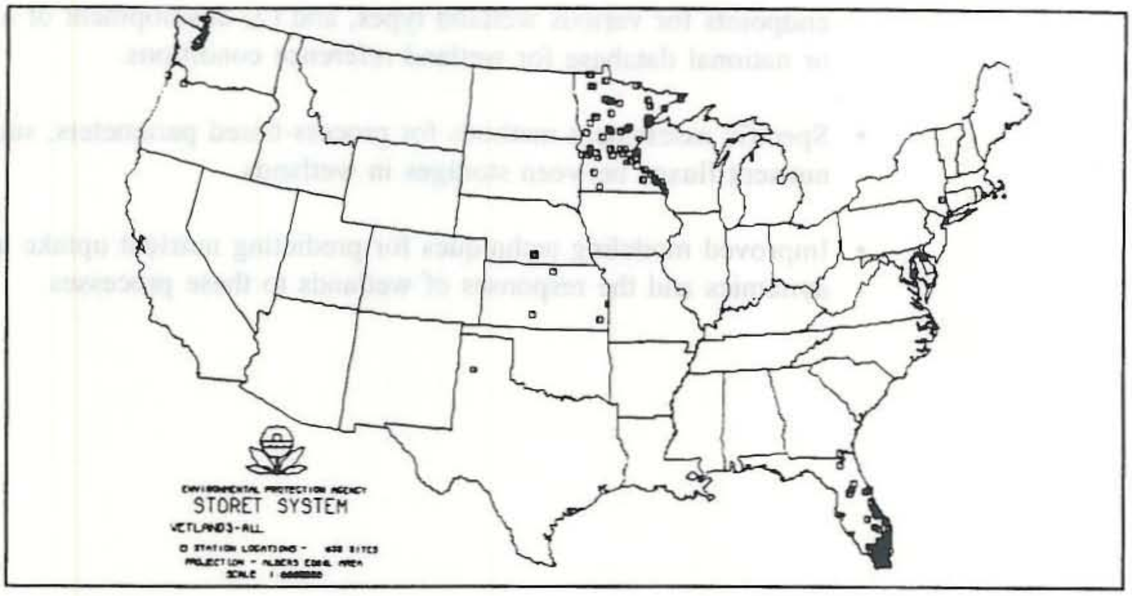

Figure 4. Distribution of sampling stations in the USEPA STORET database that are wetlands. The figure emphasizes the sparse national distribution of wetlands data.

Models that compute or predict nutrient loadings to wetlands (transport models for groundwater or surface water flow) are well developed and have been reasonably successful. Assessment of inputs/loadings using a variety of loading models (such as SWMM and HSPF) can be used to predict nutrient and sediment loads to wetlands. Loadings can also be estimated from local monitoring studies. As a general class, models that are capable of predicting *wetland responses to nutrient loads* are not well developed. However, some traditional water quality models, such as CEQUAL-W2, and WASP5, have been used for evaluating wetlands and hydrodynamic models, such as EFDC, are being applied to wetlands in Florida to assess hydrologic response (Hamrick, Virginia Institute of Marine Science, personal communication).

Summary

Because of the high degree of variability between and within wetlands, it is possible that a regional strategy for wetlands is a more prudent course than a detailed national strategys. Additionally, it is clear that more research is needed to understand the specific mechanisms that affect wetland quality. These research needs include the following:

- Additional monitoring and improved documentation of the effects of nutrient overenrichment on various wetland types. This may involve (1) synthesis and organization of wetland research to determine whether certain parameters have sufficient data for developing assessment endpoints for various wetland types, and (2) development of a regional or national database for wetland reference conditions.

- Specific assessment methods for process-based parameters, such as nutrient fluxes between storages in wetlands.

- Improved modeling techniques for predicting nutrient uptake and dynamics and the responses of wetlands to these processes.

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