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Nutrient Criteria Technical Guidance Manual

Estuarine and Coastal Marine Waters

U.S. Environmental Protection Agency

Nutrient Criteria
Technical Guidance Manual

Estuarine and Coastal Marine Waters

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Disclaimer

This manual provides technical guidance to States, Indian Tribes, and other authorized jurisdictions to establish water quality criteria and standards under the Clean Water Act (CWA), to protect aquatic life from acute and chronic effects of nutrient overenrichment. Under the CWA, States and Indian Tribes are to establish water quality criteria to protect designated uses. State and Indian Tribal decisionmakers retain the discretion to adopt approaches on a case-by-case basis that differ from this guidance when appropriate and scientifically defensible. Although this manual constitutes EPA's scientific recommendations regarding ambient concentrations of nutrients that protect resource quality and aquatic life, it does not substitute for the CWA or EPA's regulations; nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, States, Indian Tribes, or the regulated community, and might not apply to a particular situation or circumstance. EPA may change this guidance in the future.

Cover Photograph: Somewhere on the Chesapeake Bay. Supplied by David Flemer as a duplicate copy from the Chesapeake Biological Laboratory Photo Archives, University of Maryland; date unknown but earlier than 1972.

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CONTRIBUTORS

Jessica Barrera (Hispanic Association of Colleges and Universities/University of Miami)
Robert Cantilli (U.S. Environmental Protection Agency)
Ifeyinwa Davis (U.S. Environmental Protection Agency)*
Edward Dettmann (U.S. Environmental Protection Agency)
Jen Fisher (University of Georgia)
David Flemer (U.S. Environmental Protection Agency)*
Thomas Gardner (U.S. Environmental Protection Agency)*
George Gibson (U.S. Environmental Protection Agency)*
Debbi Hart (U.S. Environmental Protection Agency)
James Latimer (U.S. Environmental Protection Agency)
Scott Libby (Battelle)*
Greg Smith (GLEC, Inc.)
CarolAnn Siciliano (U.S. Environmental Protection Agency)*
Jack Word (MEC Analytical Systems)*

*Denotes primary authors

WORK GROUP MEMBERS

Ifeyinwa Davis (U.S. Environmental Protection Agency)
David Flemer (U.S. Environmental Protection Agency)
John Fox (U.S. Environmental Protection Agency)
George Gibson (U.S. Environmental Protection Agency)
Debbi Hart (U.S. Environmental Protection Agency)
Suzanne Bricker (National Oceanic and Atmospheric Administration)
Dorothy Leonard (National Oceanic and Atmospheric Administration)
Scott Libby (Battelle)
Greg Smith (GLEC, Inc.)
Jack Word (MEC Analytical Systems)

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Estuaries and coastal waters are a diverse suite of ecosystems, and differences in methods and approaches are to be expected. This and subsequent manuals are not intended to be singular, one-time publications. As experience accumulates, future editions will be prepared to reflect new understanding.

FOREWORD

This manual is intended for State/Tribal and Federal agency personnel actively engaged in water resource management data collection, assessment, planning, and project implementation. Consequently, it incorporates both a scientific rationale and enough of the "nuts and bolts" of nutrient criteria development and management to help both initiates and those experienced in water resource management.

These nutrient criteria development and management efforts are directed at anthropogenic sources. Inherent "natural" background levels are not and should not be subject to management. Our responsibility is to abate human-caused eutrophication in estuaries and coastal or "near coastal" (out to about 20 nautical miles) marine waters.

To distinguish between natural background enrichment and human impacts, it is necessary to identify localities that experience minimal human influence. Ambient nutrient measurements at these sites may then be compared to similar sites that do experience human influences. The difference in nutrient measurements is the difference between a reference site and a test site. A reference condition is a collection of measurements from several reference sites that incorporates a central tendency statistic.

Because of differences in geologic parent material, climate, and geography, reference conditions are different from one region to another. Similarly, waterbodies, especially estuaries, often respond differently to nutrient inputs. Lakes and reservoirs are different from streams and rivers, and estuaries and coastal marine waters have characteristics different from both. Criteria have to be designed for particular waterbody types and the regions in which they lie.

The primary variables of concern in criteria development are two causal enrichment variables: total phosphorus (TP) and total nitrogen (TN). These nutrients are essential to algal and plant production and are the base of the food chain that supports all other life in the system. Also, two initial response variables usually are the first indicators of biological growth reaction to enrichment. One is chlorophyll *a*, which indicates photosynthesis and biomass production; the other is Secchi depth, a measure of water clarity or a measure of turbidity, reflecting planktonic growth in the absence of inorganic suspended material. In many marine and estuarine instances dissolved oxygen concentration (DO) and macrophyte growth and density are also important measures and, where indicated, may be included as initial response variables. Other measures can also be used, but these have been selected by EPA as of primary concern.

Nutrient criteria consist of judicious incorporation of present **reference condition information** about the primary variables, together with a **knowledge of historical conditions** and trends in the nutrient quality of the resource. These two factors, possibly augmented by **data extrapolations or models**, are analyzed objectively by a **panel of regional specialists** well versed in the biology, physics, and chemistry of the systems of concern. The criteria are also evaluated with respect to the **possible consequences of their implementation on downstream receiving waters**. All of these elements are required for the development of a nutrient criterion.

With this information, the status of a given water resource can be determined, management plans can be made, and management efforts can be evaluated.

The best possible understanding of the physical, chemical, and biological interrelationships in the environment is important in nutrient criteria development and the subsequent management response. However, effective nutrient criteria can and should be developed even in the absence of an in-depth scientific investigation of the ecological processing of nutrients in the estuarine and marine environment. An adequate number of proximal reference sites and current knowledge of the system are sufficient to initiate criteria development and proposed management responses. A conservative, environmentally responsible start can be made toward alleviating nutrient pollution, subject to adjustment as more scientific knowledge is obtained and verified.

The reference condition approach to criteria development was peer reviewed by the USEPA Science Advisory Board in 1990 and 1994 and judged to be scientifically defensible. It is also likely to be the most cost-effective approach.

EXECUTIVE SUMMARY

This manual is designed for use by State, Tribal, and Federal water resource managers as they address the cultural enrichment of their waters in conjunction with the EPA National Nutrient Criteria Program. It is intended to provide a stepwise sequence of actions leading to the development of nutrient criteria for estuarine and near-coastal marine waters to be used in correcting this overenrichment problem.

The premise of the National Nutrient Criteria Program is that many, if not most, of our nation's estuarine and coastal waters are moderately to severely polluted by excessive nutrients (Bricker et al.1999), especially nitrogen and phosphorus. This nutrient pollution affects not only the biotic integrity of the waters and the decline of valuable fish and shellfish, it has the potential to cause harm to the public health through hazardous algal blooms and the propagation of waterborne diseases. To address this problem, EPA uses a regionalized, waterbody type specific approach to the development of nutrient criteria or benchmarks for management decisionmaking. These criteria are based on the measurement of the most natural (or least impacted by human development) waters of a given type in a given area reflecting the condition to be expected in that region if human impacts are not a factor or are at least minimized. The variables of specific concern are total phosphorus and total nitrogen as causal variables, algal biomass (e.g., chlorophyll *a* for phytoplankton and ash-free dry weight for macroalgae), and water clarity (e.g., Secchi depth) as early response variables. In waters that already experience hypoxia, dissolved oxygen should be added as a response variable. EPA encourages States and Tribes to consider additional response indicators such as seagrasses and algal species composition.

This natural ambient background or "reference condition" is an important element of the nutrient criteria to be developed. The other elements are: an understanding of the historical status and trend of the water resource to help put the reference condition in perspective; models of the nutrient data to help better understand historical and present information and to project future consequences; concern and attention to the effects of any criteria development on downstream receiving waters; and the objective compilation and assessment of all of this information by a skilled body of regional experts...the "Regional Technical Assistance Group" or RTAG. The regional criteria so developed are guidelines the States and Tribes of the continental United States can use as they prepare their own criteria and standards for the improvement and protection of the nation's coastal waters.

The first of the actions needed to reach this criteria objective is the organization and utilization in each EPA Region of an RTAG consisting of specialists from State and Federal natural resource management agencies versed in the management and scientific principles most appropriate to that region and those waters. These are water resource managers, oceanographers, chemists, land use specialists, biologists, estuarine ecologists, statisticians, and similar local civil service experts employed by the State or Federal government. Academicians, special interest groups, and environmental group representatives are also important participants in the criteria development process and may assist the RTAG in its efforts.

The first requirement of the RTAG is the review and refinement of ecoregional determinations as most appropriate to the area. These are the geographic boundaries surrounding the similar estuarine and coastal marine waters for which the criteria will apply. They are based on the EPA Ecoregion concept and incorporate attendant coastal Provinces, both of which are based on geographic and geologic similarities of landforms and parent material. The importance of this regionalization is the effort to deal with waters all having a similar inherent background nutrient loading and response characteristic. Once the regional boundaries and perhaps subregional divisions are completed, the RTAG investigates the physical classification of the waters into similar estuaries or coastal reaches or embayments for criteria development. In many instances the estuaries may be unique and require specific criteria.

Within the classification scheme developed, reference sites are identified as those areas suffering the least cultural development or impact, and the compilation of similar reference sites becomes a reference condition. The manual describes the scientific rationale for the variables selected, the dynamics of the receiving waters, and potentially confounding physical and chemical interrelationships influencing criteria development. It also describes sampling and analytical techniques for data gathering and processing to develop the reference conditions as well as several options for the compiling of this information. These include: (1) recognition and measurement of an excellent water body of ideal nutrient water quality with the aim of preserving this state; (2) in situ reference site determinations for moderately degraded waters; (3) hind casting for historical information from past higher nutrient quality conditions to determine the reference condition when no reference sites remain; (4) use of loading estimations from reference quality subestuarine tributary systems and projection to the estuary; and (5) options for establishing coastal nutrient reference conditions including a Nutrient Criteria Program pilot demonstration project.

Once the reference condition(s) has been determined, the RTAG then addresses the historical perspective; considers the need for models to project future consequences; considers the potential effect on receiving waters; and employs its own good judgment in collectively determining the appropriate criteria values for each of the variables to protect the waters of concern and their designated uses. A procedure is also suggested to equate the multiple criteria variables in a comprehensive dimensionless index score. The manual concludes with a chapter on model development and applications to the criteria program, and a chapter describing the application and implementation of nutrient criteria with emphasis on EPA Standards and Monitoring Divisions and a description of a comprehensive ten step sequential technique for water resource management.

This comprehensive progression from data collection to reference condition determination to criteria development and management responses, is intended to help users achieve the restoration and protection of the nutrient water quality of the nation’s estuarine and near-coastal marine water resources.

CHAPTER 1

Introduction and Objectives

Background
Definition of Estuaries and Coastal Systems
Nature of the Nutrient Overenrichment Problem in Estuarine and Coastal Marine Waters

Man has had a long and intimate association with the sea. It has borne his commerce and brought food to his nets; its tides and storms have shaped the coast where his great cities have grown; the broad estuaries have provided safe harbors for his ships; and the rhythm of its tides has taught him the mathematics and science with which he now reaches for the stars (U.S. Department of the Interior 1969).

1.1 BACKGROUND

Nutrient overenrichment is a major cause of water pollution in the United States. The link between eutrophication—the overenrichment of surface waters with plant nutrients—and public health risks has long been presumed. However, human health concerns such as (1) *Escherichia coli* and the spread of disease in sewage-enriched waters; (2) trihalomethanes in chlorine-treated eutrophic reservoirs; (3) the incidence of nutrient-stimulated hazardous algal blooms in eutrophic estuarine surface waters with suspected attendant human illnesses, including recent *Pfiesteria* investigations; and (4) the relationship of phytoplankton blooms in nutrient-enriched coastal waters of Bangladesh to cholera outbreaks (Scientific American, December 1998) all suggest that overenrichment pollution is not only an aesthetic, aquatic community problem, but also a public health problem.

The purpose of this document is to provide scientifically defensible technical guidance to assist States, authorized Tribes, and other governmental entities in developing numeric nutrient criteria for estuaries and coastal waters under the authority of the Clean Water Act (CWA), Section 304a. The objective is to reduce the anthropogenic component of nutrient overenrichment to levels that restore beneficial uses (i.e., described as designated uses by the CWA), or to prevent nutrient pollution in the first place. The primary users of this manual are State/Tribal and Federal agency water quality management specialists and related interest groups. The manual is intended to facilitate an understanding of cause-and-effect relationships in these complex systems and serve as a guide for nutrient criteria development, a resource of technical information, a summary of the scientific literature, and a brief technical account of the ecological structure and function of estuaries and coastal waters to facilitate an understanding of these complex systems.

To combat the nutrient enrichment problem and other water quality problems, EPA published the Clean Water Action Plan, a presidential initiative, in February 1998. Building on this initiative, EPA developed a report entitled National Strategy for the Development of Regional Nutrient Criteria (U.S. EPA 1998a). Criteria form the scientific basis, or yardstick, for ensuring that a desired result will occur because of a particular form of environmental stress, in this case nutrient overenrichment. The strategic report outlines a framework for development of waterbody type-specific technical guidance with emphasis on the reference condition approach that can be used to assess nutrient status and develop region-specific numeric nutrient criteria. This technical guidance builds on that strategy and provides guidance for nutrient criteria development for estuaries and coastal waters. Because estuaries and coastal waters lie at the interface of the land and include various ecoregions and their rivers, this manual departs somewhat from the freshwater manuals (e.g., Lakes and Reservoirs, EPA-822-B00-001, and Rivers and Streams, EPA-822-B-00-002; also available on the EPA web site: www.epa.gov/ost/standards/nutrient.html in PDF format) and considers both land-based ecoregions and coastal ocean provinces as the geographic framework. The freshwater nutrient guidance manuals used the ecoregion and subecoregion as the predominant geographic operational units.

Because of differing geographic and climatic conditions among the East, Gulf, and West Coasts, uniform national criteria for estuarine and coastal waters are not appropriate; they should be developed at the State, regional, or individual waterbody levels. Figures 1-1a,b illustrate the pertinent ecoregions (including geologic province) of the continental United States associated with coastal and estuarine waters. In some cases, multiple criteria may be required for large systems with extended physical gradients. This manual therefore does not provide guidance on how to set nationwide criteria, but provides State water resource quality managers with guidance on how to set nutrient criteria themselves relative to EPA regional criteria. This approach is in contrast to toxic chemical criteria, which tend toward single national numbers with appropriate modifiers (e.g., water hardness for metals). It explores some approaches to classification of estuaries and coastal shelf systems. The ability to develop useful classification schemes is still in a highly developmental stage and needs considerable improvement. The manual describes a minimum set of variables that are recommended for criteria development and describes methods for developing appropriate values for these criteria. It also provides information on sampling, monitoring, data processing, modeling, and approaches to implementation and management responses.

1.2 DEFINITION OF ESTUARIES AND COASTAL SYSTEMS

It is important to have a clear view of the ecosystems that are the focus of this manual. The term “estuary” has been defined in several ways. For example, a classical definition of estuaries focuses on selected physical features—e.g., “semi-enclosed coastal waterbodies which have a free connection to the open sea and within which sea water is measurably diluted with freshwater derived from the land” (Pritchard 1967) (see Kjerfve 1989 for expanded definition). This definition is limited because it does not capture the diversity of shallow coastal ecosystems today often lumped under the rubric of estuary. For example, one might include tidal rivers, embayments, lagoons, coastal river plumes, and river-dominated coastal indentations that many consider the archetype of estuary. To accommodate the full range of diversity, the classical definition should be expanded to include the role of tides in mixing, sporadic freshwater input (e.g., Laguna Madre, TX), coastal mixing near large rivers (e.g., Mississippi and Columbia Rivers), and tropical and semitropical estuaries where evaporation may influence circulation. Also, reef-building organisms (e.g., oysters and coral reefs) and wetlands (e.g., coastal marshes) influence ecological structure and function in important ways, so that biology has a role in the definition.

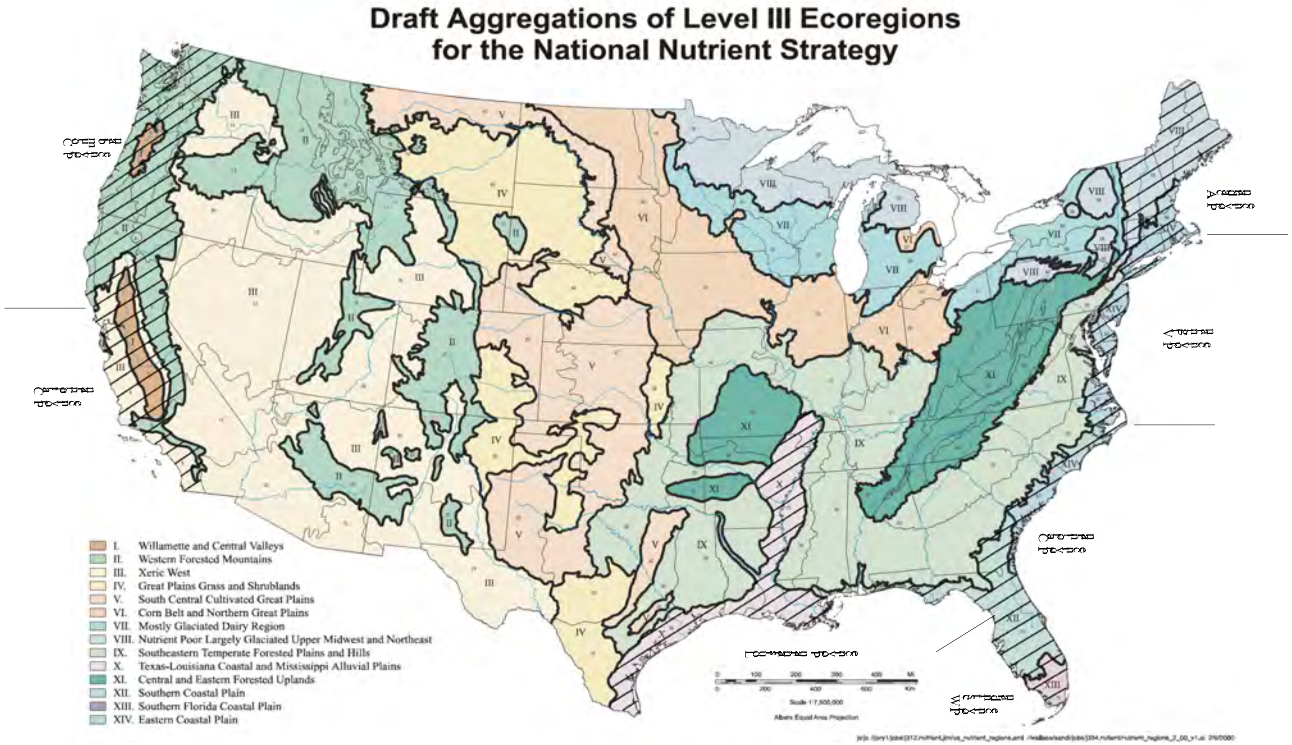

Figure 1-1a. Draft aggregation of Level III ecoregions for the National Nutrient Strategy illustrating those areas most related to coastal and estuarine criteria development.

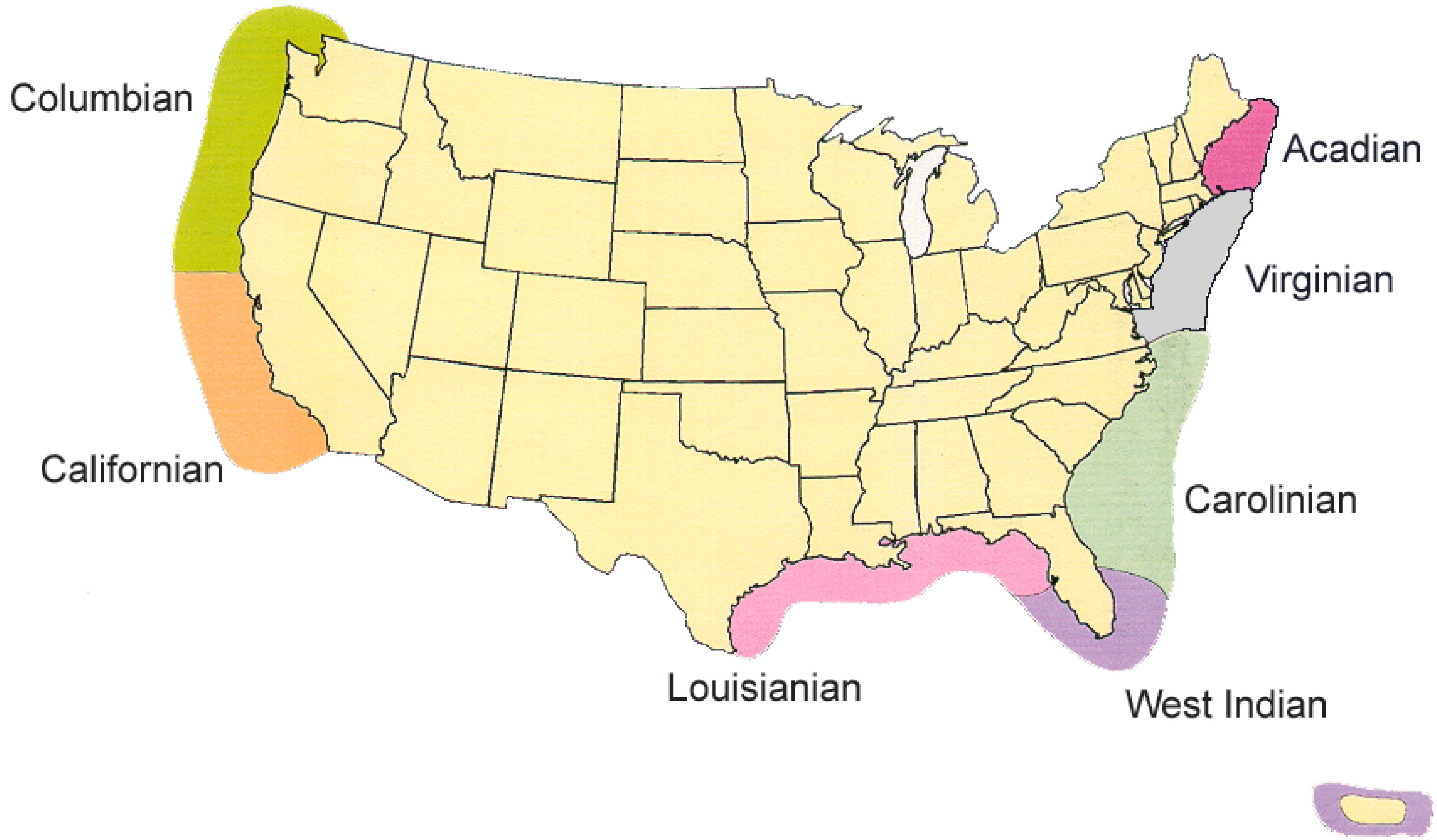

Figure 1-1b. Coastal provinces.

As will be shown, water depth plays a role in the relative importance of sediment-water column fluxes of materials, including nutrients. These features paint a picture of high ecosystem diversity, where prediction of susceptibility to nutrient overenrichment is still a scientific challenge and often requires a great deal of site-specific information. It is because of this diverse response that reference conditions are a part of nutrient criteria development.

Coastal waters are defined in this manual as those marine systems that lie between the mean highwater mark of the coastal baseline and the shelf break, or approximately 20 nautical miles offshore when the continental shelf is extensive. This area will hereafter be referred to as coastal or near-coastal waters. Most States have legal jurisdiction out to the 3-nautical-mile limit. However, coastal oceanic processes beyond this limit may influence nutrient loading and system susceptibility within the 3-mile zone.

1.3 NATURE OF THE NUTRIENT OVERENRICHMENT PROBLEM IN ESTUARINE AND COASTAL MARINE WATERS

Scope and Magnitude of the Problem

Nutrient overenrichment problems are perhaps the oldest water quality problems created by humankind (Vollenweider 1992) and have antecedents that extend into biblical history. The basic cause of nutrient problems in estuaries and nearshore coastal waters is the enrichment of freshwater with nitrogen (N) and phosphorus (P) on its way to the sea and by direct inputs within tidal systems. Eutrophication, an aspect of nutrient overenrichment, is portrayed in Figure 1-2. In recent decades, atmospheric deposition of N has been an important contributing factor in some coastal ecosystems (Vitousek et al. 1997, Paerl and Whitall 1999).

In U.S. coastal waters, nutrient overenrichment is a common thread that ties together a diverse suite of coastal problems such as red tides, fish kills, some marine mammal deaths, outbreaks of shellfish poisonings, loss of seagrass and bottom shellfish habitats, coral reef destruction, and hypoxia and anoxia now experienced as the Gulf of Mexico's "dead zone" (NRC 2000, Rabalais et al. 1991). Additionally, recent evidence suggests that nutrient enrichment can exacerbate human health effects (Colwell 1996). These symptoms of nutrient overenrichment often are preceded by primary symptoms (e.g., an increase in the rate of organic matter supply, changes in algal dominance, and loss of water clarity) followed by one or more secondary symptoms listed above (Figure 1-3). Nixon (1995) defined eutrophication as an increase in the rate of supply of organic matter to a waterbody. In this manual, nutrient overenrichment is defined as the anthropogenic addition of nutrients, in addition to any natural processes, causing adverse effects or impairments to beneficial uses of a waterbody. The scientific literature still uses overenrichment and eutrophication as synonyms. The terms have different meanings, however, because eutrophication is a natural process in freshwater lakes and presumably in coastal marine waters. An argument can be made that nutrient stress on coral reefs can cause a loss of symbiotic algae (i.e., dinoflagellates), resulting in loss of organic matter and death of the coral colony, a condition not consistent with eutrophication in the strict sense.

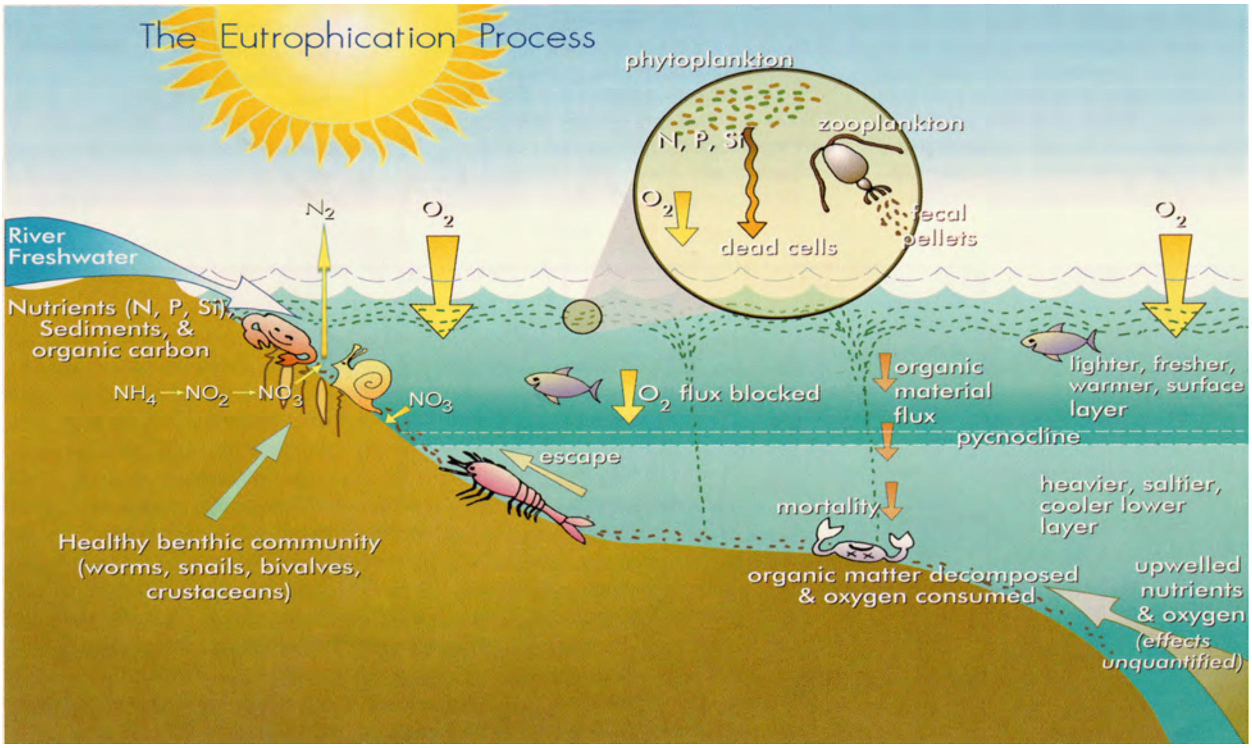

Figure 1-2. The eutrophication process. Eutrophication occurs when organic matter increases in an ecosystem. Eutrophication can lead to hypoxia when decaying organic matter on the seafloor depletes oxygen, and the replenishment of the oxygen is blocked by stratification. The flux of organic matter to the bottom is fueled by nutrients carried by riverflow or, possibly, from upwelling that stimulates growth of phytoplankton algae. This flux consists of dead algal cells together with fecal pellets from grazing zooplankton. Sediment coupled nitrification-denitrification is shown as well as NO_3 transport into sediments when it can be identified. Source: modified from CENR 2000.

Despite several decades of progress in reducing nutrient pollution from waste treatment facilities, nutrient runoff from farms and metropolitan areas, often far inland, has gone unabated or actually increased (The Pew Oceans Commission: www.pewoceans.org; Marine Pollution in the United States: Significant Accomplishments, Future Challenges, 2001; Mitsch et al. 2001). Interestingly, early marine scientists considered nutrients as a resource, not a problem (Brandt 1901), and reflected on ways to fertilize coastal seas to increase biological production. In fact, in the 1890s Brandt concluded that N was the primary limiting nutrient in marine waters and that nitrification and denitrification were important processes in the N cycle.

Nutrient overenrichment of estuaries and nearshore coastal waters from human-based causes is now recognized as a national problem on the basis of CWA 305b reports from coastal States that list waters whose use or uses are impaired; these figures vary from 25% to 50% of the waters surveyed. The National Oceanic and Atmospheric Administration's (NOAA) National Estuarine Eutrophication Assessment (Bricker et al. 1999) indicated that about 60% of the estuaries out of 138 surveyed exhibited moderate to serious overenrichment conditions. Nutrient overenrichment of coastal seas now has international implications (NRC 2000) and is especially well documented for coastal systems of Europe

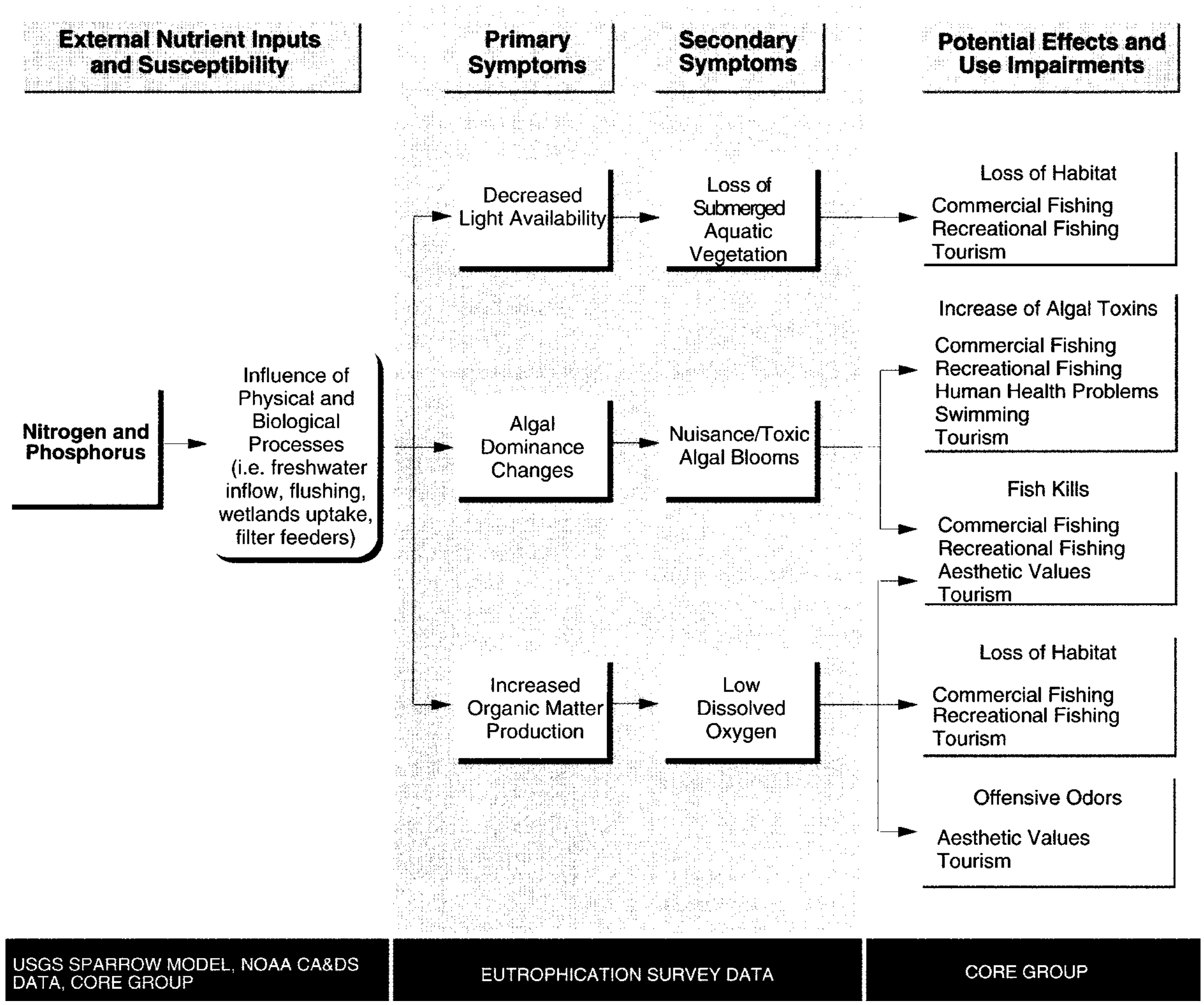

Figure 1-3. Expanded nutrient enrichment model. Source: Bricker et al. 1999.

(Justic 1987, Jansson and Dahlberg 1999, Gerlach 1990, cited in Patsch and Radach 1997, Radach 1992), Australia (McComb and Humphries 1992), and Japan (Okaichi 1997). The problem is likely underreported for developing nations. Currently, the European Union has initiated an effort to develop nutrient criteria for surrounding fresh and marine waters (personal communication, U. Claussen, German Environmental Protection Agency).

In summary, these examples demonstrate that both N and P may limit phytoplankton biomass production depending on season, location along the salinity gradient, and other factors. Nutrient overenrichment problems have been present from early history, especially in estuaries downstream of cities, and the nutrient criteria development approach that follows is a new element in EPA's effort to address these longstanding problems.

1.4 THE NUTRIENT CRITERIA DEVELOPMENT PROCESS

Preliminary Steps

It is impossible to recommend a single national criterion applicable to all estuaries. Natural enrichment varies throughout the geographic and geological regions of the country, and these subdivisions must be considered in the development of appropriate nutrient criteria. For example, “drowned river estuaries” may exhibit a range of inherent or ambient natural enrichment conditions from less than 1.3 µM TP in the thin soils of the Northeast to 2.6 µM TP in the delta regions of the South and Gulf of Mexico.

Although lakes and reservoirs and streams and rivers may be subdivided by classes, allowing reference conditions for each class and facilitating cost-effective criteria development for nutrient management, except for barrier island estuaries and mangrove bays in a given area this is not feasible for estuaries. A major distinction between this manual and the one prepared for lakes and reservoirs is that estuarine and coastal marine waters tend to be far more unique, and development of individual waterbody criteria rather than for classes of waterbodies (such as glacial temperate lakes) is a greater likelihood. Also, estuaries will likely require classification by residence time or subdivision by salinity or density gradients.

Consequently, it will be necessary in many cases to determine the natural ambient background nutrient condition for each estuary or coastal area so that the eutrophication caused by human development and abuse can be addressed. Human-caused eutrophication is the focus of this manual, but the development of nutrient criteria, frequently on a waterbody-specific basis, will require another major distinction for coastal marine criteria development. In the absence of comparable reference waterbodies, the historical record of inherent and cultural enrichment may be particularly significant to developing reference conditions of a particular estuary or coastal reach. The historical perspective is always important to criteria development, but in this instance it may also be essential to reference condition determination.

An outline of the recommended process for coastal and estuarine criteria development is as follows: (1) Investigation of historical information to reveal the nutrient quality in the past and to deduce the ambient, natural nutrient levels associated with a period of lesser cultural eutrophication, (2) determination of present-day or historical reference conditions for the waterbody segment based on the least affected sites remaining, such as areas of minimally developed shoreline, of least intrusive use, fed by those tributaries of least developed watersheds, (3) use of loading and hydrologic models to best understand the density and flow gradients, including tides, affecting the nutrient concentrations, (4) the best interpretation of this information by the regional specialists and Regional Technical Assistance Group (RTAG) responsible for developing the criteria, and (5) consideration of the consequences of any proposed criteria on the coastal marine waters that ultimately receive these nutrients to ensure that the developed criteria provide for the attainment and maintenance of these coastal uses. This concept, as illustrated in Figure 1-4, is the basis for the National Nutrient Criteria Program and is explained throughout this text.

In deriving the reference condition (Figure 1-5), the extreme values of hypereutrophy on one hand and pristine or presettlement conditions on the other can be estimated from monitoring, historical records,

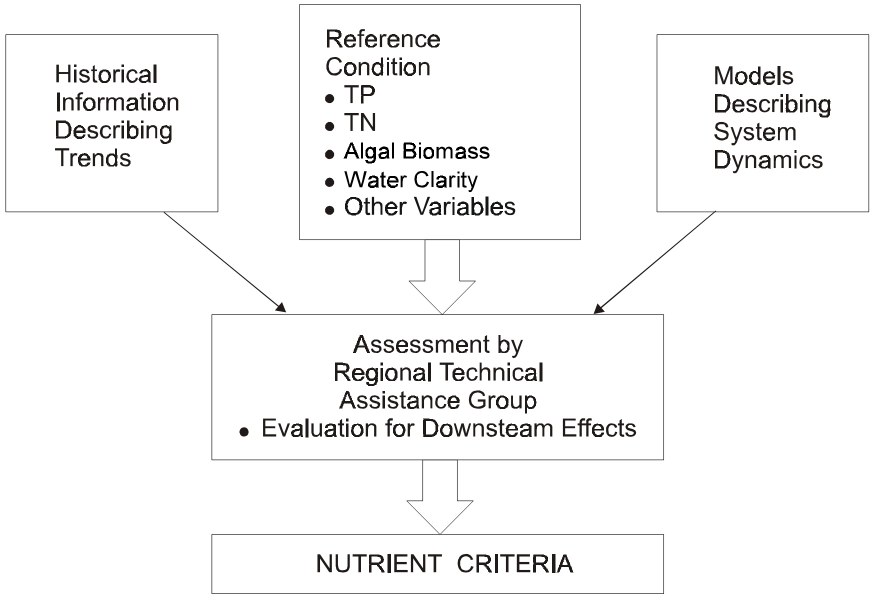

Figure 1-4. Elements of nutrient criteria development and their relationships in the process.

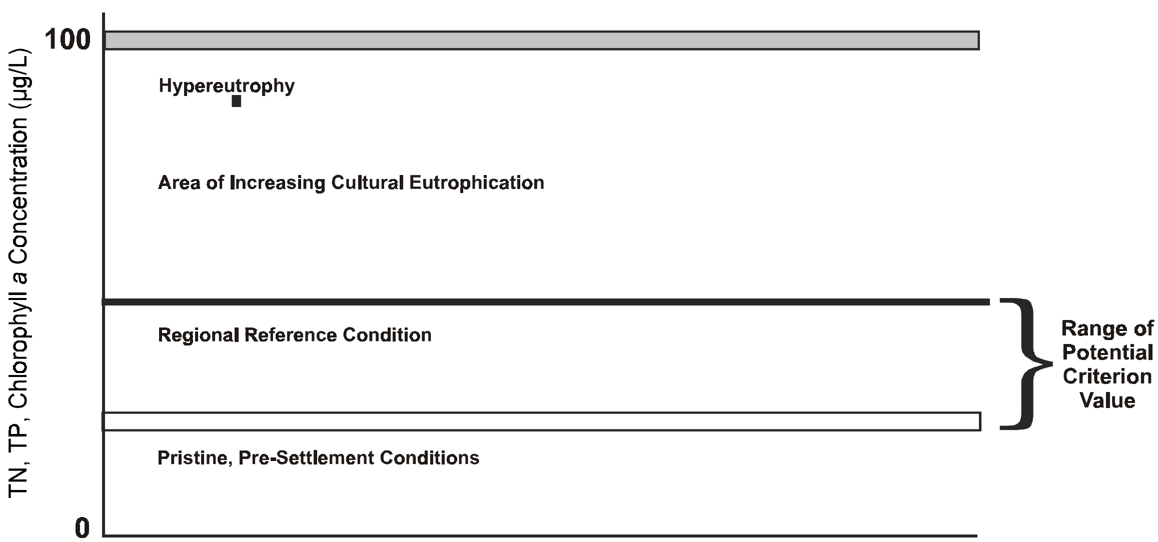

Figure 1-5. Derivation of the reference condition and the National Nutrient Criteria Program using TP, TN, and chlorophyll *a* as example variables. Clarity or Secchi depth would be on a reversed scale. Protectivity nutrient criteria should be between pristine conditions and present reference conditions, i.e., the most "natural" attainable.

and paleoecological determinations. The reference condition and the derived criteria are scientifically based estimates expected to be a present-day approximation of the natural state of the waters approaching but not likely duplicating pristine conditions. They include a conscious decision to use areas of least human impact as indicators of low cultural eutrophication. A measure of practical judgment is also necessary where scientific methods and data are not adequate.

The use of minimally impacted reference sites has been adapted from biological criteria development and is endorsed by EPA's Science Advisory Board (U.S. EPA 1992). Minimal impacts provide a baseline that should protect beneficial uses of the Nation's waters. The term "minimally impacted" implies a high percentage of conditions in reference locations and a low percentage of conditions in all locations (i.e., some enrichment is allowed, but not enough to cause adverse local effects or adverse coastal receiving water effects). The upper end of the data distribution range from reference sites represents the threshold of a reference condition, whereas lower percentiles represent high-quality conditions that may not or cannot be achieved. The upper 25th percentile represents an appropriate margin of safety to add to the minimum threshold, excludes the effect of spurious outliers, and serves as a sufficiently protective value. Where sufficient data are available, comparison and statistical analysis of causal and response variables can help determine effect thresholds and further refine reference conditions (see Figure 6-2).

Establishing the reference condition is but one element of the criteria development process. Reference condition values are appropriately modified on the basis of examination of the historical record (most important), modeling, expert judgment, and consideration of downstream effects.

Strategy for Reducing Human-Based Eutrophication

Six key elements are associated with the strategy for reducing human-based eutrophication (U.S. EPA 1998):

- EPA believes that nutrient criteria need to be established on an individual estuarine or coastal water system basis and must be appropriate to each waterbody type. They should not consist of a single set of national numbers or values because there is simply too much natural variation from one part of the country to another. Similarly, the expression of nutrient enrichment and its measurement vary from one waterbody type to another. For example, streams do not respond to phosphorus and nitrogen in the same way that lakes, estuaries or coastal waters.

- Consequently, EPA has prepared guidance for these criteria on a waterbody-type and region-specific basis. With detailed manuals available for data gathering, criteria development, and management response, the goal is for States and Tribes to develop criteria to help them deal with nutrient overenrichment of their waters and protect designated uses.

- To help achieve this goal, the Agency has initiated a system of EPA regional technical and financial support operations, each led by a Regional Nutrient Coordinator—a specialist responsible for providing the help and guidance necessary for States or Tribes in his or her region to develop and adopt criteria. These coordinators are guided and assisted in their duties by a team of inter-Agency and intra-Agency specialists from EPA headquarters. This team provides both technical and financial support to the RTAGs created by these coordinators so the job can be completed and communication maintained between the policymaking in headquarters and the actual environmental management in the regions.

- EPA will develop basic ecoregional coastal ocean province nutrient criteria for waterbody types. The Regional Teams and States/Tribes can use these values to develop criteria protective of designated uses; the Agency also may use these values if it elects to promulgate criteria for a State or Tribe. These criteria, once adopted by States and authorized Tribes into water quality standards, will have value in two contexts: (1) as decisionmaking benchmarks for management planning and assessment and (2) as the basis of National Pollution Discharge Elimination System (NPDES) permit limits and Total Maximum Daily Load (TMDL) target values. The Standards and Health Protection Division of the EPA Office of Water will be developing implementation guidance for these latter applications.

- EPA plans to provide sufficient information for States and Tribes to begin adopting nutrient standards by 2003.

- States/Tribes are expected to monitor and evaluate the effectiveness of nutrient management programs implemented on the basis of the nutrient criteria. EPA intends the criteria guidance to reflect the "natural," minimally impaired condition of a given estuary or coastal water or the class of these systems, respectively. Once water quality standards are established for nutrients on the basis of these criteria, the relative success or failure of any management effort, either protection or remediation, can be evaluated.

Thus, the six elements of the National Nutrient Criteria Program describe a process that encompasses taking measurements of the collective water resources of an area, establishing nutrient criteria for evaluating the discrete waters within that region, assessing individual waterbodies against these criteria and associated standards, designing and implementing the appropriate management, and, finally, evaluating its relative success.

Nutrient Criteria Development Process

The activities that compose the nutrient criteria development process are listed below in the order generally followed, and the subsequent chapters of this document follow this sequence. Figure 1-6 presents a schematic illustration of the process with parallel, corresponding chapter headings.

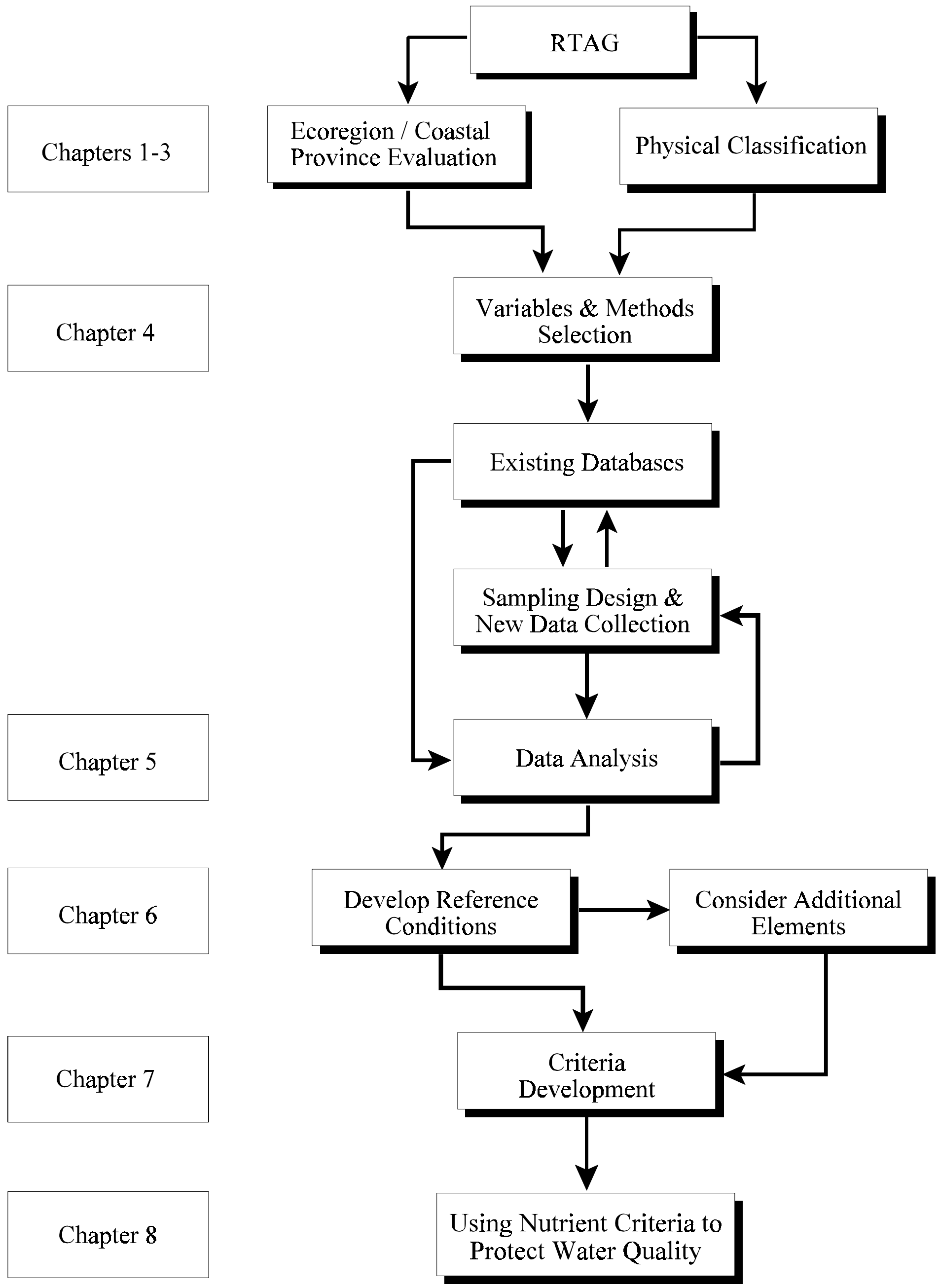

Figure 1-6. Flowchart of the nutrient criteria development process.

■ ***Preliminary Steps for Criteria Development (Chapter 1)***

Establishment of Regional Technical Assistance Groups

The Regional Nutrient Coordinator in each EPA multistate region should obtain the involvement of key specialists (e.g., estuarine and marine ecologists, water resource managers, oceanographers, stream and wetland ecologists, water chemists, and agricultural and land-use specialists) with respect to the waterbodies of concern. These experts should be recruited from other Federal and State agencies.

Experts from academia and industry may serve as technical advisors on an as need basis but not official voting members of the RTAG.

Particular Federal agencies of interest are the U.S. Geological Survey (USGS); Natural Resources Conservation Service (NRCS); National Oceanic and Atmospheric Administration (NOAA); National Marine Fisheries Service (NMFS) and National Ocean Survey (NOS); U.S. Department of the Interior; National Park Service (NPS); National Seashores; the U.S. Fish and Wildlife Service (USFWS); U.S. Army Corps of Engineers (USACE); and, in certain areas of the country the Bureau of Land Management (BLM) or special government agencies such as river basin commissions and inter-State commissions. Similarly, for information and education activities, the National Sea Grant Program and for agriculture, the USDA Cooperative Extension Service are valuable resources.

State agencies with responsibilities relevant to this effort are variously named, but are commonly referred to as Department of Natural Resources, Department of Water Resources, Department of the Environment, Department of Environmental Management, Fisheries and Wildlife Management, State Department of Agriculture, State Department of Forestry, or other land-use management agencies. Most state land-grant universities have faculty talent important to natural resource and nutrient management, and almost all colleges and universities have applied science faculty with research interests and talents appropriate to this initiative.

In selecting participants for the group, diverse expertise is an obvious prerequisite, but willingness to cooperate in the group effort, integrity, and a lack of a strong alternative interest are also important factors to consider in selecting these essential people who must make collective and sometimes difficult determinations.

The experts chosen will constitute the RTAG, which will be responsible for developing more refined nutrient criteria guidance for their respective estuaries and coastal waters. The RTAG should be large enough to have the necessary breadth of experience, but small enough to effectively debate and resolve serious scientific and management issues. A membership of about 30 approaches an unwieldy size, although that number may initially be necessary to maintain an effective working group of half that size. EPA expects that States and authorized Tribes will use the information developed by the RTAGs when adopting nutrient criteria into their water quality standards. The RTAG is intended to be composed of scientists and resource managers from Federal agencies and their State counterparts. The RTAG should not delegate its responsibility with the private sector. The perspectives of private citizens, academicians, and special interest groups are important, and these and other members of the public may attend RTAG meetings and offer opinions when invited, but the final deliberations and decisions are the responsibility of the Federal and State members of the RTAG—the States when adopting nutrient criteria into their water quality standards, and the EPA when determining whether to approve or disapprove such criteria. They must also be able to meet and debate the issues without undue outside influence.

As a matter of policy, however, EPA encourages the RTAGs to regularly provide access and reports to the public. The meetings should generally be open to the public and the schedule of those meetings published in the local newspapers. At a minimum, RTAGs are encouraged to hold regular “stakeholders” meetings so that environmental, industrial, and other interests may participate via a separate public forum associated with responding to the group’s efforts. It is important that citizens and public groups be involved, and any significant determinations of the RTAG should include a public session at which a current account of activities and determinations is presented and comments acknowledged and considered. In addition, where specific land uses or practices are addressed, those property owners, farmers, fishermen, or other involved parties should be consulted in the deliberation and decisionmaking process.

It is reasonable to expect the RTAG to meet monthly, or at least quarterly, with working assignments and assessments conducted between these meetings. To coordinate activities among the 10 RTAGS, and with the National Nutrients Team, regular conference calls are recommended. At these sessions, new developments in the Program, technical innovations and experiences, budgets, and policy evolutions will be conveyed and discussed. In the same context, an annual meeting of all Regional Nutrient Coordinators, State representatives, and involved Federal agencies should be held each spring in or near Washington, DC. At this meeting, major technical reports are presented by specialists and issues significant to the Program are discussed.

The composition and coordination discussed above are intended to establish the shortest possible line of communication between the State, region, and national Program staff members to promote a rapid but reasoned response to changing issues and techniques affecting nutrient management of our waters. This format is also designed to be responsive to the water resource user community without becoming a part of user conflicts.

Delineation of Nutrient Ecoregions/Coastal Province Appropriate to the Development of Criteria
The initial step in this process has been taken through the creation of a national nutrient ecoregion map consisting of 14 North American subdivisions of the coterminous United States (Figure 1-1). These are aggregations of Level III ecoregions revised by Omernik (2000). Alaska, Hawaii, and the U.S. Territories will be subdivided into nutrient ecoregions later, with the advice and assistance of those States and their governments.

The initial responsibility of each RTAG will be to evaluate the present ecoregional map with respect to variability on the basis of detailed observations and data available from the States and Tribes in that EPA region. This preliminary assessment will further depend on the additional nutrient water quality data obtained by those States. The databases, especially with respect to selected reference sites, may be used to refine the initial boundaries of the map in each EPA region.

EPA recognizes that the coastal margins of these ecoregions will be of the greatest concern to the States developing estuarine and coastal marine criteria, but in some instances watersheds will extend a considerable distance inland. In any case, the consistent application of the ecoregion concept facilitates both upstream and inland coordination by the RTAGs and States and integrates the coastal efforts with rivers, lakes, and streams.

■ ***Scientific Basis (Chapter 2)***

Chapter 2 emphasizes the role of physical processes interacting with biological processes in modulating the expression of nutrient enrichment effects and the potential of inaccurately assessing cause and effects in developing management plans.

■ ***Physical Classification (Chapter 3)***

The next step in evaluating the data is to devise a classification scheme for rationally subdividing the population of estuarine and coastal marine waters in the State or Tribal territory. Because identification of overenrichment is the objective of nutrient criteria development, trophic classification per se should be avoided, as should any classification based on levels of human development. Physical characteristics independent of most human-caused enrichment sources are far more appropriate.

However, as stated above, many estuarine and some coastal marine areas will probably require individual attention and development of reference conditions that are site-specific or at least specific to waterbody segments. Within these contiguous segments, the reference stations should have similar residence time, salinity, general water chemistry characteristics, depth, and grain size or bottom type.

Once the waters have been subdivided and classified, it is important to select the key indicator variables of concern and determine how much information is available on the enrichment status of these stations.

■ ***Selection of Indicator Variables (Chapter 4)***

Chapters 4 through 7 describe the variables for which EPA anticipates developing 304 (a) criteria for nutrients in estuaries and coastal waters and how they should be sampled, preserved, and analyzed. Although a wide variety of indicator variables may be possible, this technical manual describes development of numerical criteria for total phosphorus (TP) and total nitrogen (TN) as primary nutrient causal variables of eutrophication, and measures of algal biomass (e.g., chlorophyll *a* for phytoplankton and ash-free dry weight for macroalgae) and a measure of water clarity (e.g., Secchi depth or electronic photometers) as primary variables of eutrophic response. In those systems that have hypoxia or anoxia problems, dissolved oxygen also should be added as a primary response variable. States or Tribes may elect to include other indicators as well, but the four primary variables and dissolved oxygen as indicated are recommended as the essential indicators. Other variables are loss of seagrass/submerged aquatic vegetation (SAV), benthic macroinfauna, iron, and silica as well as other indicators of primary and secondary productivity.

State and Federal agency records are the basis for an initial data search. In many States, water quality information resides in more than one agency. For example, Maryland has a Department of Natural Resources and a Department of the Environment, both of which retain water quality records. To compound the data search problem further, States may also have pertinent data sets in their Department of Fisheries and Department of Public Health. It is wise to initiate the search for information with calls and questionnaires to colleagues in the State or Tribal agencies likely to be involved so an appropriate list of contacts and data sets can be compiled. In doing so, regional Federal agencies should not be overlooked either. These include the agencies described above in the selection of RTAG members.

- ***Nutrient Data Collection and Assessment (Chapter 5)***

EPA has initiated the data collection and assessment process by screening the existing STORET and ODES databases for information on lakes, reservoirs, streams, estuaries and coastal waters with respect to the four initial parameters, and dissolved oxygen where appropriate (see reference to Chapter 4 above). These primary variables were originally selected for robustness and conservativeness of estimation; however, the preliminary screening of the STORET data revealed that these measurements are also relatively abundant in the database.

Although this is an entirely appropriate starting point for nutrient criteria development, States and Tribes are not required to confine their investigations and data selection to only these variables. States and Tribes are encouraged to select additional measures that contribute to the best assessment of the enrichment of their regional waters and protect designated uses. In particular, it is advisable to use both *causal indicators* and *response indicators* as mentioned above.

Combining nutrient and biological system response information will yield the most definitive and comprehensive criteria. To use only causal or only response variables in the criteria puts the State or Tribe in jeopardy of not protecting the designated uses. For example, a highly enriched estuarine system with a rapid flushing rate may appear to be in attainment when only the biota and dissolved oxygen are measured, but the load of nutrients being delivered downstream in its coastal discharge plume is degrading the receiving waters. Using a balanced combination of both causal and response variables in the criteria, together with careful attention to tidal and seasonal variability, should mitigate against false-positive or false-negative results.

Chapters 4 and 5 both discuss proper sampling, preservation, and analysis of samples. Seasonality, spatial distribution of sample sites, composite versus discrete sampling, and fixed station versus stratified random sampling are also explored.

Establishing an Appropriate Database

Review of Historical Information. Historical information, including sediment core analysis, is important to establish a perspective on the condition of a given waterbody. Has its condition changed radically in recent years? Is the system stable over time? What is the variability? Has there been a trend up or down in trophic condition? Only an assessment of the historical record can provide these answers. Without this information, the manager risks setting reference conditions and subsequent criteria on the basis of present condition alone, which may in fact be a degraded state. Valid historical information places the current information in its proper perspective and is particularly important to coastal and estuarine nutrient criteria development because of the difficulty in establishing classes and the scarcity of reference waterbodies.

Data Screening. The first step in assessing historical or current data is to review the material to determine its suitability to support nutrient criteria development. Anecdotal information and observations are valuable, but the sources must be carefully considered. Fishermen's accounts, local sport-fishing news stories, and observational logs of scientific field crews are all legitimate sources of information, but they are subject to different levels of scrutiny before a trend is determined. The same applies to databases. Nutrient information gathered for identifying failing wastewater treatment plants cannot be assessed in the same light as similar data collected to determine overall water quality or trophic state. The analytical procedures used, type of sampling design and equipment, and sample preservation are other variables that must also be considered in any data review and compilation. Once this screening is done, the compiled data may be sorted according to station location, physical characteristics, relative depth, time, and date, and then analyzed for the establishment of reference conditions.

■ ***Establishing Reference Conditions (Chapter 6)***

Candidate reference locations can be determined from compiled data with the help of regional experts familiar with the waters of the area. Classification will be an important first step and should be based on physical characteristics of the waterbodies, including morphology, geological origin, and hydrologic factors such as residence time, flow characteristics, tidal processes, and freshwater-saltwater interchanges. An estuary may then be subclassified into lower, medium, and upper salinity regimes. Specialists can also help to select the least culturally impacted sites or stations within each area.

Three candidate approaches are recommended for development of tidal estuarine reference conditions. Two more approaches use loading information within the fluvial watershed. A sixth approach is described for coastal waters. Where several replicate systems occur, each classified as near-pristine based on recent data (e.g., past 10 years), then one can apply a frequency distribution approach, and this manual recommends that the upper 75th percentile be used as a starting point. If some minor nutrient enrichment is present, then all the data would be considered and, in this case, the lower 25th percentile is suggested. In the case of significant nutrient-based environmental degradation, where reference sites cannot be identified from current monitoring data, then hind-casting with ambient data is recommended. There are three approaches: (1) empirical in situ data analysis, (2) sediment core or paleoecological analysis, and (3) model hind-casting. Interpretation of this approach is potentially sensitive to confounding by physical factors (e.g., freshwater inflows). The watershed approach is load-based. Here, one attempts to locate a relatively nutrient-unenriched tributary, or stream segment, that is approximately representative of the watershed, and extrapolate the nutrient load for the entire watershed. This can be done empirically or, preferably, with models. The coastal approach focuses on changes in the nutrient regime of estuarine plumes and waters some distance from such plumes. An index approach is described that accounts for variability and facilitates identification of natural enrichment (e.g., upwelling). Long-term monitoring is required to distinguish anthropogenic effects from natural variability.

■ ***Criteria Development (Chapter 7)***

Nutrient Criteria Components

The move from data review and data gathering to criteria development involves a sequence of five interrelated elements:

- Examination of the historical record or paleoecological evidence for evidence of a trend.

- Determination of a reference condition using one of several alternative approaches. Remember that the reference condition, however derived, is only part of the criteria development process.

- Use of empirical modeling or surrogate data sets in some instances where insufficient information exists. This may be the case especially in estuaries with insufficient hydrological data, or significantly developed or modified watersheds.

- Objective and comprehensive interpretation of all of this information by a panel of specialists selected for this purpose (i.e., the RTAG). These experts should have established regional reputations and expertise in a variety of complementary fields such as oceanography, estuarine ecology, nutrient chemistry, and water resource and fisheries management.

- Finally, the criterion developed for each variable should reflect the optimal nutrient condition for the waterbody in the absence of cultural impacts and protect the designated use of that waterbody. Second, it must be reviewed to ensure that the proposed level does not entail adverse nutrient loadings to downstream waterbodies. In designating uses for a waterbody and developing criteria to protect those uses, the State or Tribe must consider the water quality standards of downstream waters (40 CFR 131.10 (b)). This concern extends all the way to coastal waters, but in practice the immediate downstream receiving waters are the area of greatest attention for the resource manager. The criteria must provide for the attainment and maintenance of standards in downstream waters. A criterion for that estuary or subclass of estuary will not protect downstream water quality standards, it should be revised accordingly.

Once the initial criteria (either Regional or State/Tribal) have been selected, they can be verified and calibrated by testing the sampling and analytical methods and criteria values against waterbodies of known conditions. This ensures that the system operates as expected. This calibration can be accomplished either by field trials or by use of an existing database of assured quality. This process may lead to refinements of either the techniques or the criteria.

Criteria are developed for more than one parameter. For example, all reference sites of a given class may be determined to manifest characteristics of a particular level for TP concentration, TN concentration, algal biomass,and water clarity. These four measures, and dissolved oxygen as appropriate, become the basis for criteria appropriate to optimal nutrient quality and the protection of designated uses. The policy for criteria attainment will be developed by the State or Tribe in consultation with EPA.

When the estuarine or coastal marine segment in question reveals high TN and TP concentrations, but not the expected high algal biomass and low water clarity, further investigation is indicated before deciding whether criteria have been met. Flushing rates, inorganic turbidity, water color, or toxins may be additional factors influencing the condition of the estuary.

Assessing Attainment With Criteria
An action level then is established for the nutrient criteria that have been selected for each indicator variable. The list includes two causal variables (TN and TP) and three primary response variables (e.g., when dissolved oxygen problems occur this will add an additional variable to the response variables. Failure to meet either of the causal criteria should be sufficient to prompt action. However, if the causal criteria are met, but some combination of response criteria are not met, there should be some form of decision making protocol to resolve the question of whether the waters in question meet the nutrient criteria. There are two approaches to this:

- Establish a decisionmaking rule equating all of the criteria such as the frequency and duration of exceedences and the critical combination of response variables requisite for action

- Establish an index that accomplishes the same result by inserting the data into an equation that relates the multiple variables in a nondimensional comprehensive score

■ ***Management Response (Chapter 8)***
There are a variety of possible management responses to the overenrichment problem identified by nutrient criteria. Chapter 8 describes some regulatory and nonregulatory processes that involve the application of nutrient criteria. It also presents a 10-step process that allows the resource manager to use these approaches to improve water resource condition. The emphasis is on developing a scientifically responsible, practical, and cost-effective management plan.

The chapter also describes three basic categories that encompass all management activities: education, funding, and regulation. It closes with the admonition to always carefully evaluate the success of the management project, report results, and continue monitoring the status of the water resource.

■ ***Model Applications (Chapter 9)***
A variety of empirical and theoretical models are described and discussed, and two specific illustrations of the application of models to estuarine nutrient management are presented.

■ ***Appendices***
A number of appendices supplement the primary text.

It should be noted that completion of each step may not be required of all water quality managers. Many State or Tribal water quality agencies may have already completed the identification of designated uses, classified their estuaries and coastal waters, or established monitoring programs and/or databases for their programs and therefore can bypass those steps. This manual is meant to be comprehensive in the sense that all of the criteria development steps are described; however, the process can be adapted to suit existing water quality programs.

In any event, a responsible nutrient management plan should meet three conditions. First, the plan and its component elements must be *scientifically defensible*; otherwise it might lead to well-intentioned management actions that are unnecessary or harmful. This is like the admonition to physicians, “above all do no harm.” Second, effective nutrient management must strive to be *economically feasible*. The public and local interests are more likely to support approaches that provide meaningful benefit compared with their cost. Finally, these approaches should be *practical and acceptable to the communities involved*. The approaches should address appropriate social and political issues, such as conflicts that might exist between public agencies and landowners, agricultural or other resource users, or between commercial fishermen and recreationists and environmental or industrial groups. Any management plan may fail if these three general elements are not sufficiently addressed, and it is almost certain to fail if they are all ignored.

CHAPTER 2

Scientific Basis for Estuarine and Coastal Waters Quantitative Nutrient Criteria

Controlling the Right Nutrients
Physical Processes, Salinity, Algal Net Primary Production
Nutrient Loads and Concentrations: Interpretation of Effects
Physical-Chemical Processes and Dissolved Oxygen Deficiency
Nutrient Overenrichment Effects and Important Biological Resources
Concluding Statement Regarding Nitrogen and Phosphorus

2.1 INTRODUCTION

At the turn of the last century nitrogen and phosphorus were prized as the fuel that fed the great engine of marine production. Today they are seen as lethal pollutants leading to toxic blooms and suffocation. Just as weeds are fine plants growing in the wrong place, nitrogen and phosphorus are essential chemicals that can get into the wrong places at the wrong times. We should not lose sight of their critical role in sustaining production (Nixon 2000).

Purpose and Overview

This chapter describes the scientific basis for development of nutrient criteria for estuarine and coastal waters. A number of scientific issues are addressed to develop nutrient criteria. Water quality managers can improve their application of science to nutrient criteria development if they consider these systems' large latitudinal and climatic range, high ecosystem-based variability, complexity, diversity, and broad range in land-sea margin human activities. These features suggest a high degree of system individuality, especially at larger scales. These features occur because estuaries and coastal waters are transitional ecosystems buffeted by variable landward-based freshwater input volumes and constituents, influences of oceanic provinces, and human disturbances, including nutrient enrichment, superimposed on these natural regimes (Figure 2-1). Even in a relatively narrow section of coastline, the ecosystem diversity and variability may be quite large. These characteristics challenge the investigator to develop useful predictive schemes. Some progress has been achieved, but areas of important uncertainties are also noted.

Coastal areas, including estuaries and upwelling regions, account for only 10% of the ocean by area but at least 25% of the ocean's primary productivity and upwards of 95% of the world's estimated fishery yield (Walsh 1988). These areas are also an important organic carbon sink of atmospheric CO_2. In addition, coastal counties account for only 17% of the U.S. landmass, but their population exceeds 141 million. Thus, more than half of the Nation's population lives in less than one-fifth of the total area, and this trend is expected to grow (NRC 2000). These statistics underpin the fact that estuarine and open coastal areas have, and continue to show, stress from human activities including nutrient pollution, as noted in Chapter 1. These demographics argue strongly for a scientific understanding of how nutrients flux through estuarine and nearshore coastal ecosystems and impair water quality use.

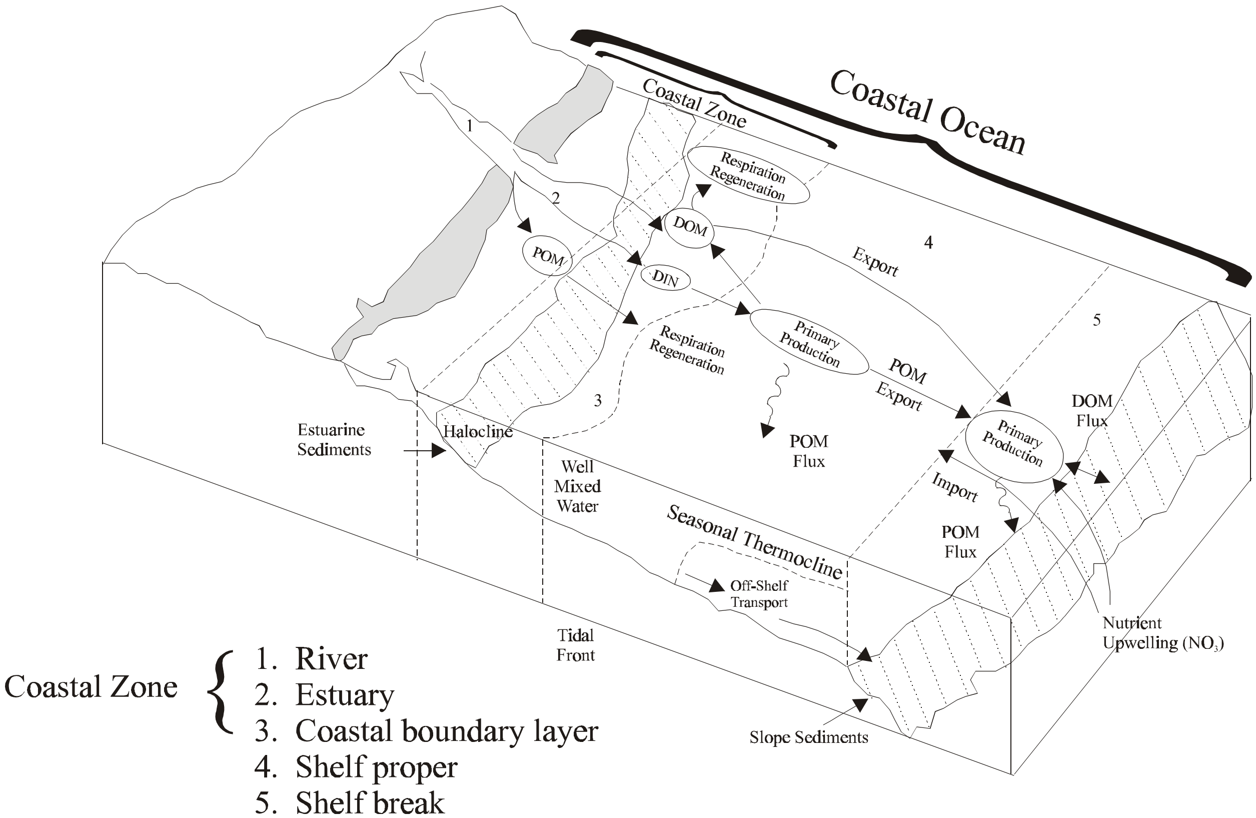

Figure 2-1. Idealized scheme defining the coastal ocean and the coastal zone, with some key biochemical fluxes linking land and sea and pelagic and benthic processes. The latter are not to scale. Source: Alongi 1998.

Some Important Nutrient-Related Scientific Issues

A large number of issues with a scientific component may complicate nutrient criteria development in estuaries and open coastal waters. Some of the more important issues are summarized below and are discussed in more detail later in this and following chapters. These issues illustrate how science underpins nutrient criteria development.

Determination of which nutrients are causing the problem is critical. In some cases, this will be known with considerable assurance, but in others further study is advisable. Without such knowledge, it is difficult to develop reliable nutrient criteria. It is important to understand at what scale one is discussing the question of nutrient limitation. The term “nutrient limitation” is often used quite loosely and without formal definition (Howarth 1988). For phytoplankton, Howarth makes the following points and argues that it matters a great deal which of the following questions is being addressed:

- Limitation of the growth rate of phytoplankton populations currently in a waterbody
- Limitation of the potential rate of net primary production, allowing for possible shifts in the composition of phytoplankton species
- Limitation of net ecosystem production

Each of these definitions can be considered “correct,” but each addresses different questions. Clearly, phytoplankton growing in an oligotrophic environment may be adapted to maximize growth rates under low nutrient conditions, as evidenced by their organic nutrient composition approaching the Redfield atomic ratio of C:N:P of 106:16:1 (Redfield 1958; Goldman et al. 1979). An increase in nutrient supply would likely shift species composition to those adapted to the higher nutrient regime, and net primary production would potentially increase. Thus, it is plausible that potential net primary production can be nutrient limited even if the growth rate of currently dominant phytoplankton species is not. If a nutrient is added to a system and net primary production increases, the system is considered to have been nutrient limited regardless of whether the species composition has shifted. Similarly, when a nutrient criterion is exceeded, enrichment is presumed to be of concern even if the system’s productivity has not responded. This is the definition used in this manual for addressing effects of nutrient overenrichment.

Why not use net ecosystem production as the preferred definition, as the ecosystem is the level of system organization that might seem most relevant? For example, the ecosystem was the level of the whole-lake experiments that contributed to defining P as the primary limiting nutrient for north temperate freshwater lakes (Schindler 1977). Net ecosystem production equals gross primary production in excess of total ecosystem respiration. For the biomass of an isolated ecosystem to be maintained, the net ecosystem organic production must equal or slightly exceed 0. Imports of organic matter can augment the internal net production. Howarth argues that it is difficult to relate nutrient supplies to net ecosystem production because the respiration term is sensitive to allochthonous input of organic matter as well as internal net production. So, for practical reasons, net primary production, which is directly related to algal biomass production, is the preferred measure of nutrient limitation.

The import of organic matter, especially in estuaries, can lead to water quality problems (e.g., hypoxia). Organic matter input from sewage was historically a major source of organic carbon that drove aquatic systems toward dissolved oxygen (DO) deficiency through direct microbial heterotrophic activity (Capper 1983). However, the input of nutrients, whether in organic form followed by recycling or inorganic form with direct nutrient uptake, is what stimulates potential phytoplankton biomass production, and this organic matter may contribute to symptoms of nutrient overenrichment identified in Chapter 1.

It is frequently difficult to distinguish natural ecosystem variability associated with net primary production from that induced by anthropogenic stress, especially nutrient enrichment, which often is a consequence of variability in physical processes. An example is the difficulty, even with a 50-year record, in distinguishing the effects of freshwater flow of the Susquehanna River and co-linear effects of nutrient loading on Chesapeake Bay phytoplankton biomass production indicated by chlorophyll *a* (chl *a*) concentrations (Harding and Perry 1997). Such indeterminancy is a condition that water quality managers must contend with, and argues for broad scientific input.

It is important to understand nutrient load and ecological response relationships because of the need to conduct load allocations (e.g., total maximum daily loads, TMDLs), and it may be necessary to perform some management triage when systems are poised along a gradient of risk and there are too many systems to treat in a timely fashion. Also, as explained later, ecological responses to nutrient enrichment may be quantitatively related to nutrient load rather than complexity in physical transport and mixing. The relationship between N load and seagrass recovery in Tampa Bay, FL, is an example of where nutrient load was predictive but concentration of N was not (Greening et al. 1997).

As discussed in Chapter 3, classification of estuaries and coastal shelf systems at large scale (e.g., Chesapeake Bay versus Delaware Bay) is in an early state of development with regard to predicting many nutrient enrichment effects. This is because of the relatively high degree of ecosystem individuality at the larger scale, where comparability among systems tends to breaks down. The result is that scientific generalizations are usually circumscribed with consequences that may lead to higher management costs. Resource managers and environmental scientists should work together to improve predictability of nutrient enrichment effects because there are too many systems in the Nation to study all estuaries and coastal systems comprehensively.

These ecosystems exhibit a notable degree of process asymmetry and lag in responses, which means that a stress at one location and time may show up as a response at another location and time. Additionally, different mechanisms may result in a similar response (Malone et al. 1999). This type of behavior enhances the tendency to confound cause-and-effect relationships.

Along the same lines, conceptual models for estuaries (and coastal waters) in particular are still evolving. These models suggest that systems modulate stresses so that a single stress does not necessarily result in a single response (Cloern 2001) (Figure 2-2). This fact alone contributes to ecological uncertainty in load-response relationships. Conceptual models help define expectations of cause-and-effect relationships and degree of nutrient-caused impairment, and refine hypotheses. Conceptual models should be a standard tool for water quality managers.

Antecedent conditions are important. This can be understood in terms of whether enough factors are present at the right place and time to lead to an integrated response, such as a dinoflagellate bloom. Such conditions resemble nonlinear dynamics, which may be a major constraint to prediction of effects. Also, estuaries and nearshore coastal waters are subject to episodic events, which injects considerable uncertainty into predictions (e.g., Tropical Storm Agnes impacted Chesapeake Bay in June 1972: Davis and Laird 1976). A relatively large database is often required to determine when effects of such major events have reached a new steady state.

Estuaries and nearshore coastal waters naturally vary in the type, abundance, and geographical coverage of biological communities at risk to nutrient overenrichment, largely because of habitat differences. This variability is partially offset by salinity, which tends to “normalize” biotic community distributions (Kinne 1964). When ambient historical data are unavailable or sediment cores are ineffective in

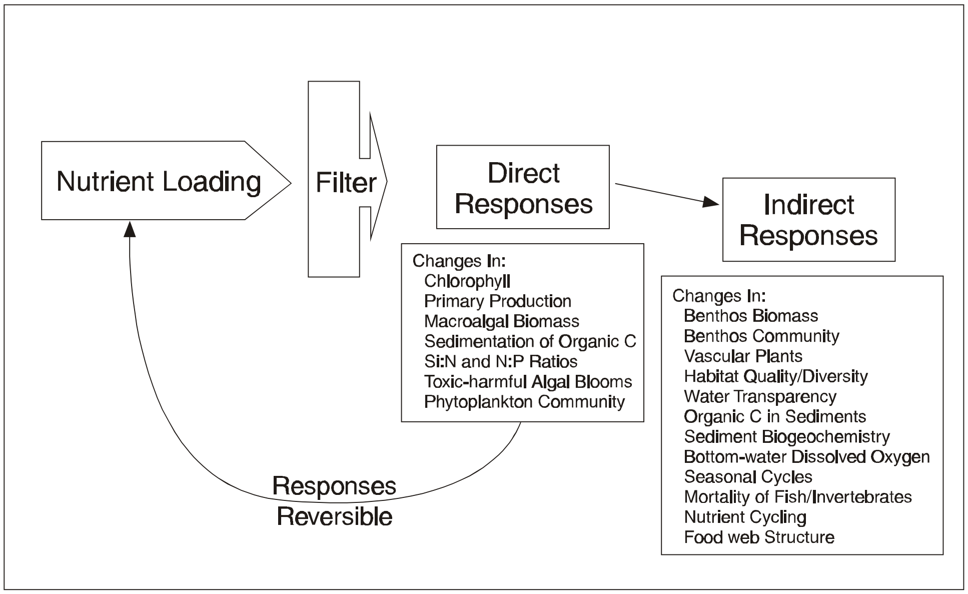

Figure 2-2. Schematic representation of the contemporary (Phase II) conceptual model of coastal eutrophication. Advances in recent decades include explicit recognition of (1) a complex suite of both direct and indirect responses to change in nutrient inputs; (2) system attributes that act as a filter to modulate these responses; and (3) the possibility of ecosystem rehabilitation through appropriate management actions to reduce nutrient inputs to sensitive coastal ecosystems. Source: Cloern 2001.

characterizing resources lost through nutrient overenrichment, it is often difficult to establish an accurate historical reference or determine the potential recovery from nutrient stress. Apparently, many estuaries became moderately to highly enriched before effective monitoring programs provided accurate descriptive information on biotic community distributions and abundance. When all else fails, professional judgment should be used to estimate reference conditions.

Finally, water quality managers should anticipate that nutrient enrichment will act with other stressors and forms of ecosystem disturbance and modify their respective ecological expressions (Breitburg et al. 1999).

These considerations suggest that water quality managers may face a large array of uncertainties regarding nutrient criteria development and implementation for estuaries and nearshore coastal waters. This manual attempts to guide application of established scientific principles and to reveal important uncertainties that bear on nutrient criteria development. This chapter begins with a contextual discussion of the watershed perspective characterized as the "river-to-ocean continuum."

River-to-Ocean Continuum: **Watershed/Nearshore Coastal Management Framework**

This section describes the physical relationship of estuaries and nearshore coastal waters to their respective water and sedimentary boundaries. This description provides a context for understanding problems of nutrient overenrichment in coastal ecosystems. Estuaries and nearshore coastal systems share some features, but important differences reflect how nutrients cause problems.

Some Important Identifying Features of Estuaries (adapted partly from Cloern 1996)

1. Estuaries are located between freshwater ecosystems (lakes, rivers, and streams; freshwater and coastal wetlands; and groundwater systems) and coastal shelf systems (Figure 2-2). These ecological boundary conditions create a transition between contrasting freshwater and open-ocean ecosystems.

2. Estuaries are relatively shallow; often, on average, only a few meters to a few tens of meters deep. This promotes a strong benthic-pelagic coupling that influences nutrient cycling through changes in system nutrient stoichiometry. A well-developed benthic community participates in nutrient cycling.

3. River-influenced estuaries are quite different from systems. Vertical mixing is regulated primarily by the seasonal cycle of heat input and thermal stratification that retards vertical mixing. However, in estuaries vertical mixing is regulated by a larger and more variable source of buoyancy: the riverine input of freshwater that acts to stabilize the water column. Also, freshwater input establishes longitudinal and vertical salinity gradients and drives nontidal gravitational circulation, a major contributor to flushing.

4. Estuaries are particle-rich relative to coastal systems and have physical mechanisms that tend to retain particles. These suspended particles mediate a number of activities (e.g., absorbing and scattering light, or absorbing hydroscopic materials such as phosphate and toxic contaminants). New particles enter with river flow and may be resuspended from the bottom by tidal currents and wind-wave activity.

5. Many estuaries are naturally nutrient-rich because of inputs from the land surface and geochemical and biological processes that act as “filters” to retain nutrients within estuaries (Kennedy 1984).

Variability in freshwater discharge is reflected in the estuarine salinity gradient, which has important consequences for stenohaline organisms, especially nonmotile forms. The salinity gradient of estuaries has been classified by on the Venice System, and salinity classes approximate the distribution of many estuarine organisms (Figure 2-3). Changes in salinity (e.g., wet and dry decadal periods) often modify population distributions and biotic community structure (Carriker 1967). Rivers and lakes process nutrients and modify nutrient ecological stoichiometry before the material arrives downstream, where receiving coastal waters further nutrient cycling (Billen et al. 1991). Nutrient cycling occurs along the continuum; phytoplankton and other algae are key agents of biochemical change (Redfield 1963) (Figure 2-4). Redfield et al. (1958) demonstrated that phytoplankton in active growth phase tend to maintain a C:N:P ratio close to 106:16:1. Annual rates of net primary production in coastal shelf environments tend to overlap rates of estuaries, but coastal shelves on average are somewhat lower in magnitude, except in upwelling areas where rates may, on average, exceed those of estuaries by a factor of two to three (Walsh 1988) (Table 2-1).

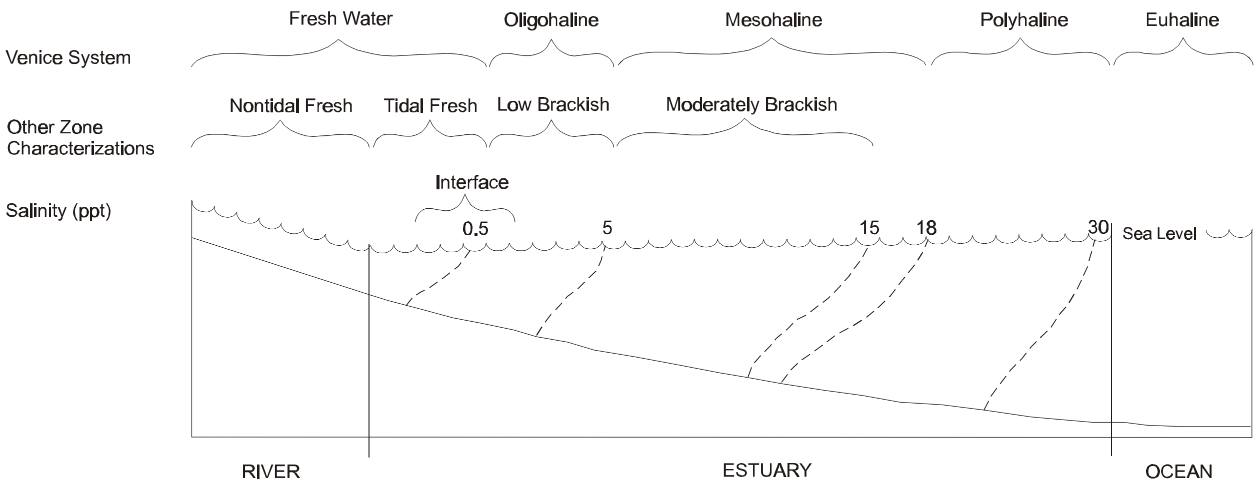

Figure 2-3. Salinity zones. The Venice System is a well-accepted method of characterizing salinity zones and covers the salinity ranges from riverine regions to the ocean. The freshwater category in the Venice System has been modified in this atlas to account for the tidal and nontidal regions found in rivers with estuarine portions. Source: Lippson et al. 1979, Environmental Atlas of the Potomac Estuary, MD Department of Natural Resources.

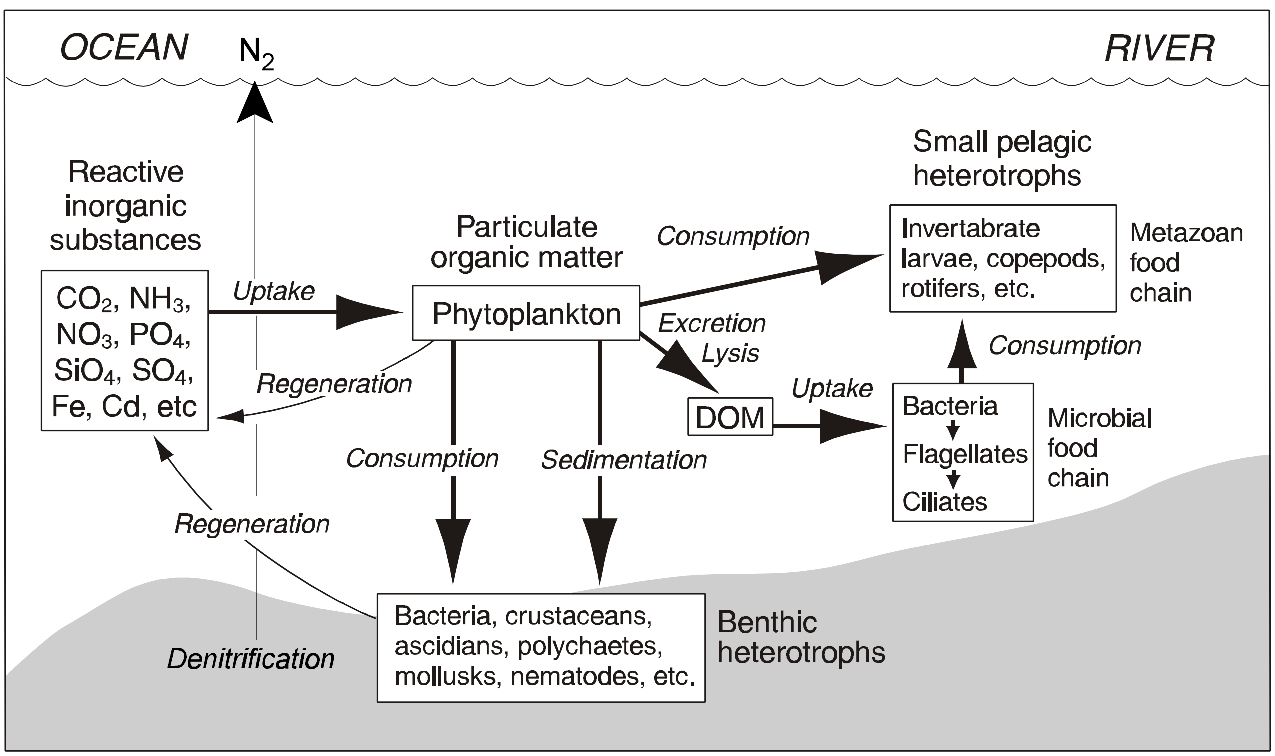

Figure 2-4. Schematic illustrating the central role of phytoplankton as agents of biogeochemical change in shallow coastal ecosystems. Phytoplankton assimilate reactive inorganic substances and incorporate these into particulate (POM) and dissolved organic matter (DOM) which support the production of pelagic and benthic heterotrophs. Arrows indicate some of the material fluxes between these different compartments. Denitrification has been added to the figure. Source: Cloern 1996.

Table 2-1. Categorization of the world's continental shelves based on location, major river, and primary productivity

Latitude (°)	Region	Major River	Primary Production (g Cm^{-2} yr^{1})
Eastern Boundary Current			
0-30	Ecuador-Chile	–	1000-2000
	Southwest Africa	–	1000-2000
	Northwest Africa	–	200-500
	Baja California	–	600
	Somali coast	Juba	175
	Arabian Sea	Indus	200
30-60	California-Washington	Columbia	150-200
	Portugal-Morocco	Tagus	60-290
Western Boundary Currents			
0-30	Brazil	Amazon	90
	Gulf of Guinea	Congo	130
	Oman/Persian Gulfs	Tigris	80
	Bay of Bengal	Ganges	110
	Andaman Sea	Irrawaddy	50
	Java/Banda Seas	Brantas	110
	Timor Sea	Fitzroy	100
	Coral Sea	Fly	20-175
	Arafura Sea	Mitchell	150
	Red Sea	Awash	35
	Mozambique Channel	Zambesi	100-150
	South China Sea	Mekong	215-317
	Caribbean Sea	Orinoco	66-139
	Central America	Magdalena	180
	West Florida shelf	Appalachicola	30
	South Atlantic Bight	Altamaha	130-350
Mesotrophic Systems			
30-60	Australian Bight	Murray	50-70
	New Zealand	Waikato	115

Table 2-1. Categorization of the world's continental shelves based on location, major river, and primary productivity (continued)

Latitude (°)	Region	Major River	Primary Production (g Cm^{-2} yr^{1})
	Argentina-Uraguay	Parana	70
	Southern Chile	Valdivia	90
	Southern Mediterranean	Nile	30-45
	Gulf of Alaska	Fraser	50
	Nova Scotia-Maine	St. Lawrence	130
	Labrador Sea	Churchill	24-100
	Okhotsk Sea	Amur	75
	Bering Sea	Kuskokwim	170
Phototrophic Systems			
60-90	Beaufort Sea	Mackenzie	10-20
	Chukchi Sea	Yukon	40-180
	East Siberian Sea	Kolyma	70
	Laptev Sea	Lena	70
	Kara Sea	Ob	70
	Barents Sea	Pechora	25-96
	Greenland-Norwegian Seas	Tjorsa	40-60
	Weddell-Ross Seas	–	12-86
Eutrophic Systems			
30-60	Mid-Atlantic Bight	Hudson	300-380
	Baltic Sea	Vistula	75-150
	East China Sea	Yangtze	170
	Sea of Japan	Ishikari	100-200
	North-Irish Sea	Rhine	100-250
	Adriatic Sea	Po	68-85
	Caspian Sea	Volga	100
	Black Sea	Danube	50-150
	Bay of Biscay	Loire	120
	Louisiana/Texas shelf	Mississippi	100

Source: Adapted from Walsh, with additional data from Alongi, and Postma and Zijlistra.

Some Identifying Features of Nearshore Coastal Waters

1. Nearshore coastal waters extend from the coastal baseline at high tide and across the mouths of estuaries to approximately three nautical miles. Coastal waters are relatively deep compared to estuaries with depths ranging from a few meters to several hundred meters, depending on coastal location.

2. Coastal longshore currents are a principal mechanism to exchange water masses.

3. Upwelling of nutrients from the deep ocean can be locally important.

4. Nearshore coastal systems tend to be particle-rich compared to the open ocean, but much less so than adjoining estuaries.

5. Nearshore coastal systems have a weaker benthic-pelagic coupling than estuaries mainly because they are deeper.

Coastal environments in the continental United States show only modest levels of upwelling compared to well-known upwelling areas, such as coastal Ecuador-Chile. The Gulf Stream, which flows northeastward along the South Atlantic coast from the Florida Straits to North Carolina, lies close enough to the shoreline to affect water temperature and circulation of nearshore waters. Dynamic core rings that slide off to the mainland side of the Gulf Stream affect local conditions. The coastal environment is dynamic in terms of phytoplankton bloom formation and dissipation (Walsh 1988). This has relevance to characterization of reference conditions and monitoring for nutrient criteria performance because the systems, though not as physically dynamic at short temporal scales as estuaries, are still difficult to assess in terms of average conditions. Synoptic survey tools such as aerial surveillance with fixed-wing aircraft and satellites can provide wide coverage, including short-term phytoplankton dynamics.

2.2 CONTROLLING THE RIGHT NUTRIENTS

Overview

Chapter 1 introduced the geographical extent and magnitude of the overenrichment problem and suggested the importance of nitrogen (N) versus phosphorus (P) as limiting nutrients. Several recent review papers (Downing 1997, Smith 1998, Smith et al. 1999, Conley 2000) and the NRC (2000) volume concluded that the major nutrients causing overenrichment problems (e.g., algal blooms) in estuaries and nearshore coastal waters are N and P. Silica (Si) may limit diatom production at relatively high levels of N and P. Iron is a co-limiting nutrient in some ocean areas and may exert some limitation in shelf waters,but its importance in open coastal waters usually is secondary to N (NRC 2000). Additionally, P limits primary production in some tropical nearshore habitats, although study of these systems is limited (Howarth et al. 1995). Often the addition of both N and P will elicit greater phytoplankton biomass stimulation than the sum of both nutrients added separately (Fisher et al. 1992). There are reported cases where both N and P are required to elicit a phytoplankton biomass production response in estuaries (Flemer et al. 1998), suggesting that N and P supply rates were equally limiting. Tropical lagoons, with carbonate sands low in P and unaffected by human activity, also are prone to P limitation. For example, the seagrass *Thalassia testudinum* was P-limited in Florida Bay **(**Powell et al. 1989, Fourqurean et al. 1992a,b).

Tidal fresh and brackish waters in many estuaries typically are more light limited than higher saline waters (Flemer 1970, Sin et al. 1999). As freshwater fluxes seaward, processes operate to modify nutrient stoichiometry (e.g., sedimentation of P-absorbed particles, denitrification, and differential microbial decomposition). A number of temperate estuaries exhibit seasonal shifts in nutrient limitation with winter-spring P limitation and summer-fall N limitation (D'Elia et al. 1986; Fisher et al. 1992, Malone et al. 1996) (Table 2-2). The Redfield ratio (C:N:P) of marine benthic plants approximates 550:30:1, substantially richer in organic carbon, much of which is structural material, and indicates that these plants require less N and P than do phytoplankton (Atkinson and Smith 1982). In summary, the foregoing results suggest that both N and P criteria are needed, depending on season and local ecosystem conditions (Conley 2000).

Some Empirical Evidence for N Limitation of Net Primary Production

Three case studies provide some of the strongest evidence available that water quality mangers should focus on N for criteria development and environmental control (see NRC 2000 for details). One study involves work in large mesocosms by the University of Rhode Island (Marine Ecosystem Research Laboratory–MERL) on the shore of Narragansett Bay. Experiments showed that P addition was not stimulatory, but N or N+P caused large increases in the rate of net primary production and phytoplankton standing crops (Oviatt et al. 1995).

In another study, nutrient releases from a sewage treatment plant were monitored in the Himmerfjarden Estuary south of Stockholm, Sweden, on the Baltic Sea (Elmgren and Larsson 1997). Throughout a 17-year field experiment (i.e., whole-ecosystem study), the concentration of total N tended to reflect the N input from the sewage treatment plant, and both abundances of phytoplankton and water clarity were clearly related to the total N concentration and not to total P. This experiment involved independent increases and decreases in N and P over the observation period.

A third whole-ecosystem study involved long-term changes in Laholm Bay, Sweden (Rosenberg et al. 1990). Early signs of overenrichment appeared in the 1950s and 1960s and steadily increased over time (Figure 2-5). Among the earliest reported signs were changes in the composition of macroalgal species. Over time the filamentous algae typical of enriched conditions became more prevalent, and harmful algal blooms (HABs) became more common during the 1980s. These changes correlated best with changes over the decades in N loads rather than P loads. These field studies are excellent examples of the power of long-term monitoring of nutrient and biological variables in estuaries (Wolfe et al. 1987). Importantly, these three ecosystem experiments correlated well with short-term bioassay experiments and ratios of dissolved inorganic N:P ratios in these ecosystems (NRC 2000). The above whole-system field experiments and the large preponderance of bioassay data in estuaries and nearshore coastal systems (Howarth 1988) and generally low inorganic N:P atomic ratios at peak primary production (Boynton et al. 1982) make a strong case for the widespread importance of N as a controlling nutrient for net coastal

Table 2-2. Estuaries exhibiting seasonal shifts in nutrient limitation with spring P limitation and summer N limitation

Estuary	Reference
Baltic Sea	
Himmerfjarden Estuary, Sweden	Graneli et al. 1990, Elmgren & Larsson 1997
Gulf of Riga, Latvia	Maestrini et al. 1997
Roskilde Fjord, Denmark	Pedersen & Borum 1996
Bay of Brest, France[a]	Del Amo et al. 1997
Chesapeake Bay, USA[a]	
Mainstem	Malone et al. 1996
Patuxent River Estuary	D'Elia et al. 1986
York River Estuary	Webb 1988
Rhode River Estuary	Gallegos & Jordan 1997
Delaware Estuary, USA	Pennock & Sharp 1994
Neuse River Estuary, USA	Mallin & Paerl 1994

[a] Systems displaying seasonal dissolved silicate limitation.
Source: Conley 2000.

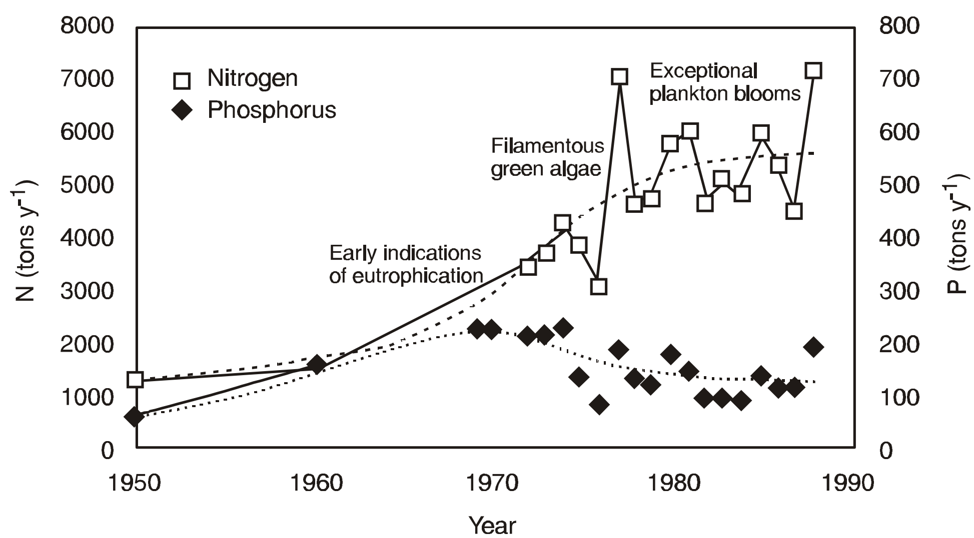

Figure 2-5. Transport of nutrients to Laholm Bay, Sweden. Periods of significant changes in the marine biota are also indicated (modified from Rosenberg et al. 1990). Source: NRC 2000.

marine primary production and a major contributor to water quality problems. Interpretation of nutrient ratios was initially applied in the open ocean by Redfield (1934) and further elaborated on by Redfield (1958) and Redfield et al. (1963). Boynton et al. suggested that when inorganic N:P ratios for a variety of estuarine systems are interpreted, atomic ratios less than 10 indicated N limitation and ratios greater than 20 indicated P limitation (Figure 2-6). Some have suggested that it matters whether the inorganic N is in the form of ammonium- or nitrate-N. High concentrations of ammonia-N may inhibit nitrate-N uptake; however, Dortch (1990) reported that this phenomenon is more variable than widely believed. Figure 2-7 summarizes major factors that determine whether N or P is more limiting in aquatic ecosystems where one of these macronutrients is limiting net primary production.

Some Threshold Responses to Nitrogen Overenrichment

Kelly (in press) summarized several generalizations that appear to hold for N overenrichment in estuaries. Over a range of average dissolved inorganic nitrogen (DIN) from <1 to >20 µM, chlorophyll *a* tends to increase at slightly less than 1 µg/L with every 1 µM increase in DIN or approximately about 0.75 µg chl/µM DIN (e.g., see Figure 3-2b in Chapter 3). Evidence is especially strong that N concentrations can reduce or eliminate growth of estuarine submerged aquatic vegetation (SAV) and higher salinity seagrasses (Sand-Jensen and Borum 1991; Dennison et al. 1993; Duarte 1995) by both water column shading and epiphytic overgrowth. Estuarine SAV and seagrasses tend to show light limitation when surface insolation approximates 11% at the surface of the canopy, but this figure varies between about 5% and 20% depending on species. Stevenson et al. (1993) transplanted plugs of *Ruppia maritima, Potamogeton perfoliatus,* and *P. pectinatus* in different areas of the Choptank Estuary, Chesapeake Bay, and reported that survival thresholds occurred when total suspended solids were between ~15 and 20 mg/L, chlorophyll *a* was 15 µg/L, DIN was below 10 µM, and PO_4 was below 0.35 µM. Kelly (in press) reviewed a number of studies and suggested that an approximate threshold for hypoxia occurred at about 80 µM TN (Table 2-3) (normalized TN loading for residence time expressed in years and divided by depth). These relationships document the importance of N as a major cause of estuarine water quality impairment. Also, these ecological response thresholds are a useful rule of thumb, but some deviations are to be expected. In data-poor estuaries, such thresholds are a first-order target until more adequate data can be developed to establish reference conditions.

Although overenrichment from N causes many symptoms of marine water quality impairment, it is the interaction of biogeochemical, biological, and physical processes that modulate the effects of a particular N supply (Cloern 2001) (Appendix A). These relationships had their genesis in the late 19th and early 20th Centuries in northern Europe, especially in German and Scandinavian marine research institutes (Mills 1989). Water quality managers who understand this interplay will assess cause-and-effect relationships with a deeper insight. Knowledge of algal nutrient physiology is necessary information, but it alone is insufficient to explain why blooms occur.

Effects of Physical Forcing on Net Primary Production

Each physical forcing (e.g., river inflows, wind velocity, irradiance, water temperature, and tidal currents) contributes to phytoplankton population variability by influencing rates of vertical mixing,

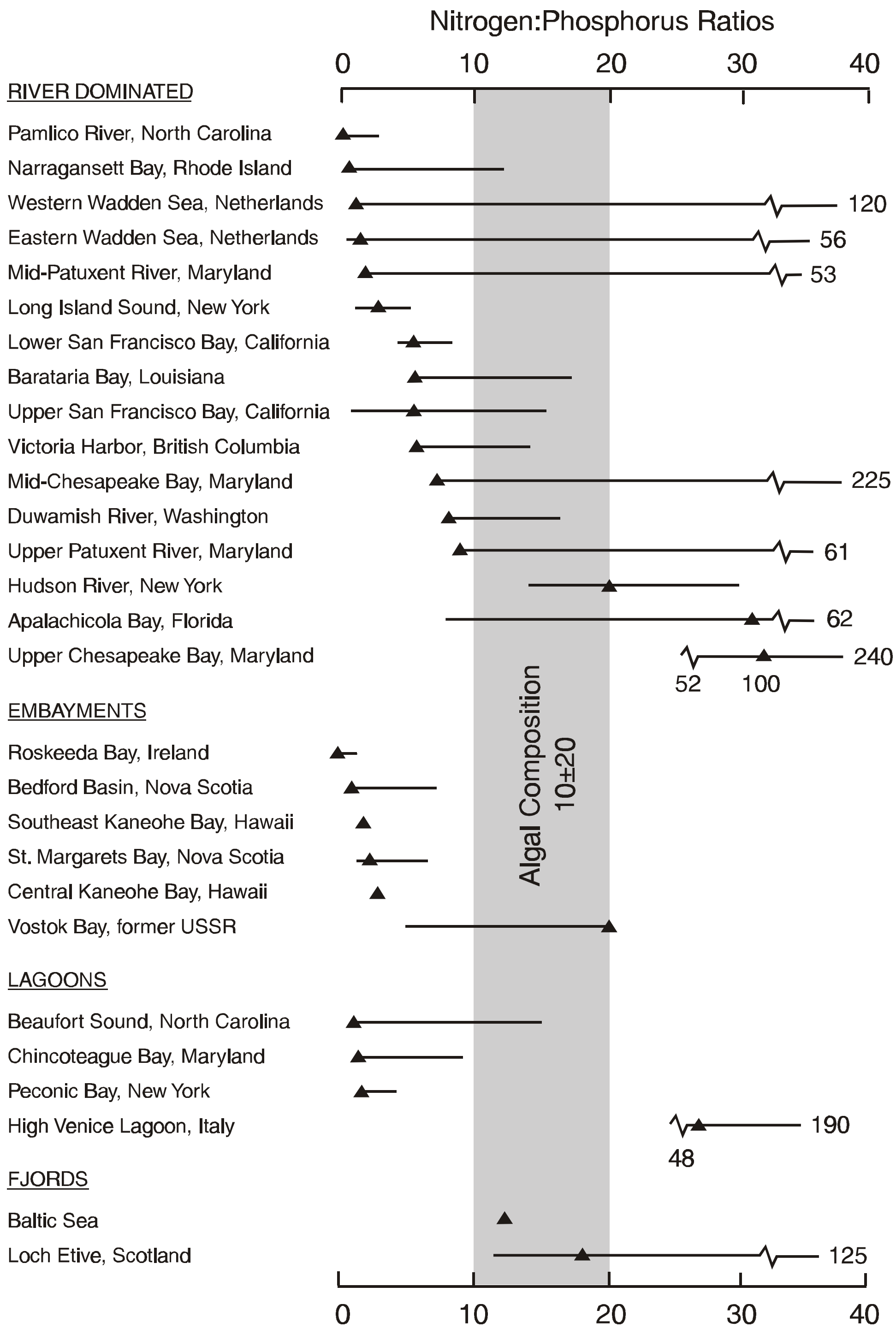

Figure 2-6. Summary of nitrogen:phosphorus ratios in 28 sample estuarine ecosystems. Horizontal bars indicate the annual ranges in nitrogen:phosphorus ratios; solid triangles represent the ratio at the time of maximum productivity. Vertical bands represent the typical range of algal composition ratios (modified from Boynton et al. 1982). Source: NRC 2000.

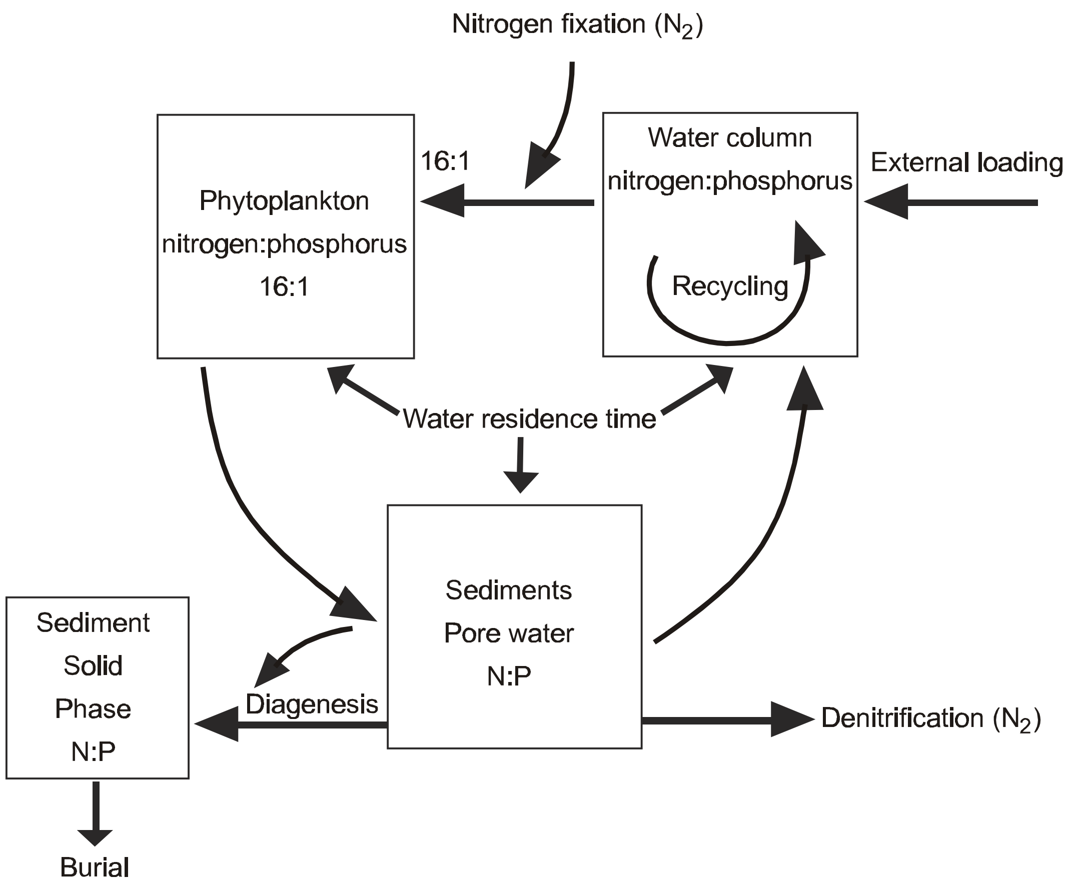

Figure 2-7. Factors that determine whether nitrogen or phosphorus is more limiting in aquatic ecosystems, where one of these macronutrients is limiting to net primary production. Phytoplankton use nitrogen and phosphorus in the approximate molar ratio of 16:1. The ratio of available nitrogen in the water column is affected by: (1) ratio of nitrogen:phosphorus in external inputs to the ecosystem; (2) relative rates of recycling of nitrogen and phosphorus in the water column, with organic phosphorus usually cycling faster than organic nitrogen; (3) differential sedimentation of nitrogen in more oligotrophic systems; (4) preferential return of nitrogen or phosphorus from sediments to the water column due to processes such as denitrification and phosphorus adsorption and precipitation; and (5) nitrogen fixation (modified from Howarth 1988; Howarth et al. 1995). Source: NRC 2000.

Table 2-3. DO, nutrient loading, and other characteristics for selected coastal areas and a MERL mesocosm enrichment experiment (source: Kelly in press)

System	Area	Depth Avg (m)	Annual TN loading (mmol m^{-2})	Res. Time (mo)	DO Status[a]	Vertical Mixing Status	Normalized TN Loading (μM)[b]	Primary Production (g Cm^{-2} y^{-1})
Experimental[c]	**(m^2)**							
MERL-control	2.63	5	800	0.9	OK	mixed	12	190 (100)
MERL-1X	2.63	5	1,750	0.9	OK	mixed	26	270 (115)
MERL-2X	2.63	5	2,950	0.9	OK	mixed	44	305 (243)
MERL-4X	2.63	5	4,850	0.9	OK	mixed	72	515 (305)
MERL-8X	2.63	5	9,000	0.9	~H	mixed	133	420 (171)
MERL-16X	2.63	5	18,500	0.9	H	mixed	274	900 (601)
MERL-32X	2.63	5	34,000	0.9	A	mixed	503	1150 (901)
Field[d]	(km^2)							
Baltic Sea [e]	374,600	55	217	250	H/A	stratified	81	~149-170
Scheldt	277	11.2	13,400	3	H/A	??	295	?
Chesapeake Bay [f,g]	11,542	6	938	7.6	A	stratified	98	~380 to 520 (361-858)
Potomac River [f]	1,210	5.9	2,095	5	H/A	stratified	146	~290 to 325
Guadalupe estuary [h]	551	1.4	548	10	?	??	322	?
	551	1.4	2,058	1	?	??	121	?
Ochlocknee Bay	24	1	5,995	0.1	OK		49	?
Delaware Bay	1,989	9.7	1,900	4	OK	stratified	64	~200 to 400
Narragansett Bay [i]	328	8.3	1,960	0.9	OK	weak strat	17	270 to 290
Providence River [j]	24.13	3.7	13,600	0.083	H	stratified	25	?
Providence Riv.[j,k]	24.13	3.7	13,600	0.233	H	stratified	70	?
Boston Harbor [l]	103	5.5	21,600	0.266	~H	weak strat	86	?
N. Outer Harbor [m]	13	10	107,692	0.03	OK	mixed	27	263 to 546
N. Gulf of Mexico [n]	20,000	30	6500	6 [O]	H/A	stratified	107	~290 to 320

[a]H= hypoxia, A= anoxia.
[b]Volumetric TN loading is normalized for residence time to yield an "expected" or potential concentration. The value is calculated as: Annual TN Loading * Residence time (expressed in years) divided by Depth. Units are thus mmol/m^3, or μM. See Kelly 1997a,b; 1998. The value is not decremented for denitrification or burial, removal processes that have greater effect on concentrations in longer residence time systems (cf. Nixon et al. 1996, Kelly 1998).
[c]See Nixon et al. 1984, Oviatt et al. 1986, Nixon 1992, Nixon et al. 1996. DIN was used to enrich treatment conditions (e.g. 1X...32X) and is represented in Figures 5, 6, and 7. TN values include input of organic forms with feedwater, which is only a substantial portion of input at the control and the low end of the enrichment gradient. Production for year 1 of experiment was extrapolated using empirical model of Keller 1988, which did not include measurements of primary production above 600 g C $m^{-2}y^{-1}$ (Nixon 1992). These values are used in Figures 6 and 7. Parenthetical production values for year 2 are from Keller 1988. Hypoxic and anoxic events were periodic, not chronic.
[d]Except for Providence River, Boston Harbor and Gulf of Mexico, loading is TN as reported by Nixon et al. 1996. With noted exceptions for individual systems below, see Nixon (1992, 1997) for productivity references.
[e]Also see Elmgren 1989, Cederwall and Elmgren 1990, Rosenberg et al. 1990. Table value for TN loading from Nixon et al. 1996 is lower than DIN input in Nixon 1997 plot, which included N input across the halocline. Lower value is labeled in Figure 6.
[f]Also see Boynton et al. 1995, Boynton and Kemp 2000; historical Chesapeake production range (parenthetical) is from Boynton et al. 1982.

Table 2-3. DO, nutrient loading, and other characteristics for selected coastal areas and a MERL mesocosm enrichment experiment (continued)

[g]Mainstem stratification, increasing anoxic extent; Officer et al. 1984, Boynton and Kemp 2000.
[h]Top line is for dry flow, bottom line is for wet flow.
[i]Only strongly stratified by freshwater at head of Bay in Providence River area, see notes j, k below. Production range is from Nixon 1997 (does not include historical presettlement estimate of 120-130 g C $m^{-2}y^{-1}$).
[j]Oviatt et al. 1984, Doering et al. 1990, Asselin and Spaulding 1993; TN loading from seaward and landward inputs, avg residence time (2.5 d), low DO in 13-15 m channel.
[k]Uses longer 7-d residence time during very low flow conditions, Asselin and Spauling 1993.
[l]TN budget inclues direct estimate of ocean loading as well as land loading. Nixon et al. 1996 gave a preliminary budget; table shows improved budget of Kelly 1998. Freshwater stratification and near hypoxia/occasional hypoxia only occur in inner harbor. See Signell and Butman 1992 for flushing estimate of whole harbor.
[m]Northern harbor section, Kelly 1998. Harbor station production of Kelly and Doering 1997.
[n]Area represents greatest measured extent of hypoxic zone. Higher production is for immediate plume (Rabalais et al. 2000). TN loading is to a 20,000-km^2 hypoxic zone only (and thus is a maximal rate) based on Mississippi/Atchafalaya input of 130 x 10^9 moles y^{-1} (Howarth et al. 1996; Turner and Rabalais 1991). Rate is consistent with long-term average (1980-1996) estimated by CENR 2000 of 1,567,900 metric tons y^{-1}.
[o]Assumed a 6-mo residence time (~seasonal turnover) *for illustration only*; if longer, then normalized concentration would increase accordingly.

sedimentation, horizontal transport, production, and grazing. Each forcing has its characteristic timescale of variability (e.g., 12.4-hr tidal period, the diel 24-hr light cycle, several days to weeks-long storm events of enhanced river flow and wind stress, and seasonal cycles of irradiance and temperature; Cloern 1996).

Phytoplankton growth depends on nutrient supplies, as expected, but growth is significantly modulated by complex physical processes that operate at virtually every physical scale (Giller et al. 1994). For this reason, it is desirable for RTAGs and State water quality managers to have ready access to individuals with a specialty in physical oceanography.

In estuaries, bottom topography and bathymetry form the basin in which tidal currents, freshwater inflow, and wind vectors act as principal drivers of estuarine and coastal physical processes and contribute to variability in mixing and circulation of waters (Cloern 1996) (Figure 2-8). Physical processes can attenuate or exacerbate nutrient enrichment effects depending on the form of interaction. For example, the Delaware River Estuary receives TN and TP loads somewhat larger than does the mainstem Chesapeake Bay, yet the Delaware Estuary has lower phytoplankton production and does not have a hypoxia problem, largely because of its relatively strong vertical mixing (i.e., a weak vertical density stratification) and horizontal water exchange with the open ocean system (Pennock 1985).

Freshwater inflow is the “master driver” that defines the ecological character of river-dominated estuaries. Boynton and Kemp (2000) proposed a simple conceptual model to explain effects of river flow on Chesapeake Bay ecological processes associated with nutrient inputs (Figure 2-9). These authors stated:

> *The importance of freshwater inputs is obvious; it is a central feature in the definition of estuarine systems, it influences physical dynamics (Boicourt 1992), is well correlated with nutrient inputs (Summers 1993), and has been implicated in regulating either directly or indirectly estuarine processes ranging from primary production (Boynton et al. 1982; Cloern et al. 1983) to benthic secondary production (Flint 1985) to fish recruitment (Stevens 1977) and catch (Sutcliffe 1973; Sutcliffe et al. 1977; Ennis 1986).*

Boynton and Kemp applied regression techniques to datasets from mid-Chesapeake Bay, a mesohaline area, to test the ideas represented in Figure 2-9. They showed that Susquehanna River flow was significantly related to annual average primary production, annual average surface chlorophyll *a*, spring deposition of total chlorophyll *a* per square meter, and total chlorophyll *a* deposition rate (meter squared per day). They also showed that the decline in dissolved oxygen concentrations in deep water during the spring bloom period was also related to flow (Figure 2-10). Although this relationship could be driven by riverflow effects on stratification, which in turn regulates dissolved oxygen depletion, they argue that river inputs of nutrients are of primary concern. This is because years of high and low stratification did not correlate well to years of high and low rates of oxygen decline. The implication is that nutrient enrichment played a key role in deep-water hypoxia.

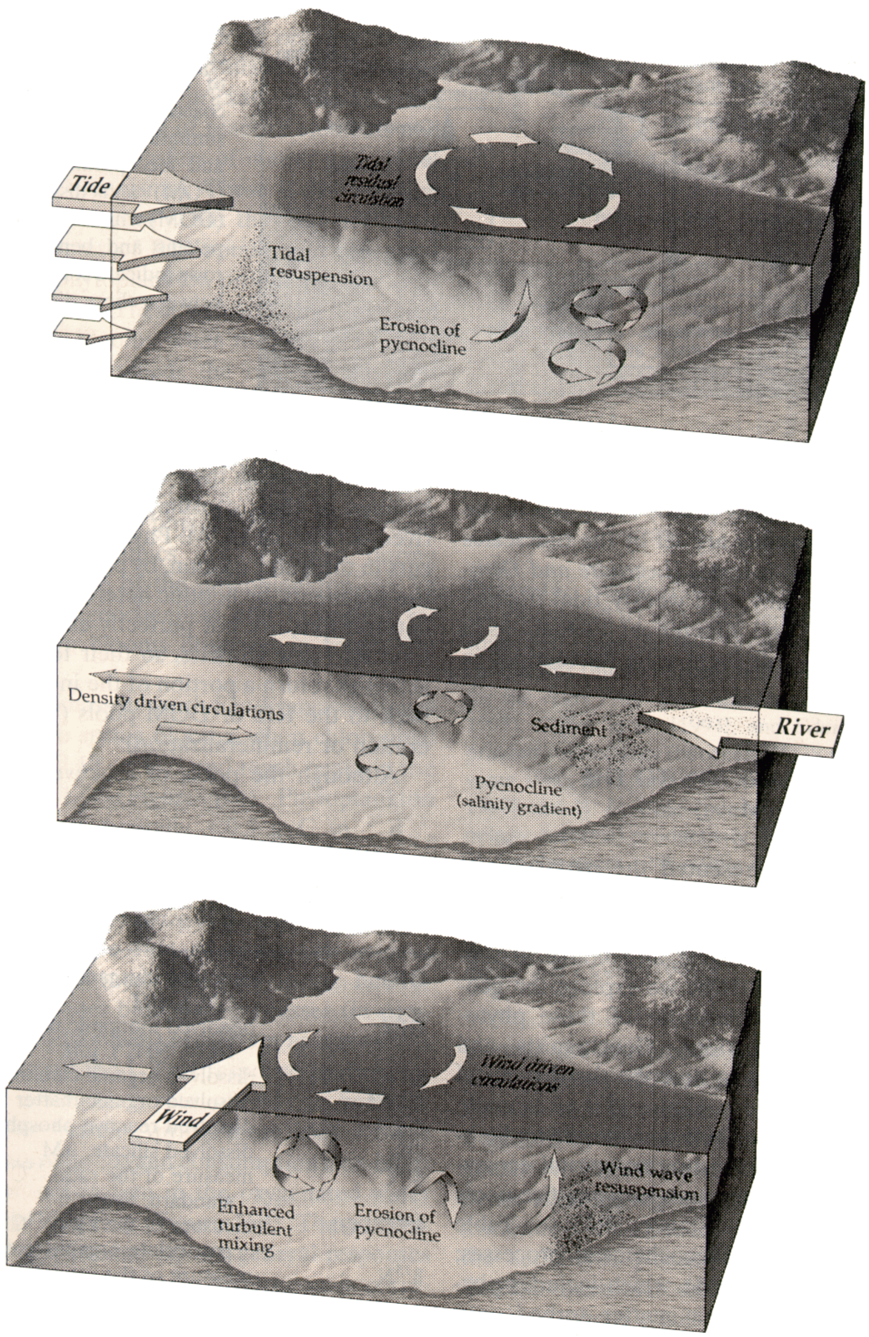

Figure 2-8. Cartoon diagrams of three physical forcings that operate at the interface between SCEs and the coastal ocean (tides), watershed (river inflow), and atmosphere (wind). Each physical forcing influences the growth rate of the resident phytoplankton population through, for example, its influence on the distribution of suspended sediments and turbidity. Each forcing also influences the rate of vertical mixing, with riverine inputs of freshwater as a source of buoyancy to stratify the water column and the tide and wind as sources of kinetic energy to mix the water column. Each forcing is also a mechanism of water circulation that transports phytoplankton horizontally. Much of the variability of phytoplankton biomass during blooms can be understood as responses to fluctuations in these interfacial forcings. Source: Cloern 1996.

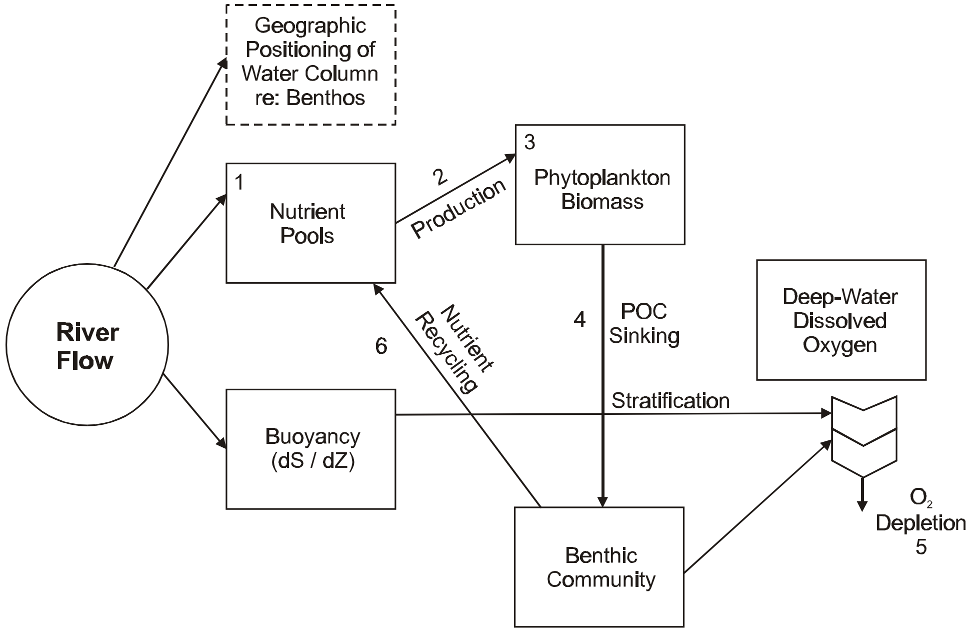

Figure 2-9. Simple schematic diagram showing the influences of river flow on ecosystem stocks and processes examined in this study. The mechanistic relationships between river flow and the stocks and processes shown in the diagram are explained in the text. Source: Boynton and Kemp 2000.

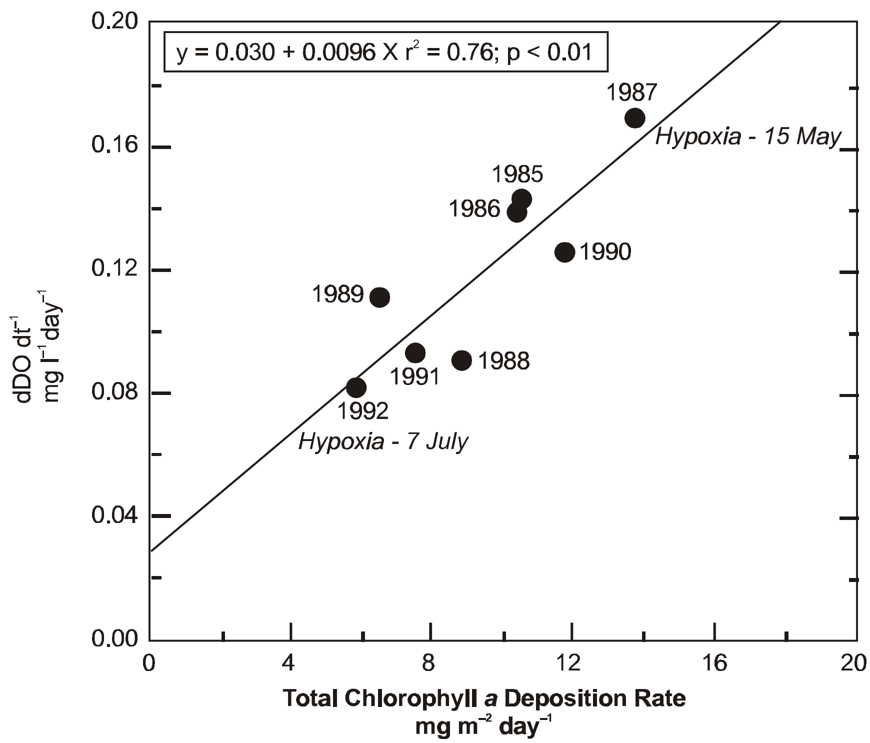

Figure 2-10. Scatter diagram showing the relationship between the rate of decline in dissolved-oxygen concentrations in deep water (dDO dt^{-1}) and average deposition rates of total chlorophyll *a* during the spring-bloom period. Data are from the 1985-1992 period and were collected at the R-64 site. The date on which hypoxia (DO concentration <1 mg 1^{-1}) was first encountered during highest (1987) and lowest (1992) deposition years is also indicated.

Freshwater inflow plays a major role in the degree of stratification (Figure 2-11a-d) and nontidal flushing (Figure 2-12) of estuaries. Density stratification influences the depth of vertical mixing relative to the euphotic zone depth and the tendency toward hypoxia formation, that is, the effect of sealing off bottom waters from reaeration. On a seasonal basis, stratification greatly influences the degree of hypoxia, but seems to have a lesser role on an interannual scale (see above paragraph). Tidal displacement also contributes to flushing (Figure 2-13). Numerous studies have documented the role of freshwater inflow regulation of primary production through interaction with other estuarine processes via different mechanisms (Pennock and Sharp 1994, Harding and Perry 1997, Cloern 1996, Sin et al. 1999). Freshets deliver substantial quantities of nutrients to an estuary and lead to blooms (Mallin et al. 1993, Rudek et al. 1991). Effects of rainfall operating on hydrographic processes have been shown to influence trophic organization (Livingston 1997). A significant effect of episodic freshwater inflow is determining the appropriate averaging period for reference conditions applicable to nutrient criteria development. The issue applies to decadal wet and dry cycles as well. Water quality managers should anticipate that even in estuaries relatively free of anthropogenic nutrient enrichment, some level of hypoxia may occur during wet weather cycles. This "natural" condition, should it be observed will need to be factored into nutrient criteria development.

Other Physical Factors

Other physical factors (e.g., salinity, temperature, and light) influence the expression of nutrient enrichment effects and are extensively reported in standard textbooks. For example, salinity can influence enrichment effects and can also influence biotic distributions (e.g., grazing populations), primarily through the osmotic capabilities of resident organisms (Kinne 1964). Temperature and light availability to photosynthetic organisms is obviously important. Temperature regulates, within certain limits, the metabolic rates of organisms, especially poilkilotherms, and influences the distribution of many species. Light also influences the feeding behavior of many planktonic animal forms, especially crustacean filter feeders, which has relevance to algal grazing. Climatic factors influence phytoplankton biomass production in estuaries (Lehman 2000). Additional information on the roles of temperature and light as limiting factors to net primary production and effects of nutrient overenrichment is provided in Appendix B.

2.3 NUTRIENT LOADS AND CONCENTRATIONS: INTERPRETATION OF EFFECTS

> *The issue of whether or not to focus on nutrient concentration versus loading criteria has been a contentious one among both scientists and managers. Whether or not to use concentrations or loading as criteria largely depends on the spatial and temporal scales of assessing ecosystem responses to nutrient inputs* (H. Paerl, personal communication).

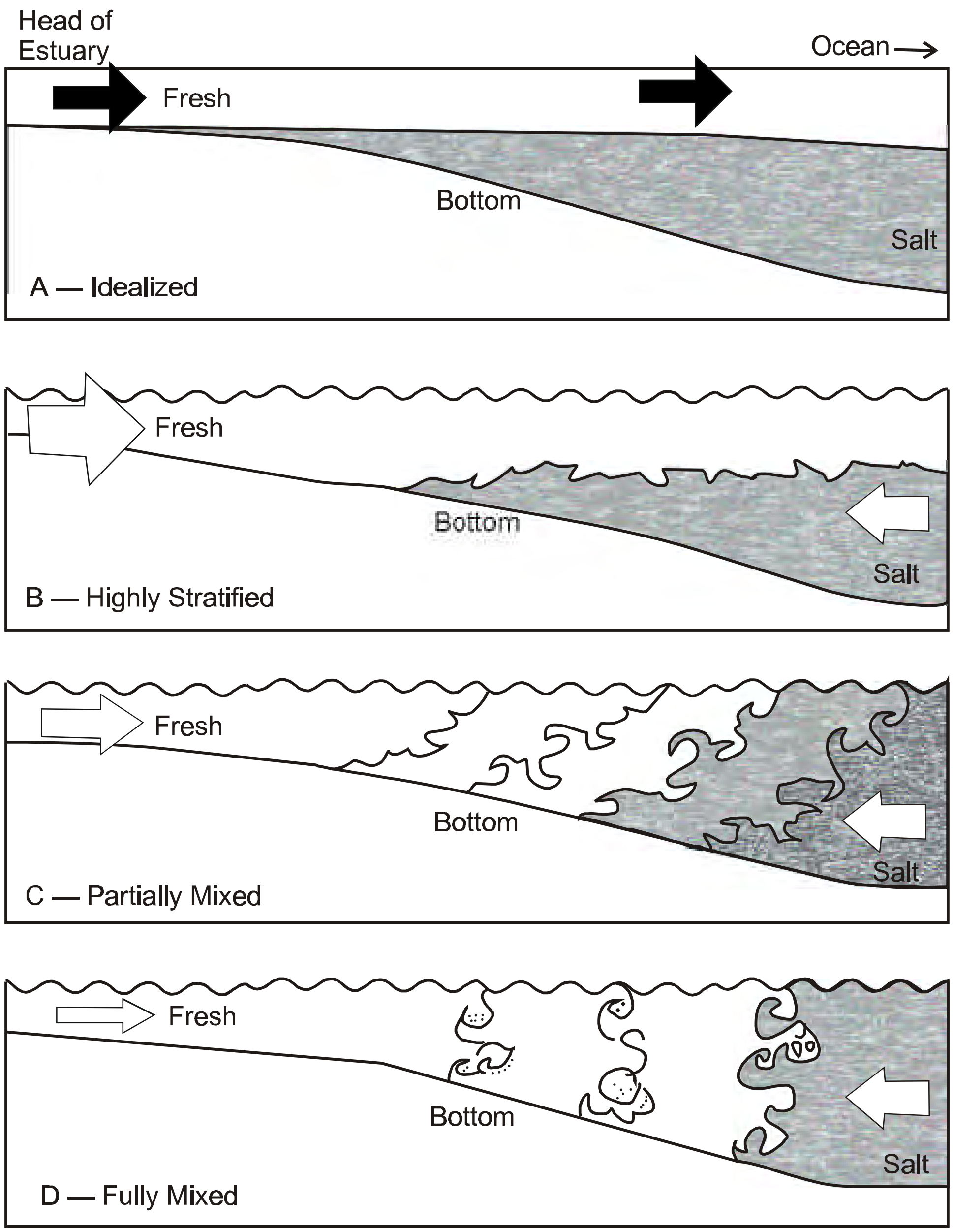

Figure 2-11a–d. Schematic diagram of coastal plain estuary types, indicating direction and degree of mixing. Arrows show direction of net mass transports of water, and the arrow size indicates the relative magnitudes of the transports. Source: Lippson et al. 1979.

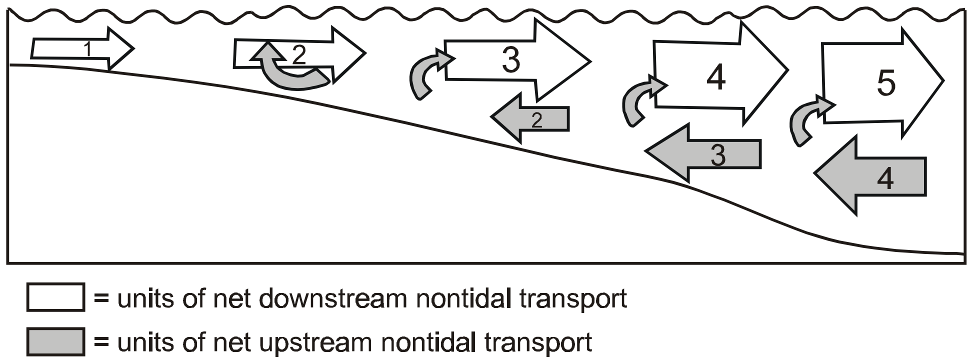

Figure 2-12. Net transports in estuaries resulting from estuarine flows and mixing. At any one point along an estuary, the difference between upstream- and downstream-directed transports is equal to the freshwater input to that point. In this example with no tributaries, the difference is equal to the input at the head of the estuary. Source: Lippson et al. 1979.

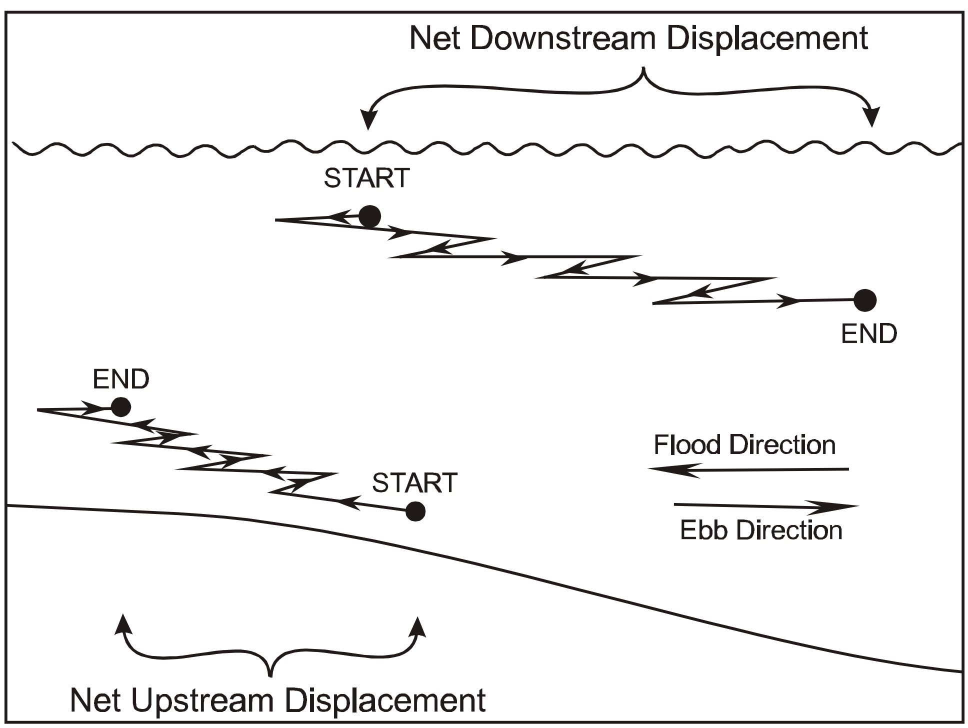

Figure 2-13. Net movement of a particle in each layer of a two-layered flow system. Source: Lippson et al. 1979.

Conceptual Framework

Nutrient concentrations are what phytoplankton (and other plants) respond to instantaneously or on very short time scales. The dissolved inorganic and, to some extent, organic nutrient concentrations that remain in a water parcel after a short period of phytoplankton growth are largely what is left over or unused. (Note: Some dinoflagellates can obtain nutrients from particulate materials and exhibit other complex forms of nutrition.) Nutrient uptake, including any luxuriant uptake, will be mostly converted into organic form, given a suitable short period for growth. Thus, total concentration is a measure of the nutrient in living form as well as any unused organic and inorganic forms. If concentrations of nutrients are to be used as criteria, the total concentration is most likely to reflect the short-term phytoplankton growth potential (Boynton and Kemp 2000).

Recycling is an important aspect of phytoplankton biomass production. If nutrients in a water parcel are all converted into algal biomass, then maintaining the algal biomass requires rapid recycling or additional supplies to the water parcel. With loss of phytoplankton from the water column through sedimentation, grazing and conversion of phytoplankton to animal biomass, dispersion, and advection, maintenance and any further net primary production require new supplies of nutrients. These processes all involve longer time scales that include seasonal and interannual considerations of ecosystem water quality (i.e., use impairments) and habitat response.

Examples

Some examples of regression relationships between nutrient load and concentration and response variables are instructive because nutrient concentration often does not provide a useful relationship. There is a range in the lag time between nutrient load and coastal water ecosystem responses. Such lags have been reported for a number of estuaries, including the Patuxent (Kemp and Boynton 1984), mainstem of the Chesapeake Bay (Malone et al. 1988), mesohaline York River estuary (Sin et al. 1999), and Logan River and Moreton Bay, Australia (O'Donohue and Dennison 1997). Nixon et al. (1996) developed a number of regressions between residence time and response variables (e.g., percent total N, percent P exported, percent N retained from land and atmosphere, and percent N denitrified) from a number of estuaries and coastal marine systems. Dettmann (in press) developed relationships somewhat similar to those of Nixon et al. that included some different estuaries and coastal waters employing a modified algebraic expression for residence time (e.g., Figure 2-14). The temporal scale of these regressions typically ranges from months to annual averages. These regressions help frame causal relationships but usually are not adequate by themselves to establish nutrient criteria. For example, the Delaware Bay lies between the northern Adriatic Sea and Chesapeake Bay in terms of the fraction of N exported, but the Delaware Bay has few symptoms of nutrient overenrichment.

For a number of coastal embayments in Virginia and Maryland, chlorophyll *a* concentration regressed on a TN loading rate that was scaled to a unit area loading rate of the receiving waterbody surface area, resulting in a relatively high R^2 (Boynton et al. 1996). Peak chlorophyll *a* concentrations in the Potomac Estuary regressed against peak TN load showed the highest chlorophyll *a* concentrations occurred under average flow conditions (Boynton 1997). Maximum freshwater inflows resulted in a very strong density stratification, but the nutrients were advected into the lower Chesapeake Bay, and thus no bloom formed

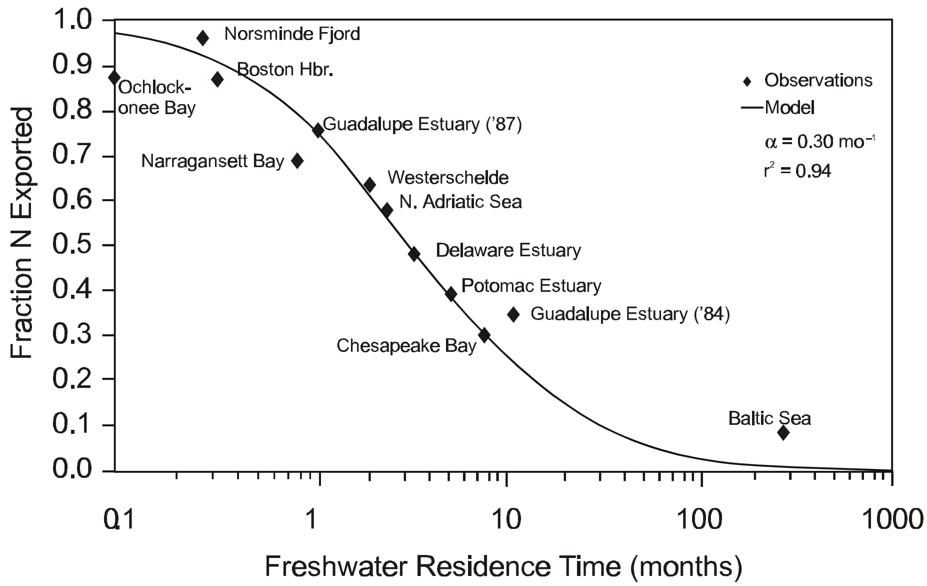

Figure 2-14. The fraction of landside nitrogen input exported from 11 North American and European estuaries versus freshwater residence time (linear time scale). Baltic Sea not shown. Source: Dettmann (in press).

in the lower Potomac estuary. Low freshwater inflows resulted in much weaker vertical density stratification and apparently a low nutrient supply that limited phytoplankton bloom potential (Figure 2-15).

Using an interannual time scale, Harding (1994) summarized the historical (1950–1994) nutrient and chlorophyll *a* trends for the mainstem of the Chesapeake Bay. Nitrogen, P, and chlorophyll *a* concentrations increased considerably over the period of record. Harding and Perry (1997) applied a statistical time series model and determined that confounding effects of freshwater inflow did not explain the chlorophyll *a* increase in the lower bay. The DIN:DIP ratios suggested a greater influence of DIN as a limiting nutrient to biomass production. Variation in the flow of the Susquehanna River over the period of record tends to cloud the empirical relationships, especially in the oligohaline region and brackish zone.

By inference, nutrients were hypothesized to be the principal causative agent. Since the 1970s, the winter-spring freshet has been associated with a strong diatom bloom, and in 1989 a drought delayed delivery of DIN and Si to the mesohaline reach of the bay until late spring, thus leading to a late-season phytoplankton biomass increase composed primarily of flagellates.

Phytoplankton growth and biomass accumulation appear to be directly related to riverborne nutrient inputs in the Chesapeake Bay (Boynton et al. 1982, Malone et al. 1988). Typically, years with higher river flow (within limits) are marked by greater algal biomass, which supports elevated respiration and more rapid depletion of bottom water DO in deep, stratified estuaries (Boicourt 1992). However, this relationship is confounded by interannual variations in salinity stratification because stratification is directly related to river flow (Seliger and Boggs 1988, Officer et al. 1984). Distinguishing between the

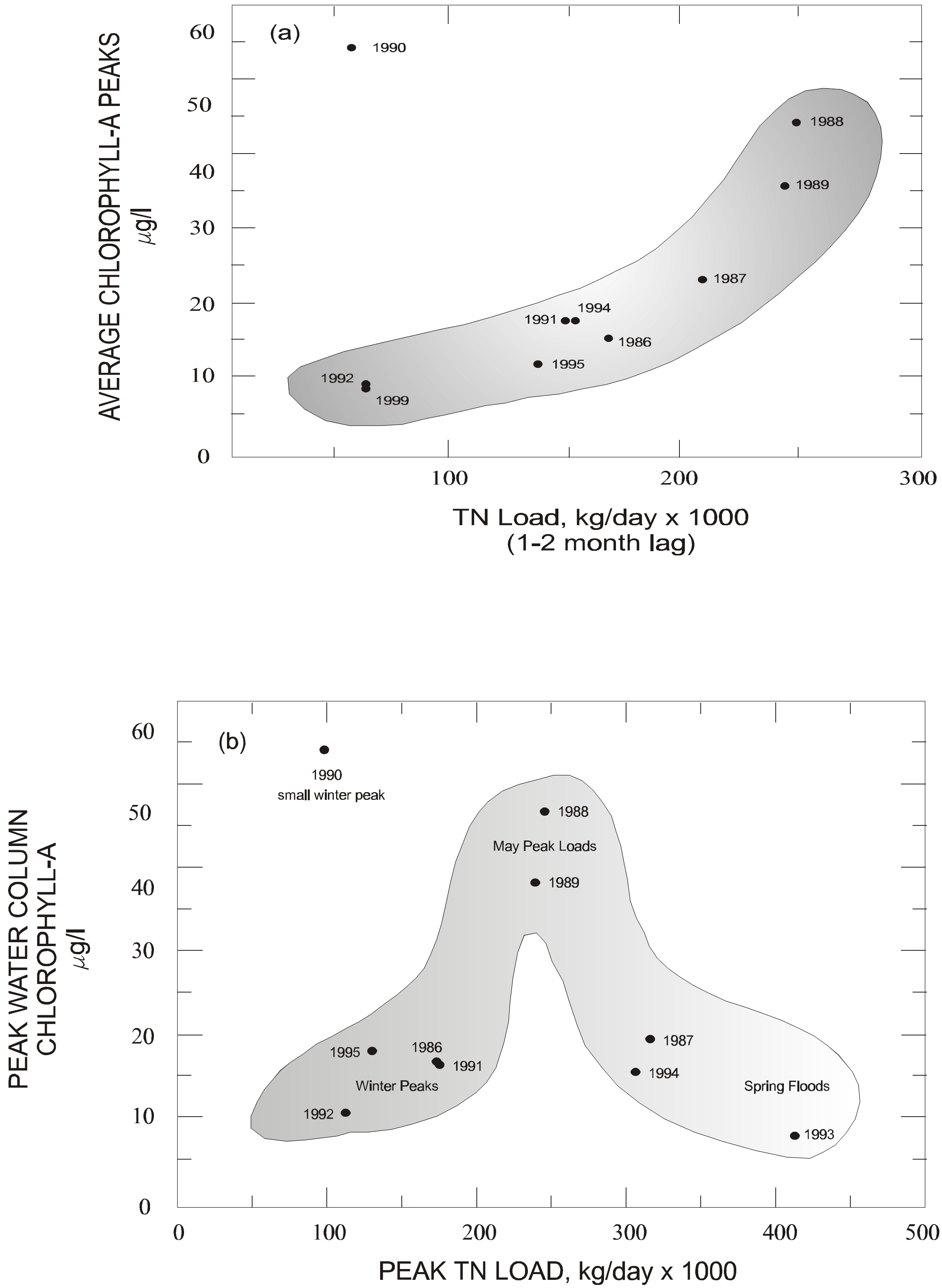

Figure 2-15. Scatter plots of water column averaged chlorophyll *a* at a mesohaline station (MLE 2.2) versus several different functions of total nitrogen (TN) loading rate measured at the fall line of the Potomac River estuary. Source: Boynton 1997.

effects of physical and biological processes on interannual variations in anoxia/hypoxia is only now beginning on the basis of mathematical modeling and long-term empirical monitoring data. Stratification from freshwater inflow from the Susquehanna River apparently is insufficient by itself to explain the increased hypoxic volumes in the Chesapeake Bay from the early 1980s to 1999 (J. Hagy, personal communication). In shallow estuaries the hypoxic volume, if present, is likely to be highly variable spatially owing to the influence of variable freshwater inputs and estuarine in situ physical factors that cause wide excursions and mixing of water masses (e.g., Neuse River estuary, H. Paerl, personal communication).

A detailed study of nutrient and phytoplankton relationships in the mesohaline region of the mainstem of the Chesapeake Bay demonstrated that "despite high inputs of DIN and dissolved silicate relative to DIP (molar ratios of N:P and Si:P > 100), seasonal accumulations of phytoplankton biomass within the salt-intruded reach of the bay appear to be limited by DIN supply while the magnitude of the spring diatom bloom is governed by the dissolved Si supply" (Malone et al. 1996, Conley and Malone 1992). The maximum chlorophyll-specific productivity occurred in the late summer, the maximum biomass occurred in the spring, and volumetric-based productivity occurred in midsummer (see their Figure 4). This temporal asymmetry leads to difficulties in ascribing simple empirical relationships between phytoplankton biomass and nutrient concentrations.

2.4 PHYSICAL-CHEMICAL PROCESSES AND DISSOLVED OXYGEN DEFICIENCY

Dissolved oxygen deficiency, or hypoxia, is of critical importance to the health of aquatic life. The role of physical processes, especially mixing and physical circulation of estuarine waters, has been widely reported in the literature (Smith et al 1992). "There is no other environmental variable of such ecological importance to coastal marine ecosystems that has changed so drastically in such period of time as dissolved oxygen" (Diaz and Rosenberg (1995). One of the earliest studies to measure DO in a U.S. estuary occurred in the Chesapeake Bay and Potomac River in 1912 (Sale and Skinner 1917), approximately two decades after Winkler developed his now legendary method for determining the concentration of DO in aquatic systems. Hypoxia was already present in the bottom waters of the lower Potomac River estuary at this early date because a measurement indicated only a DO < 2.0 ml/L, or 35% saturation.

Individual species exhibit a range in adaptability to relatively low DO concentrations (e.g., see "EPA 822-D-99-002 Draft Ambient Water Quality Criteria for Dissolved Oxygen [Saltwater]: Cape Cod to Cape Hatteras"). Hypoxia and H_2S apparently cause synergetic effects that make marine benthic animals more sensitive to hypoxia when H_2S is present (Diaz and Rosenberg 1995). These authors suggest that the occurrence of hypoxia in shallow coastal and estuarine areas appears to be increasing, and evidence suggests that the increase has global dimensions and seems most likely to be accelerated by human activities (Nixon 1995, Bricker et al. 1999). Although hypoxia has undesirable consequences, when bottom waters go anoxic wholesale biogeochemical changes occur. These changes can include release of phosphate from sediments, emergence of highly toxic hydrogen sulfide, elimination of nearly all multicellular animals from sediment habitats, reduction in the coupled nitrification-denitrification, and changes in metal solubilities, with many metals becoming toxic.

Diaz and Rosenberg (1995) concluded that should DO concentrations become slightly lower, catastrophic events may overcome the systems and alter the productivity base that leads to economically important fisheries and amenities. Aquatic biota exposed to low DO concentrations may be more susceptible to the adverse effects of other stressors such as disease, toxic chemicals, and habitat modification (Holland 1977). Low DO conditions can increase the vulnerability of the benthos to predation, as the infaunal animals extend above the sediment surface to obtain more oxygen (Holland et al. 1987). Dissolved organic carbon apparently is a major carbon and energy source for bacteria (i.e., microbial loop; Azam et al. 1983), whose metabolism is a major cause of hypoxia. Hypoxia and anoxia indicate that a coastal ecosystem is severely stressed by nutrient overenrichment and should receive immediate attention by water quality managers.

2.5 NUTRIENT OVERENRICHMENT EFFECTS AND IMPORTANT BIOLOGICAL RESOURCES

Benthic Vascular Plant Responses to Nutrients

A major lesson learned over the past 25 years is that nutrient overenrichment has had a devastating effect on SAV, whether estuarine species or higher salinity seagrasses. This conclusion is based on work conducted mostly on the U.S. Gulf of Mexico and Atlantic Coasts (Tomasko et al. 1996, Tomasko and LaPointe 1991, Kemp et al. 1983, Orth and Moore 1983, Burkholder et al. 1992, Taylor et al. 1995, Short et al. 1995). Dennison et al. (1993) reported the following habitat criteria for SAV: DIN of 10.7 µM, DIP of 0.33 µM; N:P (atomic) of 32; and chlorophyll *a* of 15 µg/L. These criteria are being re-analyzed by the EPA Chesapeake Bay Program.

The relationship between N load and concentration and chlorophyll *a* is not limited to phytoplankton. Predictive regression relationships between N and chlorophyll *a*, water column light attenuation, and seagrass recovery in Tampa Bay were found for N loading, not ambient N concentrations (Janicki and Wade 1996, Greening et al. 1997). Tomasko et al. (1996) detected a negative correlation between N loads and turtle grass (*Thalassia testudinum*) biomass and productivity in Sarasota Bay, FL.

Moore and Wetzel (2000) determined experimentally that eelgrass (*Zostera marina*) in the York River estuary, lower Chesapeake Bay, is exposed to N concentrations adequate to stimulate enough epiphytic growth to shade out this vascular plant. In mesocosms containing a complex of species characteristic of shallow marine coastal lagoons along the Narragansett Bay coast, Taylor et al. (1995) showed that N alone—but not P alone—caused an increase in water column concentrations of chlorophyll *a* and particulate N, increased daytime net production, and increased growth of juvenile winter flounder. Eelgrass beds and drift algae apparently were shaded out by phytoplankton at high nutrient levels. Experiments conducted by Neundorfer and Kemp (1993) on the submersed plant *Potamogeton perfoliatus* in microcosms using lower Choptank Estuary water demonstrated that effects of N and P on algal densities were synergistic in that responses to N addition were greatest at high P loading and vice versa. Also, combined amendments (N+P) at highest treatment rates resulted in epiphytes and phytoplankton increasing more than when these nutrients were added individually. On the basis of microcosm studies and the literature, Sturgis and Murray (1997) suggested that there may be a more complex relationship between nutrient enrichment and SAV growth and survival. For example, the relationship may depend on the form, delivery frequency, and loading rate of nutrients.

There now appears to be enough scientific data and knowledge to establish nutrient regimes that will protect temperate and subtropical seagrass ecosystems.

Other Examples of Important Biotic Effects of Nutrient Overenrichment

It is difficult to find recent quantitative relationships between nutrient loading and fishery impacts for coastal systems. One explanation is that the large marine vertebrate species which are mostly extinct or severely over-fished help determine the nutrient assimilative capacity of marine ecosystems including estuaries and coastal waters (Jackson et al. 2001). For economically important fisheries, variable fishing pressure may cloud the analysis and other factors may vary to obscure nutrient-related patterns. Often, one is left with mostly anecdotal insights as to potential negative effects of overenrichment on higher trophic levels focusing on data and insights only from recent decades. There is a plausible and positive relationship between marine fisheries yield and nitrogen supply, with a wide range in estuarine and coastal marine habitats represented (Nixon 1992). This approximately natural response is analogous to what mariculturists attempt to achieve when they fertilize fish enclosures, but these enclosures, whether on land or in the marine environment, are known to cause local water quality problems. The relationship Nixon reported on involved a two-step function: a positive relationship between primary production (g C m^{-2} y^{-1}) and DIN input (moles m^{-2} y^{-1}) and between fisheries yield (kg ha^{-1} y^{-1}) and primary production (Figure 2-16a-c). In contrast to the foregoing positive relationship, a pelagic-demersal ratio from fishery landings from 14 study areas in European coastal waters appeared to be a proxy for the differential impact of nutrients on pelagic and benthic systems mediated by nutrient enrichment, resulting in hypoxia (de Leiva Moreno et al. 2000). A general model suggests that overenrichment can lead to decreased fisheries productivity (Figure 2-17).

Oysters are ecosystem engineers that create biogenic reef habitat important to estuarine biodiversity, benthic-pelagic coupling, and fishery production (Lenihan and Peterson 1998). These authors conducted an analysis of habitat degradation (i.e., oyster dredging) through fishery disturbance that enhanced impacts of hypoxia on oyster (*Crassostrea virginica*) reefs in North Carolina. This is a fairly complicated story but the conclusions from the analysis seem inescapable. Dredging lowered the oyster reef into the hypoxic zone where the reef and associated organisms died from DO depletion. Another example of effects of nutrient overenrichment causing impacts on oysters was reported by Ryther (1954) for Long Island, New York duck farms where nutrient enrichment caused phytoplankton to grow that were indigestible for oysters.

Hypoxia is known to kill other benthic organisms. Diaz and Rosenberg (1995) cited many studies where hypoxia resulted in the deaths of benthic communities. A related cause with hypoxia is that polychaetes may extend themselves out of their sediment burrows and become easier prey to fish predators. Another

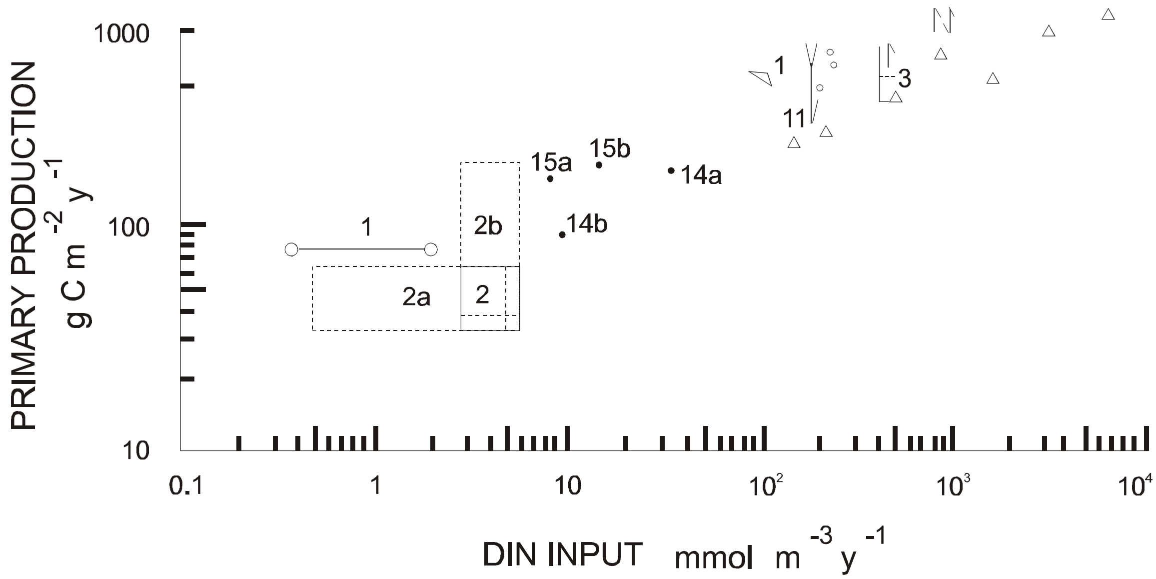

Figure 2-16a. Primary production by phytoplankton ([14]C uptake) as a function of the estimated annual input of dissolved inorganic nitrogen per unit volume of a wide range of marine ecosystems. Source: Nixon (1992).

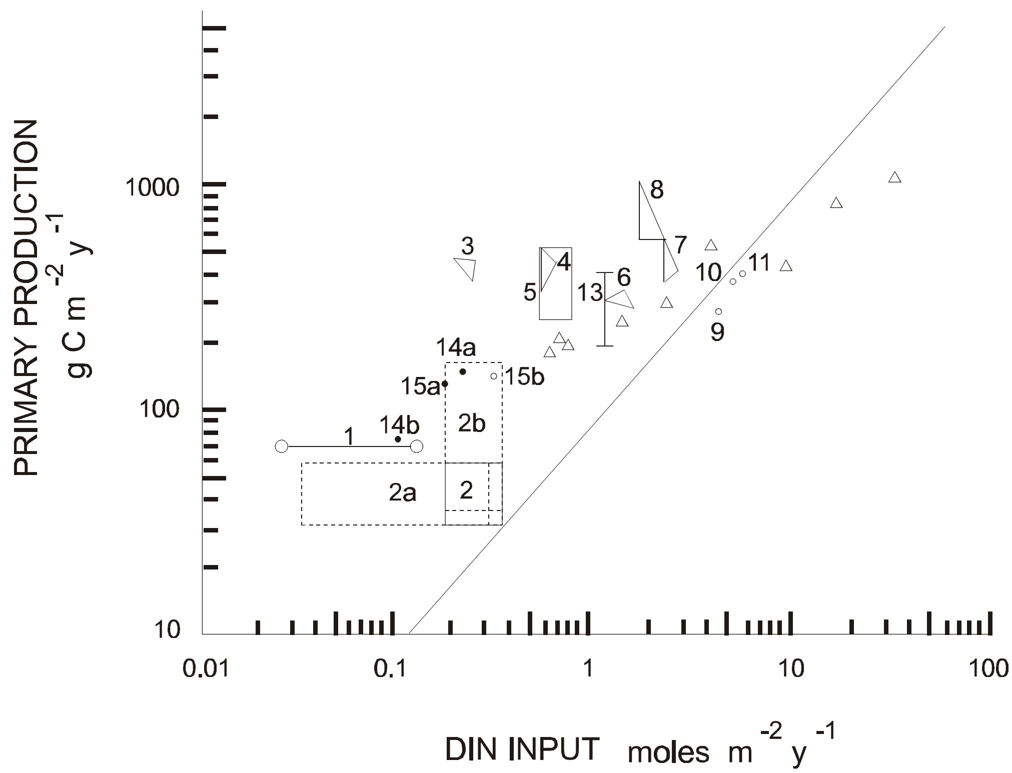

Figure 2-16b. Primary production by phytoplankton ([14]C uptake) as a function of the annual input of dissolved inorganic nitrogen per unit area of a wide range of marine ecosystems. Source: Nixon (1992).

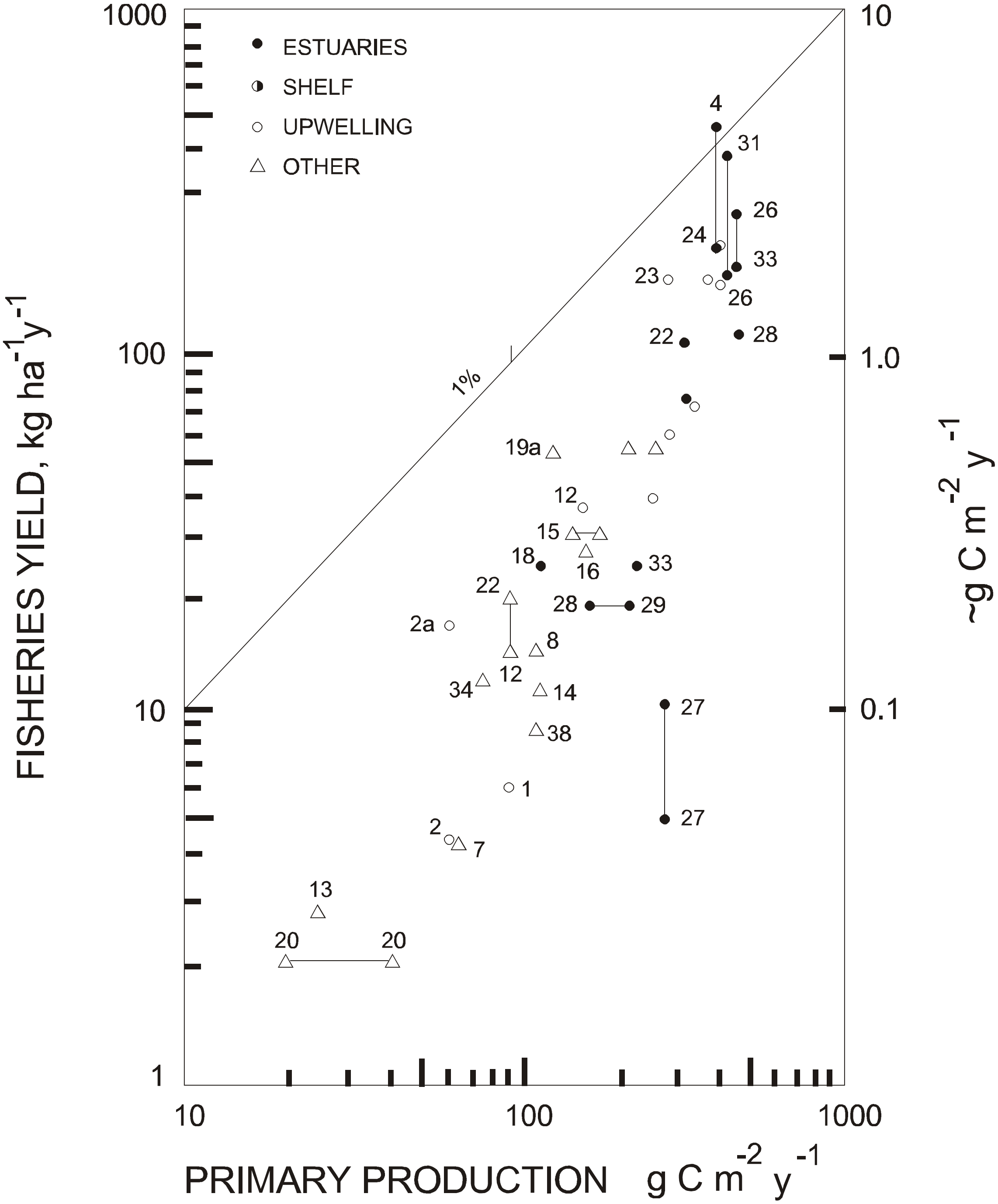

Figure 2-16c. Fisheries yield per unit area as a function of primary production in a wide range of estuarine and marine systems. Modifed from Nixon (1988) to include a revised primary production estimate for the Peru Upwelling from Guillen and Calienes (1981). Systems identified and data sources in Nixon (1982) and Nixon et al. (1986). Source: Nixon (1992).

Although higher nutrient concentrations initially increase the productivity of fisheries, ecological systems worldwide show negative effects as nutrient loading increases and hypoxic or anoxic conditions develop. Each generic curve in the lower half of the figure represents the reaction of a species guild to increasing nutrient supplies. The top half of the figure illustrates trends in various marine systems around the world. Reversals show that trends toward overenrichment have been turned around in several areas.

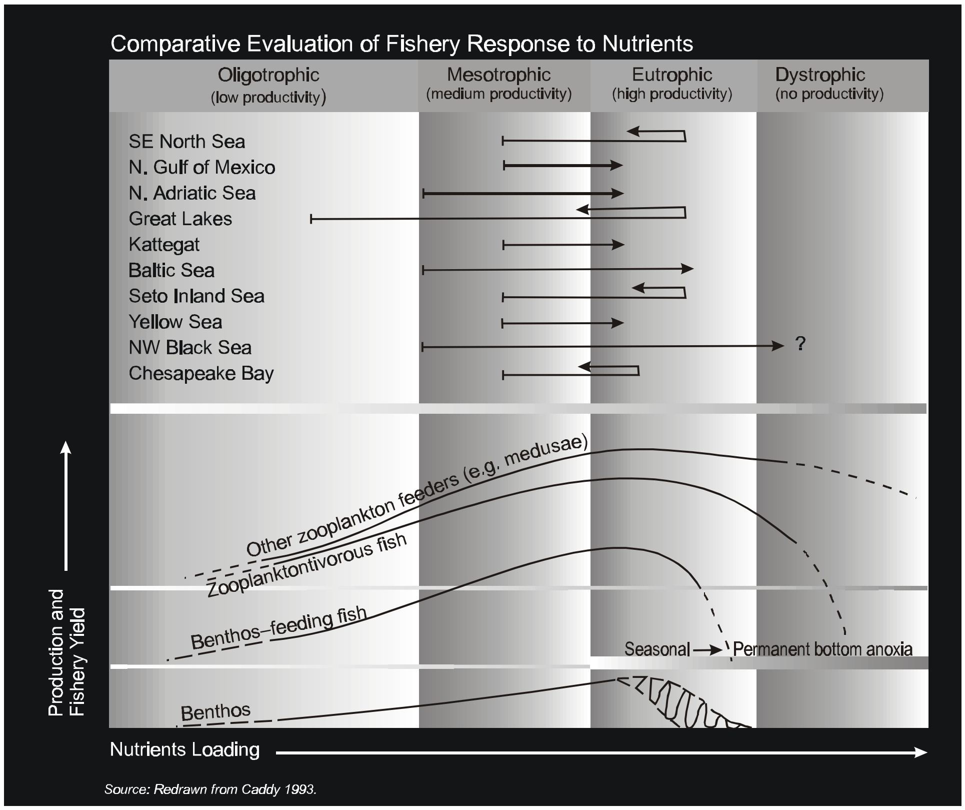

Figure 2-17. Comparative evaluation of fishery response to nutrients. Although higher nutrient concentrations initially increase the productivity of fisheries, ecological systems worldwide show negative effects as nutrient loading increases and hypoxic or anoxic conditions develop. Each generic curve in the lower half of the figure represents the reaction of a species guild to increasing nutrient supplies. The top half of the figure illustrates trends in various marine systems around the world. Reversals show that trends toward overenrichment have been turned around in several areas. Source: CENR 2000.

effect of hypoxia on the biota is the loss of sufficient bottom habitat. This is often difficult to quantitatively relate to economically important species but the negative effect may still be real. If endangered species are present, this hypoxic effect is one of direct societal and legal concern.

2.6 CONCLUDING STATEMENT ON NITROGEN AND PHOSPHORUS CONTROLS

It is important to note that in estuaries and nearshore coastal marine waters, the fact that nitrogen often limits algal biomass production does not mean that managers should be unconcerned about phosphorus enrichment. In river-dominated temperate estuaries, the upper reaches of estuaries, such as lakes and rivers, are often phosphorus limited. The manager who therefore concentrates on phosphorus management alone risks letting an undue amount of nitrogen proceed downstream to exacerbate problems where an abundance of P allows the excess N to drive trophic conditions to unacceptable levels of nutrient enrichment.

Similarly, any reductions achieved in P loadings and concentrations at the coastal margin will limit potential eutrophy/hypertrophy even in the face of abundant nitrogen. Consequently, the prudent management strategy is to limit both phosphorus and nitrogen. Emphasis on one or the other as an element of symptomatic management in fresh or saline waters may be appropriate in some cases, but the manager must always be concerned about the downstream consequences and the net enrichment effects to the larger system.

In summary, attempting to understand the nutrient overenrichment problem in estuaries and coastal ecosystems primarily from a bottom-up perspective provides a limited perspective. This manual has included references to the historical past that reported on potential positive effects of top-down controls on nutrient overenrichment. It is likely that the most scientifically robust nutrient criteria will need to take into account the effects of past overfishing and its consequences for marine eutrophication (Jackson et al. 2001). Thus, higher trophic levels are more than just a thermodynamic response to nutrient enrichment because they help modulate many of the negative consequences of overenrichment. Ecological feedback mechanisms that involve higher trophic levels can be a positive tool in nutrient management.

CHAPTER 3

Classification of Estuarine and Coastal Waters

Major Factors Influencing Estuarine Susceptibility to Nutrient Overenrichment
Examples of Coastal Classification
Coastal Waters Seaward of Estuaries

3.1 INTRODUCTION

Purpose and Background

Classification is an important step in addressing the problem of degradation, especially because of nutrient overenrichment. There are too many nutrient-degraded estuaries in the United States for the Nation to conduct comprehensive ecosystem studies of all those affected by overenrichment. Where possible, similar estuaries, tributaries, or coastal reaches should be equated through physical classification to reduce the magnitude of the criteria development problem and to enhance predictability of management responses. To be useful, classification should reduce variability of ecosystem-related measures (e.g., water quality factors) within identified classes and maximize interclass variability. This is important because managers need to understand how different types of estuaries and coastal waters, as well as important habitat differences within these systems, respond to nutrient overenrichment in order to plan effective management strategies.

The ecosystem processes that regulate nutrient dynamics, discussed in Chapter 2, should provide the elements for initial development of a useful classification system. Although predicting susceptibility of estuarine and coastal waters to nutrient overenrichment is in a primitive state, several approaches are reviewed because they have some utility even if they are only marginally adequate for prediction of nutrient effects. The general approach is also appropriate for coastal systems.

General trends relate N loading with chlorophyll and primary productivity; however, these trends are seldom usefully predictive for individual systems or for all classes of coastal systems (Kelly in press). Progress has been especially slow in predicting many of the secondary, but societally important, effects of nutrient overenrichment, e.g., bottom water dissolved oxygen (DO) deficiency, harmful algal blooms (HABs) or species-specific HABs, formation of macroalgal mats, fisheries productivity, and species composition. For many cross-system comparisons, N loading and SAV decline have become more predictive than for other indirect effects (Duarte 1995; Dennison et al. 1993), but even here the predictions may be confounded by highly variable ecosystem factors. A major impedance to effective understanding is limited comparative studies designed to test hypotheses regarding estuarine susceptibility to nutrient enrichment (Turner 2001). Post hoc comparative approaches and assessment of disparate studies have been useful but clearly inadequate (Livingston 2001b).

Post hoc statistical approaches have helped explain some of the variability in eutrophication, but have not captured the actual mechanisms and their interactions controlling eutrophication across estuaries and coastal waters (NRC 2000). The ability to explain the mechanisms in a predictive manner is clearly a critical national and global research need, as nutrient overenrichment of coastal ecosystems extends far beyond the shores of the United States. For site-specific criteria, several approaches are available, including empirical regression and mechanistic simulation models. The large effort typically required to calibrate and verify mechanistic models is an indicator of the difficulty in understanding the many potential confounding factors of ecosystem-level prediction. A basic premise of this manual is that knowledge of the physical setting and the minimally disturbed ecosystem reference condition must underpin monitoring and management efforts to protect and restore coastal systems impaired by nutrient overenrichment.

Only in approximately the past two decades have comparative studies of nutrient dynamics among two or more relatively large estuaries been published (e.g., Fisher et al. 1988; Malone et al. 1999; Pennock et al. 1994). Comparative analysis of the Delaware and Chesapeake Bays has provided insights regarding processes that control expression of nutrient enrichment (Chapter 2). For example, both systems are drowned river mouth coastal plain estuaries and are located adjacent to each other along the coast, but have very different responses to nutrient loading. Delaware Bay has a somewhat larger nutrient load than the Chesapeake, but has few of the nutrient enrichment symptoms well chronicled for the Chesapeake Bay (Flemer et al. 1983, Sharp et al. 1994, Chesapeake Bay Program Periodic Status Reports). Similar insights have been provided by comparing nutrient processes between Delaware and Mobile Bays (Pennock et al. 1994). Susceptibility appears to be largely explained by differences in the physics of flushing, including bathymetry and related physical habitat differences.

Defining the Resource of Concern

As a first step in classification, defining the resource of concern is important. Resources of concern are estuaries and coastal waters located in the contiguous States or within authorized Tribal lands. Managers must decide which waterbodies to include in the population to which criteria will be applicable. A lake classification may exempt small ponds that might be excluded because of their size and man-made nature, whereas tidal creeks, although small, still have a functional connection to the larger estuary and might not be excluded because of size. Many estuaries and coastal waters share multiple political boundaries, and for the sake of consistency all involved jurisdictions should jointly decide on the scale of inclusiveness. For open coastal waters, a State or authorized Tribe’s legal authority may extend for a relatively short distance on the continental shelf, e.g., 3 nautical miles. However, coastal oceanographic processes seaward of the statutory limit likely influence nutrient overenrichment processes and exacerbate the difficulty of diagnosing the anthropogenic contribution to nutrient problems.

3.2 MAJOR FACTORS INFLUENCING ESTUARINE SUSCEPTIBILITY TO NUTRIENT OVERENRICHMENT

The NRC (2000) publication summarized approximately a dozen factors deemed important to characterize the susceptibility of estuaries to nutrient loading. A short list is provided; however, it is expected that the following list will be modified and refined as more is learned about the subject:

1. System dilution and water residence time or flushing rate
2. Ratio of nutrient load per unit area of estuary
3. Vertical mixing and stratification
4. Algal biomass (e.g., chlorophyll *a*, and chlorophyll *a* corrected for nonchlorophyll *a* light attenuation over seagrass/SAV beds and macroalgal biomass as AFDW)
5. Wave exposure (especially relevant to seagrass potential habitat)
6. Depth distribution (bathymetry and hypsographic profiles)
7. Ratio of side embayment (s) volume to open estuary volume or other measures of embayment influence on flushing.

Several terms listed above are briefly discussed because their significance often is not adequately appreciated.

Dilution

The volume of an estuary affects its ability to dilute inflowing nutrients. Thus, the loading rate of nutrient per unit volume of the estuary is a better indicator of the potential for exceeding the assimilative capacity of the estuary as a whole than is the absolute loading rate. This ratio may not express the potential for local effects near the point of entry into the estuary, as nutrients there are diluted by only a fraction of the total estuary volume. The potential for such local effects is reduced if mixing into the main body of the estuary is rapid.

Water Residence Time

Estuaries that flush rapidly (i.e., have a short residence time) will export nutrients more rapidly than those that flush more slowly, resulting in lower nutrient concentrations in the estuary. Dettmann (in press) has derived a theoretical relationship between the mean residence time of freshwater in an estuary and the increase in the average annual concentration of total nitrogen in the estuary as a result of inputs from the watershed and atmosphere. In addition, estuaries with residence times shorter than the doubling time of algal cells will inhibit formation of algal blooms. Residence time or flushing rate is discussed in more detail in Appendix C.

Stratification

Highly stratified systems are more prone to hypoxia than are vertically mixed systems. Stratification not only limits downward transport of oxygen from atmospheric reaeration, it also retains nutrients in the photic zone, making them more available to phytoplankton. In stratified systems, it may be more appropriate to estimate the dilution potential of the estuary using the volume above the pycnocline rather than the entire volume of the estuary.

It is expected that the shortened list will be revised and modified as more is learned about factors important in estuarine and coastal waters classification. Some of these factors will apply to estuaries and others to coastal waters.

3.3 EXAMPLES OF COASTAL CLASSIFICATION

Scientists and resource managers have used various classification schemes for many years to organize information about ecological systems. As discussed earlier, estuaries and coastal water systems are characterized by a suite of factors (e.g., river flow, tidal range, basin morphology, circulation, and biological productivity) that are ultimately controlled largely by geology and climate. A review of Chapter 6 in the NRC (2000) publication provides useful descriptions of the various approaches to estuarine classification, and some pertinent features are highlighted below.

Geomorphic Classification

Geomorphic classification schemes provide some insight into the circulation structure and are a first-order estimate of water residence time or flushing characteristics. Such classifications may not in themselves be predictive of susceptibility to nutrient enrichment, but they are a useful place to begin a first-order assessment of susceptibility. Knowledge of deep channels, however, identifies potential areas subject to hypoxia, and the extent of shallow waters and associated factors (e.g., wind fetch) often provides insights into potential seagrass habitat.

Estuaries can be divided geomorphically into four main groups (Pritchard 1955, 1967; Dyer 1973): (1) coastal plain estuaries, (2) lagoonal or bar-built estuaries, (3) fjords, and (4) tectonically caused estuaries. This classification frequently appears in textbooks, and only some important features relative to nutrient susceptibility are described.

Coastal Plain Estuaries: Classical and Salt Marsh

Both subclasses are characterized by well-developed longitudinal salinity gradients that influence development of biological communities. Examples of the classical type include the Chesapeake Bay (the largest estuary of this type), Delaware Bay, and Charleston Harbor, SC. Vertically stratified systems with relatively long residence times (e.g., Chesapeake Bay) tend to be susceptible to hypoxia formation. Pritchard (1955) further classified drowned river valley estuaries into four types (A-D) depending on the advection-diffusion equation for salt (Table 3-1). Type C estuaries are less sensitive to algal bloom formation and hypoxia because of mixing features.

The salt marsh estuary lacks a major river source and is characterized by a well-defined tidal drainage network, dendritically intersecting the extensive coastal salt marshes (Day et al. 1989). Exchange with the ocean occurs through narrow tidal inlets, which are subject to closure and migration following major storms (e.g., Outer Banks, NC). Consequently, salt marsh estuarine circulation is dominated by freshwater inflow, especially groundwater, and tides. The drainage channels, which seldom exceed a depth of 10 m, usually constitute less than 20% of the estuary, with the majority consisting of subaerial and intertidal salt marsh. These systems are a common feature of the Atlantic coast, particularly between Cape Fear, NC, and Cape Canaveral, FL. Mangrove estuaries occur from around Cape Canaveral south on Florida’s east coast and on Florida’s west coast from around Tarpon Springs south. Nutrient dynamics, primary production, and system respiration that occur within emergent marshes may greatly affect water quality in the estuarine channels (Cai et al. 1999).

Table 3-1. General drowned river valley estuarine characteristics

Estuarine type[a]	Dominant mixing force	Mixing energy	Width/depth ratio	Salinity gradient	Mixin g index[b]	Turbidity	Bottom stability	Biological productivity	Example
A	River flow	Low	Low	Longitudinal vertical	≥1	V. high	Poor	Low	Southwest Pass Mississippi River
B	River flow, tide	Moderate	Moderate	Longitudinal vertical	<1/10	Moderate	Good	V. high	Chesapeake Bay
C	Tide, wind	High	High	Longitudinal lateral	<1/20	High	Fair	High	Delaware Bay
D	Tide, wind	V. high	V. High	Longitudinal	?	High	Poor	Moderate	?

[a]Follows Pritchard's advection-diffusion classification scheme.
[b]Follows Schubel's definition: MI = equation here▸ (vol. freshwater discharge on ½ tidal period) / (vol. tidal prism).
Source: Neilson and Cronin, 1981.

Lagoons

Lagoons are characterized by narrow tidal inlets and are uniformly shallow (i.e., less than 2 m deep) open-water areas. The shallow nature enhances sediment–water nutrient cycling. Flushing is typically of long duration. Most lagoonal estuaries are primarily wind-dominated and have a subaqueous drainage channel network that is not as well drained as the salt marsh estuary. Lagoons fringe the coast of the Gulf of Mexico and include the mid-Atlantic back bays; Pamlico Sound, NC; and Indian River Lagoon, FL. Although these systems are typically shallow, they may have pockets of hypoxic water subject to spatial variability because of freshwater pulsing and wind effects. Some lagoonal systems have relatively strong vertical stratifications near the freshwater river mouth and may be subject to hypoxia formation (e.g., Perdido Bay, AL/FL; Livingston 2001a).

Fjords and Fjordlike Estuaries

Classical fjords typically are several hundred meters deep and have a sill at their mouth that greatly impedes flushing. Hypoxia/anoxia is often a natural feature but anthropogenic nutrient loading can severely exacerbate the problem. Examples of classical fjords on the North American continent can be found in Alaska and Washington State (Puget Sound). Some other estuaries were also formed by glacial scouring of the coast, but in regions with less spectacular continental relief and more extensive continental shelves. Examples of these much shallower, fjordlike estuaries can be found along the Maine coast.

Tectonically Caused Estuaries

Tectonically caused estuaries were created by faulting, graben formation (i.e., bottom block-faults downward), landslide, or volcanic eruption. They are highly variable and may resemble coastal plain estuaries, lagoons, or fjords. San Francisco Bay is the most studied estuary of this type (Cloern 1996).

Man-Made Estuaries

Especially around the Gulf of Mexico, dredged bayous, canals, and salt water impoundments with weirs function as estuaries but do not fit well any of the other types presented. As a special case, especially in the Gulf of Mexico, the passes of some estuaries periodically were closed off by storms and historically remained closed until a natural event reopened them (e.g., Perdido Bay, AL/FL; R. Livingston, personal communication). In recent years, these systems typically are maintained in an open condition by dredging. Dredged inlets such as at Ocean City, MD, also fit this classification.

Physical/Hydrodynamic Factor–Based Classifications

Classification Using Stratification, Mixing, and Circulation Parameters

Estuarine circulation was a dominant consideration used in an earlier classification of the Chesapeake Bay, a coastal plain system, and major tributaries, and is largely utilized today with some modifications (Flemer et al. 1983). Coastal plain estuaries are sometimes classified by mixing type: highly stratified, partially mixed (moderately stratified), or well mixed (vertically homogeneous). The flow ratio of these estuaries (the ratio of the volume of freshwater entering the estuary during a tidal cycle to its tidal prism) is a useful index of the mixing type. If this ratio is approximately 1.0 or greater, the estuary is normally highly stratified; for values near 0.25 the estuary is normally partially mixed; and for ratios substantially less than 0.1, it is normally well mixed (Biggs and Cronin 1981).

Stratification/Circulation Parameters

Hansen and Rattray (1966) developed a two-parameter classification scheme based on circulation and stratification of estuaries. Circulation is described by the nondimensional parameter U_s/U_f, where U_s is the net (time-averaged) longitudinal surface current and U_f is the cross-sectional average longitudinal velocity. Stratification is represented by the nondimensional parameter $\delta S/S_o$, where δS is the top-to-bottom difference in salinity and S_o is the mean salinity. Jay et al. (2000) review alternative two-parameter classification schemes involving parameters such as the ratio of tidal amplitude to mean depth, along-estuary and vertical density differences and vertical tidal excursion of isopycnals, or other factors that take into account effects of tidal flats and provide additional discussion to which the reader is referred for additional insights. They argue that the merit of the approach is its simplicity of parameters employed and the predictive ability with regard to salt transport needed to maintain salt balance in modeling.

Classification Using Water Residence Time

Water residence time, the average length of time that a parcel of water remains in an estuary, influences a wide range of biological responses to nutrient loading. The residence time of water directly affects the residence time of nutrients in estuaries, and therefore the nutrient concentration for a given loading rate, the amount of nutrient that is lost to internal processes (e.g., burial in sediments and denitrification), and the amount exported to downstream receiving waters (Dettmann in press, Nixon et al. 1996). Residence times shorter than the doubling time of algae will inhibit bloom formation because algal blooms are exported from the system before growing to significant numbers. Residence time can also influence the degree of recruitment of species reproducing within the estuary (Jay et al. 2000).

There are a number of definitions of water residence time, including freshwater residence time and estuarine residence time (Hagy et al. 2000; Miller and McPherson 1991), each with its own interpretation and utility. Freshwater residence time is the mean amount of time required for freshwater entering the estuary to exit the seaward boundary, whereas estuarine residence time is the average residence time in the estuary for all water, regardless of its origins. Because nutrient loading is generally associated with freshwater inputs, freshwater residence time is generally the most useful measure in considering estuary sensitivity to nutrient loading. Freshwater residence time of a given estuary is influenced by numerous factors, including freshwater loading rate (Pilson 1985; Asselin and Spaulding 1993; Hagy et al. 2000), tidal range, and wind forcing (Geyer 1997), and therefore varies over a range of time scales.

Residence time and volume together may be used to scale nitrogen loading to estuaries to permit calculation of nitrogen concentrations and perform cross-system comparisons. Dettmann (in press) uses a model that includes mechanistic representations of nitrogen export and loss within estuaries to show that $[N_u]$, the contribution to the annual average concentration of total nitrogen in an estuary from upland sources (watershed, direct discharges, and atmosphere), may be calculated as

$$[N_u] \quad \left(\frac{L_l \tau_{fw}}{V}\right) \frac{1}{1 \quad \alpha\tau_{fw}}$$

where L_l is the annual average loading rate (mass/month) of total nitrogen from all upland sources (watershed and atmosphere), J_{fw} is the freshwater residence time in months, V is the estuary volume, and " is a parameter (value = 0.3 month^{-1}) related to losses of nitrogen to processes such as denitrification and burial in sediments within the estuary.

Definitions of residence time, the methods used to measure or calculate them, variability of residence time, and other estimators of residence time are described further in Appendix C.

River Flow, Tides, and Waves

Dronkers (1988) proposed an estuarine classification that distinguished various types of estuarine ecosystems based on water exchange processes (e.g., river flow, tides, and waves) that greatly affect energy and material fluxes including mixing (Table 3-2). This classification suggests that river flow in partially mixed estuaries is essentially neutral, but its variation relative to hydrodynamic residence time can be important in interpreting property-salinity diagrams (Cifuentes et al. 1990) (Figure 3-1). River flow in the partially mixed mainstem of the Chesapeake Bay is seasonally important.

Tidal Amplitude—A Dominant Physical Factor

Tidal amplitude provides a means to broadly classify estuaries relative to their sensitivity to nutrient supplies. Monbet (1992) analyzed phytoplankton biomass in 40 estuaries and concluded that macrotidal estuaries (mean tidal range $\geq$2 m) generally exhibit a tolerance to nitrogen pollution despite high loadings originating from freshwater outflows (Figures 3-2a, b). These systems generally exhibit lower concentrations of chlorophyll *a* than do systems with lower tidal energy, even when they have comparable concentrations of nitrogen compounds. Estuaries with mean annual tidal ranges $\leq$2 m seem more sensitive to dissolved nitrogen, although some overlap occurs with macrotidal estuaries.

NOAA Scheme for Determining Estuarine Susceptibility

NOAA (Bricker et al. 1999) developed a categorical approach based on surveys and decision rules that led to a classification of estuarine nutrient export potential (e.g., dilution potential and flushing potential). From this information a susceptibility matrix was constructed. The low, moderate, and high susceptibility indices were combined with low, moderate, and high human levels of nutrient input, resulting in a final matrix of overall human influence (see Appendix D for details).

Comparative Systems Empirical Modeling Approach

The empirical regression method can be used to determine the response of estuarine systems to nutrient loading. This approach requires that the response factor be common to all systems in the analysis and assumes that any graded response among systems is due to a common form of disturbance, e.g., nutrients. The space-for-time paradigm (Pickett 1988) posits that relationships between nutrient inputs and

Table 3-2. Classification of coastal systems based on relative importance of river flow, tides, and waves to mixing

Type	River flow	Tide	Waves	Description
I		–	–	River delta
II		–		River delta (plus barriers)
III			–	Tidal river delta
IV	0		–	Coastal plain estuary
V	–			Tidal lagoon
VI	–		–	Bay
VII	–	–		Coastal lagoon

Plus and minus designations indicate relative impacts; e.g., – means that river discharge is very small relative to tidal and wave energy.
Source: Adapted from Dronkers 1988.

ecologically meaningful estuarine responses, using multiple systems, have predictive capability, at least for the systems used in the model development. This allows for a wide range in nutrient loading and estuarine types to be included. The comparative-systems empirical approach has been used to determine, for example, relationships between nutrient inputs and fish yields (Lee and Jones 1981, Nixon 1992), benthic biomass, production and abundances (Josefson and Rasmussen 2000), summer ammonia flux (Boynton et al. 1995), chlorophyll *a* concentration (Boynton et al. 1996, Boynton and Kemp 2000, Monbet 1992), primary productivity (Nixon et al. 1996), and the dominant source of primary productivity (Nixon et al. in press). In many of these cases, important environmental factors such as flushing time and depth are used to normalize the nutrient loading in a similar way as Vollenweider (Vollenweider 1976) Did for lakes to yield more precise relationships. Appendix E provides additional details.

Other Considerations

Habitat Type

The presence and extent of different habitat/community types may help distinguish one or more estuaries within a region. These types may include seagrasses, mangroves, mudflats, deep channels, oyster reefs, dominance of sand versus mud bottoms, extensive emergent marshes (typically coastal plain systems), and the presence of unconsolidated versus rocky shorelines. Some of these categories may be subclassified by salinity ranges (e.g., oligohaline, mesohaline, and polyhaline). Although related more to water quality, blackwater versus turbid versus relatively clear estuaries defines a group representative of estuaries around the Gulf of Mexico.

Theoretical Considerations

Coastal zone managers may wish to consider more theoretical approaches to classification as ecosystem science develops a more in-depth understanding of ecosystem processes for estuaries under their purview. Several different approaches are described in Appendix F.

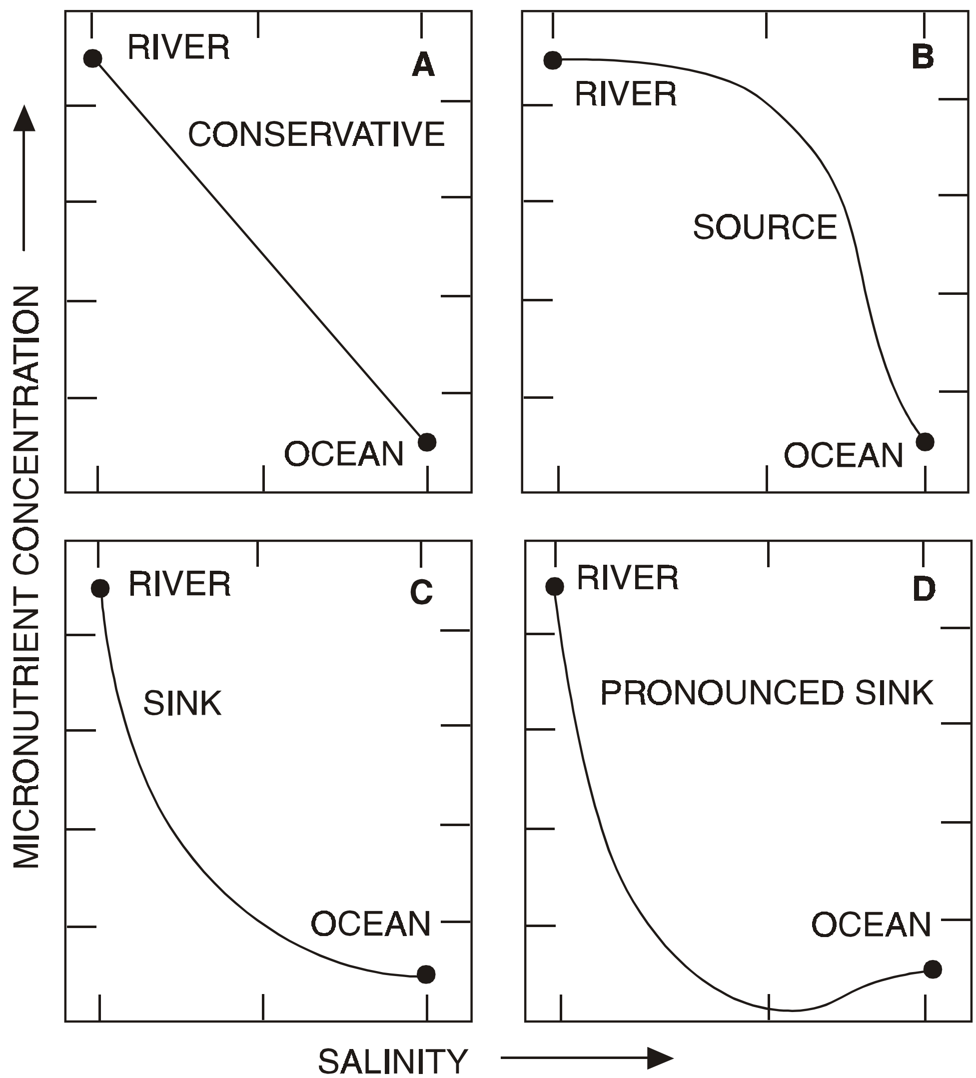

Figure 3-1. Idealized micronutrient-salinity relations showing concentration and mixing of nutrient-rich river water with nutrient-poor seawater. Source: Peterson et al. 1975. A. Expected concentration-salinity distribution of a substance behaving in a conservative manner (e.g., chloride) in an estuary. B. Expected concentration-salinity distribution of a substance for which the estuary is a source (e.g., particulate carbon). C. Expected concentration-salinity distribution of a substance for which the estuary is a sink (e.g., phosphorus). D. Expected concentration-salinity distribution of a substance for which the estuary is a pronounced sink, that is, where the concentration of the substance in the estuary is lower than the river and the ocean (e.g., Si). Source: Biggs and Cronin 1981.

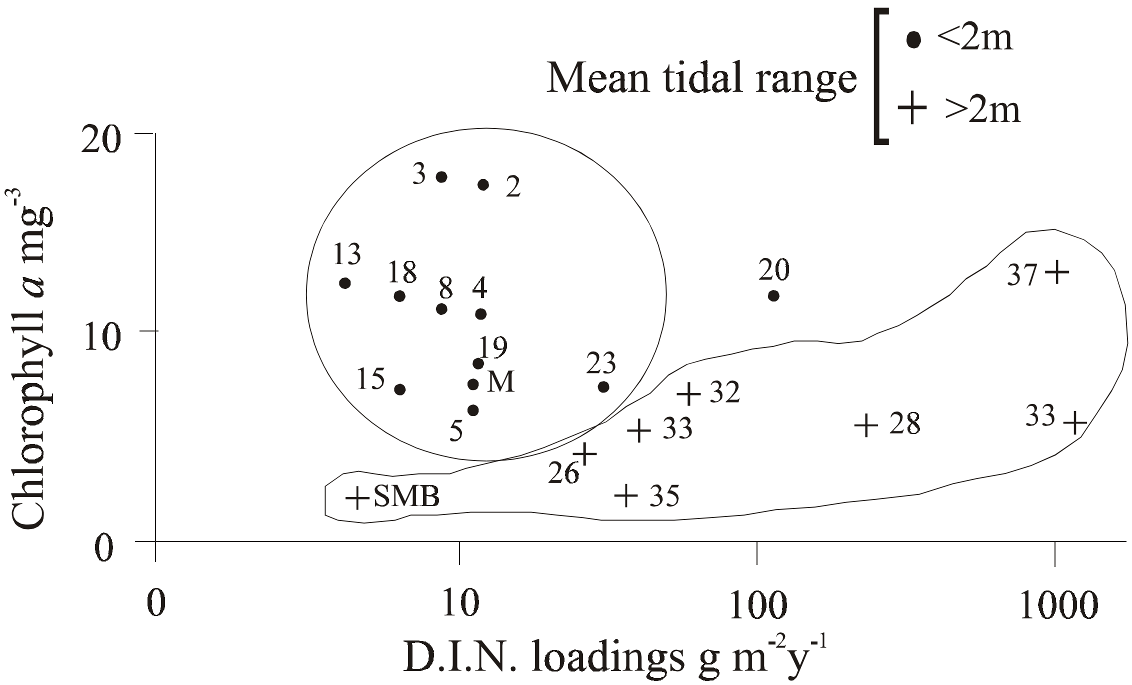

Figure 3-2a. Relationship between the mean annual loadings of dissolved inorganic nitrogen (DIN) and the mean annual concentration of chlorophyll *a* in microtidal and macrotidal estuaries.

Summary

Various ways are available to classify estuaries regarding their vulnerability to nutrient enrichment. None appear to provide all the information a resource manager may want for decisionmaking. The NOAA estuarine export potential (EXP) appears to have the current greatest utility for predictive purposes for large systems, but even this approach embodies considerable variability (e.g., see Figures 6-5 and 6-6 in NRC 2000). For embayments within a larger estuary, the comparative empirical modeling approach has been demonstrated to have considerable utility. The more theoretical models eventually may provide greater predictive power, especially as to biological sensitivities to nutrient enrichment. They are data intensive and may become more useful at a future time.

3.4 COASTAL WATERS SEAWARD OF ESTUARIES

Several approaches are available to classify coastal waters. The geomorphic focus is a good place to begin, hydrographic considerations should follow, and finally habitat and community features should be considered. Although functional considerations and theoretical indices are not described for coastal waters, they have as much relevance for these waters as they do for estuaries. Even though much of the concern for coastal waters will be within 20 nautical miles of shore, and most of that within the 3-mile limit, elements of the following large-scale classification scheme will have value to the manager and investigator.

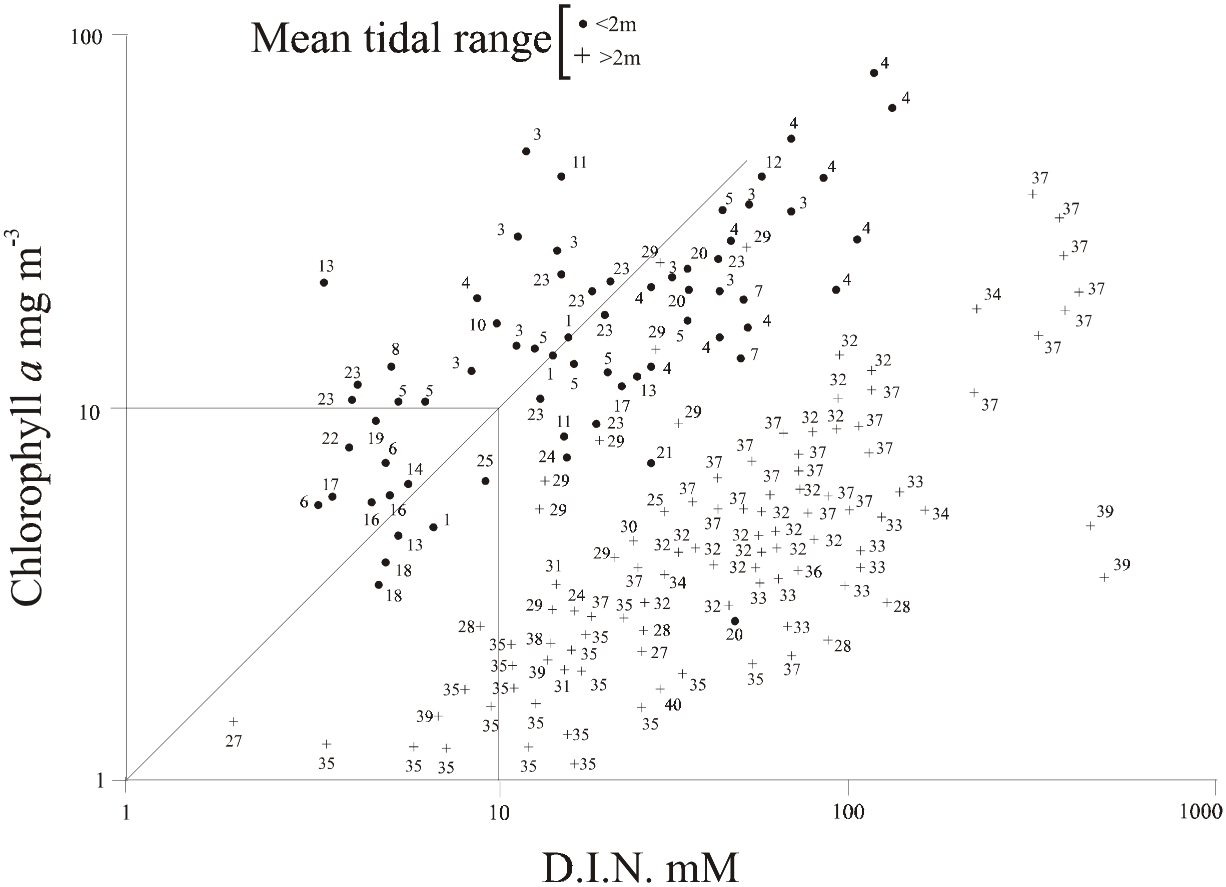

Figure 3-2b. Relationship between the mean annual concentrations of dissolved inorganic nitrogen (DIN) and chlorophyll *a* in microtidal and macrotidal estuaries. Source: Monbet 1992.

Geomorphic Classification

The flow of energy and nutrients through coastal food webs differs greatly among continental shelves, and is driven largely by differences in the form and amount of primary production (e.g., seagrasses are important in the Big Bend area of Florida and kelp forests are important habitats along much of the U.S. Pacific Coast and sections of coastal Maine). These differences in turn ultimately are determined by differences in local and ocean-scale patterns of climate (e.g., light and temperature effects), water circulation, chemistry, and shelf geomorphology (Alongi 1998). The spring bloom, especially along the U.S. Atlantic Coast, generally progresses from low to higher latitudes but with sharper seasonal peaks toward higher latitudes. Variability in the progression should be considered in any classification scheme. Because near-coastal shelf oceanographic processes usually are not limited by the jurisdiction of a single State, it is important that a similar classification approach be shared among coastal States, where that oceanography determines the sensitivity of the ecosystem to nutrient enrichment. The geographic extent of the shelf in which a State has jurisdiction is a useful place to begin classification. Here one should consider whether the shelf is wide or narrow (e.g., mid-Atlantic versus Pacific Coast). The Texas coastal shelf is very wide, with a gentle slope compared with much of the northern Gulf of Mexico. The steepness of the slope is another useful factor, as it may influence bottom sediment stability and upwelling. The degree of bottom roughness or sculpture may influence vertical mixing, which may in turn influence water column stability and depth of the euphotic zone versus mixing depth.

Nongeomorphic Classification

Walsh characterized the world's continental shelves on the basis of their location, major rivers, and rates of primary production, and included some U.S. coastal waters for comparison. The shelf proper is where oceanic and estuarine boundaries often intermingle. At the shelf edge, cold, nutrient-rich water and associated materials intrude onto the shelf. There, exchanges often are rapid, promoting conditions favorable for higher fertility than in the open ocean. Higher primary productivity is the main reason why approximately 90% of the world's fish catch is harvested on the continental shelves versus the open sea (Alongi 1998).

Water Quality Trend Detection on the Shelf

Physical mixing and advective processes may add considerable variability to water column measures. Therefore, it is important to consider detection of trends in nutrient concentrations and measures such as chlorophyll *a* based on comparisons at a reference salinity (e.g., 30 psu). Otherwise, classification schemes may incur extraneous variability. A common approach is to use "mixing diagrams" to compare measured changes in an ambient constituent among sampling periods. At mid- and higher latitudes, winter measures of DIN and DIP may provide insight into long-term trends of changes in nutrient concentrations available to drive the spring bloom. At low latitudes winter values will likely have less applicability, as primary production has a smaller seasonal signal.

Presence of Large Rivers

Although large rivers are included in Walsh's characterization of shelf systems, it seems useful to distinguish shelf areas based primarily on large rivers, such as the Mississippi River in the Gulf of Mexico and the Columbia River off the Washington-Oregon coast. Large rivers on the shelf dominate local ecological relationships.

Hydrographic Features

Vertical salinity differences tend to decrease toward the open ocean boundary. The principal reason is summertime thermal stratification. The thermocline tends to be deeper toward the open sea margin, except where buoyancy effects are associated with large rivers that flow onto the shelf. Coastal waters contain a variety of biotic communities, including a diverse assemblage of macroepifauna and -infauna, kelp forests, coral reefs, bottom and pelagic fishes, marine mammals, and seabirds. The relationship of these communities to physical-ordering factors can assist in classification.

Temperate and subtropical coastal waters also experience a seasonal sea-level fluctuation, whereby summer levels rise approximately 0.2 m by upper-ocean heat expansion, producing what is known as thermosteric effects (Pattulo et al. 1955, Bell and Goring 1998). This nontidal process operates in conjunction with other factors affecting apparent mean sea level (e.g., near and far-field wind effects and barometric pressure). Depending on local conditions, water levels overlying the continental shelf and in estuaries can rise from 0.1 to 0.2 m. Such a rise may seem nominal but can have a significant impact in wetlands and other low-lying areas that potentially exchange nutrients and suspended sediments with the coastal ocean.

Coastal ocean waters range from quite cold (e.g., Gulf of Maine) to quite warm (e.g., Gulf of Mexico). Where large rivers enter coastal waters, such as the Mississippi River Plume and Chesapeake Bay Plume, visual discoloration can be observed because suspended material from land runoff and relatively high plankton concentrations contrast with the predominantly blue color of the open ocean. The Columbia River and the Mississippi River form an "estuary" mostly at sea, as very little of the diluted seawater is bounded by land.

Physical gradients are dynamic and change at multiple scales. Seasonal or wet and dry periods frequently differ depending on the various shelf gradients associated with estuarine, riverine, and ocean/shelf break processes. Regional geomorphology and physical mixing processes play a pivotal role in energy flow and material cycles. For example, the Loop Current in the eastern Gulf of Mexico may show seasonal reversals and vary seasonally in its penetration onto the shelf. Along-shore drift inside the north-flowing Gulf Stream off the Mid-Atlantic Bight tends to transport materials southward toward the North Carolina coast. Further south, the Gulf Stream forms a seaward boundary that tends to significantly isolate in-shore waters from those beyond the shelf break. Local current maps are available from the National Ocean Service of NOAA (www.noaa.gov; then click on nos).

Many different types of boundaries or fronts occur in coastal seas, but no formal classification exists. Alongi (1998) lists five categories:

- Shelf-sea (tidal) fronts
- Estuarine fronts or plumes
- Shelf-break fronts
- Upwelling fronts
- Island wakes and fronts caused by other land features

Fronts provide increased physical stability at local scales, which may positively influence primary production and energy flow to higher trophic levels (see Chapter 2).

Habitat/Community Differences

Presence of Mangrove/Seagrass and Coral Communities

Along the southeastern Florida Atlantic coast exists a combination of mangrove, seagrass, and coral reef ecosystems. In some localities, each community type may dominate the others, but often they co-occur. Seagrass communities may dominate certain shelf areas along the west coast of Florida (e.g., Big Bend region). The Flower Gardens, a disjunct coral community, exist off the southern coast of Texas. Alongi (1998) devotes chapters to coral reefs and mangrove ecosystems including factors regulating primary productivity (e.g., N and P). The role of N and P enrichment versus grazing in coral reef ecosystems is still strongly debated in the scientific literature (e.g., Miller et al. 1999). A paper by Chen and Twilley (1999) discusses soil nutrient relationships and productivity in a Florida Everglades mangrove ecosystem along an estuarine gradient (see the references cited above for the most recent perspective). These distinctive ecosystems provide a basis for local coastal waters classification. Mangrove communities also occur along the lower Texas coast, and seagrasses are a dominant community in Laguna Madre, TX.

Presence of Seaweed

Seaweeds are common algal communities in rocky intertidal zones (e.g., *Fucus* spp.), attaching themselves by means of a holdfast. Seaweeds belong to three marine algal classes: *Chlorophyceae* (green algae), *Rhodophyceae* (red algae), and *Phaeophyceae* (brown algae). The kelps (*Laminariales*), members of the brown algae, live subtidally but in relatively shallow waters and can form large forests along the cooler north Atlantic and Pacific coasts. These communities also may occur in the higher salinity reaches of estuaries. Alongi (1998) provides a discussion of primary production, factors limiting growth, nutrient cycling, and grazing in these communities.

CHAPTER 4

Causal and Response Variables
Field and Laboratory Methods
Nutrient Enrichment and Ammonia Toxicity

Variables and Measurement Methods To Assess and Monitor Estuarine/Marine Eutrophic Conditions

4.1 INTRODUCTION

This chapter provides an overview of several measurable trophic state variables that can be used to establish nutrient criteria for estuaries and nearshore coastal waters. Trophic state variables are those variables that can be used to evaluate or predict the trophic status or degree of nutrient enrichment of estuaries and nearshore coastal waters, especially when compared with reference conditions. The primary variables include two causal variables (TN and TP) and two response variables including a measure of algal biomass (e.g., chlorophyll *a* for phytoplankton or macroalgal biomass (AFDW) and water clarity, e.g., Secchi depth or electronic photometer), and the addition of dissolved oxygen, as appropriate. These variables are relevant at the national scale to practically all estuaries and are potentially relevant to nearshore coastal waters.

Several variables are important indicators of nutrient overenrichment for a large number of estuaries, but in many cases the data and supporting science are inadequate for most systems (e.g., algal species composition). Important secondary variables include seagrass and estuarine submerged aquatic vegetation (SAV) distribution and abundance, macroinfaunal community structure, phytoplankton species composition, and organic carbon concentrations, respectively. Seagrasses and SAV typically provide important shallow water habitat information, and hypoxia/anoxia are measures of loss of bottom habitat often associated with deeper waters. Organic carbon (total, particulate, and dissolved) is also included as a secondary variable because this variable is consistent with Nixon's (1995) definition of eutrophication. Changes in benthic macroinfaunal community structure often correlate with organic carbon enrichment and degree of hypoxia and anoxia (Diaz and Rosenberg 1995). The importance of algal species composition has implications for food webs (Roelke 2000). These variables are discussed in Chapter 2.

As indicated in Chapter 2, the concentration of the primary nutrient variables may not correlate well with one or more response variables in estuaries, especially hypoxia or anoxia and measures of phytoplankton biomass. In this case, predictive relationships should be attempted with nutrient loads using first empirical regression models or other statistical approaches if necessary to account for ecosystem-based nonlinearities. Application of mechanistic computer models is another approach (see Chapter 9).

Interpretation of nutrient enrichment indicators, especially for estuaries, is complicated by the interaction with measures of mixing and flushing as discussed in Chapters 2 and 3. Salinity gradients are associated with flushing but also play an important role in the type of biological communities exposed to nutrient enrichment. These physical considerations must always play a part in nutrient enrichment predictions including establishment of reference conditions as discussed in Chapters 2 and 6.

4.2 CAUSAL AND RESPONSE INDICATOR VARIABLES

Nutrients as Causal Variables

Nitrogen

Nitrogen is one of the most important limiting nutrients of autotrophic assemblages (e.g., phytoplankton and periphyton) incorporated into estuarine and nearshore coastal marine bioassessments. In those estuaries where N has been demonstrated to limit algal biomass production, it typically does so at higher salinities along the salinity gradient (Chapter 2). Most research has focused on the role of inorganic-N as a stimulant to algal biomass production (Stepanauskas et al. 1999). However, about 70% of the dissolved N transported by rivers worldwide (10^{12}g yr^{-1}) is dissolved organic N (DON) (Meybeck 1982). In contrast to P, control of N sources is more difficult because diffuse gaseous sources of N (N_2) can be assimilated directly from the atmosphere by N fixation, a process conducted by a variety of bacteria and cyanobacteria (blue-green algae). Also, dissolved inorganic N forms, especially nitrite and nitrate, are highly soluble and do not precipitate easily or sediment out when freshwater enters the brackish zone of estuaries as inorganic P is likely to do.

Total N measured as a water quality indicator consists of organic and inorganic forms. Although some dissolved organic N may be used for algal growth, especially if remineralized by bacterioplankton (Carlson and Graneli 1993; Seitzinger and Sanders 1999), it and particulate organic forms participate in algal biomass production through recycling processes (Chapter 2). In systems with hypoxic or anoxic conditions, the rate of decomposition is reduced. Although still an open question, apparently relatively little of the DON is directly utilized by phytoplankton, except for urea and free amino acids (Antia et al. 1991; Paerl et al. 1999). Dissolved organic N in rainwater (synthetic addition of urea and other constituents in bioassays) was shown experimentally to stimulate bacterioplankton and phytoplankton growth; however, the DON resulted in the dominance of diatoms and dinoflagellates whereas ammonium-N stimulated production more of small monads (Seitzinger and Sanders 1999). Further work is required to test whether this response is widely applicable. Thus, the source of DON can influence the degree of DON utilization by the microbial community. Inorganic N consists of ammonia, nitrite, and nitrate N. Ammonia N is a primary product of microbial degradation of organic N, and, if not used directly by autotrophic algae and vascular macrophytes and microbial heterotrophs for growth, it may be oxidized through nitrification to nitrite and nitrate. Varying proportions of organic N may be relatively refractive and contribute very little to N overenrichment problems. However, the readily recyclable component may contribute to N enrichment problems locally and further seaward. Some experimental or model analysis (e.g., box model) of the utilization of DON and in some cases particulate organic N for each coastal system is usually warranted.

In estuaries, N concentrations, especially the inorganic forms, typically vary widely seasonally, interannually, and along salinity gradients. In temperate river-dominated estuaries, nitrate concentrations may reach very high concentrations (e.g., >100 µM) in tidal fresh to brackish reaches (see Appendix G;

Neilson and Cronin 1981) due to wash-off associated with various land use activities including point and nonpoint sources (e.g., agricultural cropland). By late spring to early summer, the nitrate concentration may be below analytical detection limits. Nitrite concentrations seldom reach high levels in surface waters due to plant utilization and conversion to nitrate through nitrification. The principal bacteria genera that mediate nitrification include *Nitrosomonas*, but species of *Nitrosococcus*, *Nitrobacter*, and *Nitrospina* are also important (Sharma and Ahlert 1977, Watson et al. 1981). If dissolved oxygen is limiting nitrification, then nitrite may accumulate (Helder and de Vries 1983). Ammonia concentrations in open estuarine and nearshore coastal waters located away from point sources typically vary from below detection limits to approximately 1.0 to 5 µM, depending on growing season and rates of organic N decomposition. Much higher values may occur for relatively short periods. The ionized form of ammonia/ammonium is the most abundant reduced form and represents approximately 97% of the total (Sillen and Martell 1964). The equilibration between the ionized and un-ionized fractions is controlled by temperature, salinity, and pH, resulting in a range of un-ionized ammonia of 1% to 5% of the total at typical salinities, pH, and temperature (Emerson et al. 1975). Ammonia may be toxic to marine larvae, not just a stimulus to algal growth. Unionized ammonia concentrations in the range of 1.0µM approximate those that are known to be toxic to marine larvae, especially molluscs (U.S. EPA 1989). Denitrification may remove from a few to approximately 50% of the TN load entering temperate estuaries annually (Seitzinger 1988, Cornwell et al. 1999) depending largely on residence time of the water, sediment biogeochemical conditions (macroinfauna present to maintain irrigation, oxic conditions in the overlying bottom water), and water column depth. This process helps to modulate extreme DIN concentrations (Chapter 2). Typical values for dissolved inorganic N (DIN) and a few TN concentrations in estuaries and coastal nearshore waters are presented in Appendix G as a basis to help establish expectations for various coastal systems. It should be noted that N concentrations vary widely in space and time and the values in Appendix G are only intended to be rough guides. Specifics of analytical techniques to measure the various forms of N are included at the end of this chapter (Field Sampling and Laboratory Analytical Methods).

In open coastal waters of the North Atlantic Ocean at temperate latitudes, there is a typical seasonal progression in DIN and DIP concentrations associated with phytoplankton blooms. The spring bloom reduces these inorganic forms while phytoplankton biomass accumulates. This progression begins at lower latitudes and moves to higher latitudes. The spring bloom typically crashes in late spring, and summer biomass levels often are nutrient limited. Often a small bloom occurs in the fall following the fall thermocline breakdown that allows mixing and replenishment of nutrients from deeper waters into the upper surface layers, where a short burst of production occurs before light becomes limiting. Accumulation of deepwater nutrients during the winter has been used to assess the potential for spring-summer overenrichment in coastal seas based on trends in “salinity-nutrient mixing diagrams” (European Union Northern Marine Eutrophication Criteria Program, Ulrich Claussen, Germany, personal communication). Seasonal nutrient patterns in estuaries are quite variable. In some estuarine systems, a winter buildup of N and P has been observed (e.g., Patuxent River Estuary), especially when freshwater flows remained low and point sources dominated the nutrient supply (e.g., Flemer et al. 1970). Mixing diagrams also help interpret nutrient behavior in estuaries; however, some precautions are important to recognize (e.g., see Sharp et al. 1986).

At the interface between fresh and marine waters, a process occurs that results in an apparent increase in the ionized ammonia concentration. This process is apparently driven by the increased electrolyte solution of the salts, which has a significant impact on the production and nitrification process, thus yielding higher ionized ammonia levels (Rysgaard et al. 1999). Ionized ammonia adsorption to particles was decreased, especially in the 0 to 10% salinity range, as were the nitrification and denitrification processes. Further evaluation showed that the reduction in nitrification and denitrification processes was due not only to the displacement of bacteria and ionized ammonia from particles, but also to decreased bacterial activity. The projections from these studies were that ionized ammonia would be produced at a rate of 1 µM/g of sediment in the water. The changes in N dynamics that affect adsorption of suspended solids may need to be included when considering acceptable levels in fresh water sources to estuaries.

Phosphorus

Phosphorus is an important plant nutrient that may limit algal biomass production in tidal fresh to brackish zones of estuaries and some subtemperate to tropical marine coastal systems (Chapter 2). There are no common stable gaseous forms of phosphorus, so the phosphorus cycle is endogenic, without an atmospheric component (Manahan 1997). The main natural reservoirs of phosphorus are poorly soluble minerals (e.g., hydroxyapatite) in the geosphere. Erosion of these materials from terrestrial sources and their transport to the sea are important sources of new phosphorus in seawater. The phosphorus entering the sea is mostly orthophosphate, PO_4^{-3} (Kennish 1989). In previous decades, prior to widespread phosphate bans in detergents, estuaries received a considerable portion of P from detergents. The ban resulted for many estuarine systems in an elevated DIN:DIP ratio. In estuaries and nearshore coastal waters, phosphorus is present in dissolved inorganic form as well as dissolved and particulate organic form. Some fraction of P may be strongly embedded in a mineral matrix, and this renders that fraction relatively inert to biological utilization. For this reason, often measures of TP may represent some component that is not biologically available and managers should consider this in developing P criteria. Plants directly take up the phosphates as essential nutrients during photosynthesis. Some algae have the capability to break down dissolved organic P (DOP) with alkaline phosphatase (algal and free phosphatases) and utilize the phosphate as inorganic phosphate (Huang and Hong 1999). Alkaline phosphatase apparently is located on phytoplankton cell membranes, which makes it difficult to determine whether the uptake is direct for DOP or the DOP undergoes enzymatic hydrolysis on the cell membrane. Malone et al. (1996) suggested by inference that Chesapeake Bay phytoplankton may utilize organic sources of P, in part, because the DIN:DIP thresholds approach 160, which is considerably greater than the N:P ratio reported by Redfield et al. (1963). Orthophosphates are typically preferred by autotrophic phytoplankton, although some assimilation of organic phosphorus may occur, especially during periods of P deficiencies (Boney 1975). When plants die, or are eaten, the organic phosphorus is rapidly converted to orthophosphates through the action of phosphorylases within fecal material, phosphatases in the plant cells, and finally by bacteria (Riley and Chester 1971).

To summarize, phosphorus occurs in natural waters and in wastewaters almost solely as phosphates. These are classified as orthophosphates, condensed phosphates, and organically bound phosphates (common analytes are total phosphorus [TP] and dissolved or particulate organic phosphorus [DOP, POP]). These compounds may be soluble, in particulates or detritus, or incorporated as organic P in organisms. Phosphorus is essential to the growth of organisms and can limit phytoplankton biomass production, which is most commonly observed in freshwater systems (Hecky and Kilham 1988) and some estuaries and coastal marine systems (Chapter 2). In instances where phosphate is limiting, the discharge of raw or untreated wastewater, agricultural drainage, or certain industrial wastes may stimulate the growth of algae. Appendix G provides examples of P concentrations in several forms.

Silica

Silica, as an important algal nutrient, has received much less attention in estuarine nutrient overenrichment studies than N and P based on the limited volume of literature citations (e.g., see Malone et al. 1996) and recent reviews of estuarine eutrophication (Chapter 1). Silica limitation of diatom production, a major algal group that requires Si (and silicoflagellates), often is a measure of N or P overenrichment (D'Elia et al. 1983; Conley and Malone 1992). Dissolved Si is a product of weathering and erosion of rocks on land with subsequent transport to the sea (Conley and Malone 1992). Because Si has essentially no human sources, except possibly from erodible soils under human influence, it is not a strong candidate for regulation. In some parts of the ocean, organisms (such as diatoms and radiolarians) abound that have produced skeletons of a noncrystalline form of hydrated silica-opal. As these skeletons settle to the sea floor they slowly dissolve, releasing silica. Officer and Ryther (1980) predicted that increases in N and P to estuaries and coastal waters from human activities, coupled with the reduction in silicates to the sea from construction of artificial lakes, would alter the N:Si and P:Si ratios. These alterations were postulated to alter phytoplankton populations to reduce the relative abundance of diatoms and enhance the relative abundance of flagellates. Egge and Aksnes (1992) showed that diatoms always numerically dominated the phytoplankton community when concentrations of silica were in excess of 2.2 µM. Dominance by diatoms ceased or became more variable when concentrations of Si were less than this value.

Ryther and Officer (1981) reinterpreted the relationship of N pollution in Long Island Inlets during the 1950s. Nitrogen may have limited the nuisance *Nannochloris*, blooms but they hypothesized that the bloom persisted because diatoms had been eliminated by Si depletion. Also, the degree of Si limitation of spring diatom blooms in Chesapeake Bay that fuel summer anoxia has direct ecological implications (Conley and Malone 1992, Malone et al. 1996). Freshwater sources of Si dominate estuarine supplies (Fisher et al. 1988). Typically, Si limitation can be potentially deduced from ambient ratios relative to the nutrient-sufficient N:Si:P biomass ratios of 16:16:1 (Redfield et al. 1963; Conley et al. 1993). In Chesapeake Bay, the dissolved Si:DIP ratio often approximates 100-300 (Malone et al. 1996), suggesting strong Si limitation. Significant increases in Mississippi River N and P concentrations and loading and decreases in silicate have occurred during the 20th century (Rabalais et al. 1996). The increased P loading and associated increased diatom production and eventual burial in river sediments, as predicted by Officer and Ryther (1980), has resulted in a reduced Si supply to the coastal environment. The consequence is that diatom production, generallly a preferred phytoplankton group to support higher trophic levels, is now more Si limited than in previous decades. The N:P:Si ratios on coastal Louisiana and Texas now suggest the possibility of a joint nutrient limitation of phytoplankton production.

Silica concentrations for the Coastal Texas/Louisiana coast averaged approximately 5.3 µM in the late 1980s but averaged about 9.0 µM during the early 1960s. Silicate concentrations in the Chesapeake and Delaware Bays and the Hudson River Estuary ranged from about 90 to near detection levels, 30 to near detection limits, and 30 to 3 µM, respectively (Fisher et al. 1988). Eyre and Balls (1999) reported that Si was less likely to limit diatom production in tropical estuaries than in temperate ones because concentrations tend to be much higher in tropical estuaries.

The role of silica may be more important to diatom species composition and food quality as future research may document. More attention in the future should be given to the measurement and assessment of the role of Si in estuarine and nearshore coastal primary productivity and food web dynamics and as a basis for controlling co-limiting N and/or P.

Response Variables

Chlorophyll a and Macroalgal Biomass

Chlorophyll *a* is the molecule mediating photosynthesis in most all green plants (except prochlorophytes, which contain divinyl chlorophyll), including phytoplankton; it is relatively easy to measure either spectrophotometrically or by fluorescence and is commonly used to indicate phytoplankton biomass. However, the amount of chlorophyll per cell can vary widely. Conversion factors from weight of chlorophyll to weight of carbon (a desired biomass unit) can vary by a factor of 10. Adaptation to light levels is the primary reason for observed variability; photoadaptation can cause the chlorophyll per cell to vary widely. The technology for measuring chlorophyll has greatly improved over the decades. The Welschmeyer (1994) fluorometric analysis reduces the interference due to chlorophyll *b* and phaeopigments. The HPLC procedure is capable of detecting and quantifying various pigments characteristic of different algal groups (e.g., diatoms, cyanophyta, chlorophyta, and dinoflagellates) (Jeffery et al. 1997).

Rapid proliferation or blooming of phytoplankton, as reflected in chlorophyll *a* measurements, occurs throughout the ocean but is most often associated with temperate coastal and estuarine waters and at higher latitudes. In winter months, growth of phytoplankton populations is generally minimal because of insufficient light and also because a turbulent and unstable upper water column carries the phytoplankton cells below the euphotic zone (where light is not sufficient) before they can divide.

Chlorophyll *a* concentrations vary widely as a function of nutrient supply, water column stability, euphotic zone depth (light availability), sinking, grazing, disease organisms (e.g., viruses), and flushing/mixing (Chapter 2). Values in excess of 12 to 15 µg/L are likely to cause severe shading of seagrasses (Kelley in press). Concentrations in estuaries during summer optimum growing conditions may exceed 50 to 80 µg/L when nutrient loading is high (Monbet 1992). Summer values in the range of 20 to 40 µg/L are frequently observed in enriched estuaries. In contrast, concentrations in overenriched temperate U.S. estuaries during the winter may decrease to 1 to 5 µg/L Nearshore coastal areas removed from high nutrient loads may experience chlorophyll concentrations in the range of approximately 1 to 3 µg/L (Appendix G). Very high values may occur during the summer under conditions of high levels of nutrient enrichment (e.g., the Mississippi River Plume on the Texas/Louisiana Shelf [Rabalais et al. 1996]).

Macroalgal biomass, especially benthic unattached forms (i.e., *Ulva* spp.), often becomes abundant in relatively shallow estuaries that experience nutrient overenrichment. In estuaries that receive most of their nutrient load from groundwater (e.g., Waquoit Bay, Cape Cod, MA; see Chapter 2) benthic macroalgae may shade out seagrasses. Continued enrichment typically leads to reduction of macroalgae as phytoplankton predominate in the water column. Macroalgae are difficult to adequately sample for chlorophyll *a*, and thick mats often contain sheets of algal material that has begun to degrade. The most common method to sample benthic macroalgae is to collect samples and express the biomass on a dry weight basis.

Measures of Water Clarity

Light Attenuation Coefficient

The Secchi disc has been a mainstay as a tool in estimating water clarity; however, this simple and inexpensive tool does not provide all of the information required to distinguish the light attenuation effects of living phytoplankton pigments (i.e., traditionally estimated by chlorophyll *a*) from other factors (e.g., inorganic suspended sediments, organic nonchlorophyll-based detritus, and humic-like materials) that reduce water clarity. EPA's Chesapeake Bay Program (Chapter 2) has developed an analytical approach that partitions the effect of chlorophyll *a* from total suspended solids that contribute to reduction in water clarity. This approach has been used successfully in estimating the combined factor contribution to light attenuation over submerged aquatic vegetation beds (Dennison et al. 1993). In turbid coastal waters, the analyst should be aware of lower values for the constant 1.7 to estimate the light attenuation coefficient (see Giesen et al. 1990 and references in Chapter 2). More precise estimates of the light attenuation coefficient can be made with electronic submersible light meters including PAR meters (photosynthetic active radiation) and submersible spectral radiometers. These meters are now in widespread use, and their use should be encouraged because they give a direct measure of light attenuation, especially in shallow water where depth may limit use of the Secchi disc.

Attenuation of light in the sea in nonalgal bloom areas is determined principally by the amount of suspended matter present, but in estuaries and nearshore coastal waters, color from humic-like materials may significantly compete with particulate material in light attenuation. In moderately turbid coastal waters, 1% of the surface visible light energy may penetrate to a depth of only 10 to 20 m, but in shallow estuaries depths often are from 10 cm to 3 m or so. There typically is a strong seasonal variability in water clarity in temperate estuaries between the active growing season and the winter, and in subtemperate to tropical estuaries water clarity often is a function of the wet season. In the Atlantic temperate open coastal areas with the coming of spring, the depth of the euphotic zone often increases and the depth of the mixed layer decreases because of the development of the seasonal thermocline. This allows a spring bloom to develop. The thermocline tends to confine the algal cells to the euphotic zone, which becomes rich with nutrients as a result of winter mixing. In estuaries, the pycnocline may also have this effect. In partially mixed estuaries where light is adequate at depth, diatoms may grow below the pycnocline (Malone et al. 1996). If the necessary growth-promoting factors are also present, conditions are optimal for proliferation of phytoplankton from seed stock, which may be either the plankton cells themselves or their resting stages (Riley and Chester 1971).

Secchi Depth

The Secchi disc is a useful tool to estimate water clarity (Holmes 1970). Secchi disc measurements often have a longer historical record than electronic measurements, which facilitates assessment of trends in water clarity. Secchi depth measurements are obtained with a 40 cm plastic or metal Secchi disk that is either white or is divided into black and white quadrants on a nonstretchable line that is calibrated in decimeters. The disc should be weighted to maintain a level position, especially under strong current conditions. The disk is lowered into the water until it disappears from view and the depth is recorded. The disk is then slowly raised to the point where it reappears and the depth is recorded again. The mean of these two measurements is the Secchi depth. Observations are made from the shady side of the vessel to reduce problems of glare; however, when a small boat is used for field work a "viewing tube" allows readings under full sunlight conditions. Measurement should be made without sunglasses.

Dissolved Oxygen

Dissolved oxygen (DO) is an integrative measure of ecosystem health and habitat function. As a first-order estimate, the percent saturation of surface and bottom waters is an index of the production/respiration ratio. Dissolved oxygen in bottom waters serves as a measure of habitat availability for benthic animals and pelagic animals that feed on the bottom. EPA has developed saltwater DO criteria for coastal waters between Cape Cod and Cape Hatteras (see www.epa.gov/ost/standards/dissolved). Profiles of DO are indicative of oxygen depletion conditions such as hypoxia and anoxia. Lack of oxygen in bottom waters causes sediment to release dissolved nutrients including orthophosphorus, ammonia, and in addition, toxic hydrogen sulfide.

Carbon Compounds

Organic matter content is typically measured as total organic carbon (TOC) and dissolved organic carbon and is an essential component of the carbon cycle. The rate of organic carbon production and decomposition and the resulting microbial biomass are at the heart of the eutrophication problem. Evaluation of the carbon-containing compounds in an aquatic ecosystem can indicate its organic character. The larger the carbon or organic content, the greater the growth of microorganisms that can contribute to the depletion of oxygen supplies. TOC is a more convenient and direct expression of organic carbon content than are the biochemical oxygen demand (BOD), assimilable organic carbon (AOC), or chemical oxygen demand (COD) methods. TOC is independent of the oxidation state of the organic matter and does not measure other organically bound elements, such as N and hydrogen, or inorganics that can contribute to the oxygen demand measured by BOD and COD. In spite of its versatility, TOC does not provide the same kind of information as BOD, AOC, or COD, and should not be used to replace these methods.

At the surface of the sea, the concentrations of particulate and dissolved organic carbon range up to 12.5 µM and between 75 and 150 µM, respectively. In coastal environments, concentrations of dissolved and particulate organic carbon are greater by factors of ~7-fold. Concentrations of dissolved and particulate organic carbon in surface waters are equivalent to 150 to 1,800 µg C/L (Millero 1996). Organic carbon represents approximately 50% of the dissolved and particulate organic material in seawater (Millero 1996). However, the major form of carbon in seawater is associated with inorganic carbonate systems.

Benthic Macroinfauna

Benthic macroinfauna are an important biological component of estuarine and nearshore coastal marine ecosystems. These communities contribute to benthic food webs, contribute to nutrient cycling and system productivity through benthic-pelagic coupling of nutrient recycling, help stabilize bottom habitats, and contribute to marine biodiversity. Benthic infaunal communities are quite diverse within an estuary or coastal region. Diversity is a function of salinity, with higher diversities associated with higher salinities (Carriker 1967). Sediment irrigation provided by benthic infauna enhances denitrification by increasing the flux of ammonium into oxic microenvironments where nitrification can occur and the flux of nitrite and nitrate into the anoxic sediment zone where denitrification becomes possible (Chapter 2).

4.3 FIELD SAMPLING AND LABORATORY ANALYTICAL METHODS

The following sections provide additional information on field sampling and laboratory methods for selected variables. A list of suggested methodologies for analysis of biochemical parameters is provided in Table 4-1. These methods have been summarized from nationally or regionally recognized reference compendiums (APHA 1998, ASTM 1976, U.S. EPA 1979, Spotte 1992) and provide acceptable methods for determining the concentrations of nutrients as well as acceptable methods for measuring the effects of those nutrients in estuarine and marine waters.

Field Sampling Methods

Nutrients, Hydrography, and Sediments

Physiochemical profiles should be recorded for each field sampling station. Important parameters to be measured include water temperature, pH, dissolved oxygen, salinity, light attenuation, surface radiation, and total depth. Generally, a multiparameter water quality instrument CTD is used. Sampling depth will vary depending on specific objectives; however, enough vertical depth reading should be taken to characterize the physical structure of the water column. For example, CDT measurements might be taken at frequent intervals in the vicinity of the pycnocline (e.g., every 0.1 m in highly stratified estuaries). Overall current dynamics can be mapped with oceanographic tools such as current meters, drift cards, and acoustic Doppler sounders.

Field sampling of discrete water samples for laboratory analysis can be performed using standard nonmetallic plastic water bottles. Samples are drawn into prelabeled bottles and fixatives are applied as appropriate to the subsequent analysis. Nutrient and organics samples are stored on ice until reaching a shoreside sample handling location. Nutrient samples are filtered using graduated syringes and then frozen. Samples for total TN and TP are filtered or unfiltered as appropriate, and 20 mL of sample is frozen for analysis. See Chapter 5 for additional sampling protocols.

Table 4-1. Suggested methods for analyses and monitoring of eutrophic conditions of coastal and marine environments (* = primary EPA preferred causal and response variables)

Eutrophication indicators	Suggested methods	Detection limit or range	Comments	References
Field				
*Water clarity	Secchi depth	0.1 m	—	EPA 903-R-96-006
pH	CTD probe	0.01 pH	—	—
Dissolved oxygen	CTD probe	0.02 mg DO/L	or Winkller Azide Mod.	—
Salinity	Salinometer	0.1 psu	—	—
Light attenuation	Sensor	0.05%@100%light	e.g., LI-COR-LI-192S A sensor	—
Temperature	CTD probe	0.1°C	—	—
Laboratory analyses				
*Total phosphorus	SM 4500P-E	0.3 µM	Ascorbic acid method	APHA 1998
(including orthophosphate,	SM 4500P-E	0.32 µM	Auto. persulfate method	APHA 1998
POP, and DOP)	EPA 365.2	—	—	EPA 600/4-79-020
	CBP IV.D.2	0.03 µM	Auto. persulfate method	EPA 903-R-96-006
Dissolved orthophosphate	CBP IV.D.3	0.02 µM	Ascorbic acid method	EPA 903-R-96-006
Particulate phosphorus	CBP IV.D.4	0.04 µM	Ascorbic acid method	EPA 903-R-96-006
*Total N, incl. DON,	SM 4500N-C	0.36 µM	Persulfate method	APHA 1998
DIN, and PON[a]	ASTM D3867	0.7-143 µM	Persulfate method	ASTM 1976
	EPA	—	Persulfate method	EPA 903-R-96-006
	EPA-AERP18	—	—	EPA 600/4-87-026
	CBP IV.D.8	1.9 µM	Auto. persulfate method	EPA 903-R-96-006
Total Kjeldahl N	SM 4500org-C with	—	Semi-micro-Kjeldahl method	APHA 1998
	SM 4500NH3-H	1.4-1429 µM	Auto. phenate method	APHA 1998
	EPA 351.3/.1 (mod.)	—	Colorimetric/titration	EPA 600/4-79-020
Ammonia/ammonium	SM 4500NH3-B/H	1.4-1429 µM	Auto. phenate method	APHA 1998
	EPA 350.1	0.7-1429 µM	Colorimetric phenate	EPA 600/4-79-020
	CBP IV.D.7	0.3 µM	Auto. phenate method	EPA 903-R-96-006
Nitrate	SM 4500NO3-F	35.7-714 µM	Auto. cadmium reduction	APHA 1998
	EPA 353.2	—	—	EPA 600/4-79-020
Nitrite	SM 4500NO3-F	35.7-714 µM	Auto. cadmium reduction	APHA 1998
	EPA 353.2	—	—	EPA 600/4-79-020
	SM 4500NO2-B	0.7-71 35.7-714 µM	Colorimetric method	APHA 1998
	EPA 354.1	—	—	EPA 600/4-79-020

Table 4-1. Suggested methods for analyses and monitoring of eutrophic conditions of coastal and marine environments (* = primary EPA preferred causal and response variables) (continued)

Eutrophication indicators	Suggested methods	Detection limit or range	Comments	References
	CBP IV.D.5	0.01 µM	Auto. colorimetric method	EPA 903-R-96-006
Nitrate + nitrite	SM 4500NO3-F	35.7-714 µM	Auto. cadmium reduction	APHA 1998
	EPA 353.2	—	—	EPA 600/4-79-020
	EPA 4.1.4	0.7-143 µM	Technicon autoanalyzer	EPA 503/2-89/001
	CBP IV.D.6	0.01 µM	Auto. colorimetric method	EPA 903-R-96-006
Particulate N	CBP IV.D.8.10	1.36 µM	Filtration/combustion	EPA 903-R-96-006
Total organic carbon	SM 5310TOC-D	>0.1 mg C/L	Wet oxidation method	APHA 1998
	SM 5310TOC-C	>0.01 mg TOC/L	Persulfate method	APHA 1998
	EPA 415.1	—	—	EPA 600/4-79-020
Dissolved organic carbon	SM 5310TOC-C	>0.01 mg TOC/L	Persulfate method	APHA,1998
	EPA 415.1	—	—	EPA 600/4-79-020
	ASTM D2574-79	—	—	ASTM 1976
	CBP IV.D.10	0.5 mg/L	Catalytic combustion	EPA 903-R-96-006
Particulate carbon	CBP.IV.D.9	0.097 mg/L	Filtration/combustion	EPA 903-R-96-006
Total silicates	SM 4500SiO2-D	0.33-0.83 µM	Heteropoly blue method	APHA 1998
	ASTM D859-68	—	—	ASTM 1976
	CBP-IV-15	0.17-23.3 µM		EPA 903-R-96-006
	EPA 370.1	—	—	EPA 600/4-79-020
	CBP IV.D.15	0.22 µM	Molybdosilicate method	EPA 903-R-96-006
Total suspended solids	SM 2540-D	2-20,000 mg/L	Dried at 103-105°C	APHA 1998
	CBC IV.D.13	2.0 mg/L	Filtration/heat	EPA 903-R-96-006
Total volatile solids	SM 2540-E	—	—	APHA 1998
	Estuarine	—	—	EPA 430/9-86-004
BOD	SM 5210-B	—	5-day method	APHA 1998
	EPA 405.1	—	—	EPA 600/4-79-020
	CBP IV.D.11	—	5-day method	EPA 903-R-96-006
COD	SM 5220-D	—	—	APHA 1998
	EPA 410.4	—	—	EPA 600/4-79-020
Biological measures				
Phytoplankton biomass	—	—	—	—
Zooplankton biomass				
Chlorophyll *a*[b]	SM 10200-H	0.01 mg/M^3	Fluorometric, HPLC, Spectro.	APHA 1998

Table 4-1. Suggested methods for analyses and monitoring of eutrophic conditions of coastal and marine environments (* = primary EPA preferred causal and response variables) (continued)

Eutrophication indicators	Suggested methods	Detection limit or range	Comments	References
	EPA AERP12	—	—	EPA 600/4-87-026
	ASTM D3731-79		Spectrophotometer	ASTM 1976
	CBP IV.D.12	1.0 µg/L	Spectrophotometer	EPA 903-R-96-006
Phaeophytin	SM 10200-H	0.01 mg/M^3	Fluorometric, HPLC, Spectro.	APHA 1998
	EPA AERP12	—	—	EPA 600/4-87-026
	ASTM D3731-79	—	Spectrophotometer	ASTM 1976
	CBP IV.D.12	1.0 µg/L	Spectrophotometer	EPA 903-R-96-006
Dinoflagellate density	—	—	—	—
Diatom density	—	—	—	—
Dinoflagellate/diatom	—	—	—	—
Perennial plant density	—	—	—	—
Ephemeral plant density	—	—	—	—
Epiphytic growth	—	—	—	—
Phytoplankton blooms	—	—	—	—
Fish kills	—	—	—	—

[a] DON, dissolved organic N, DIN, dissolved inorganic N; PON, particulate organic N.
[b] Phytoplankton segments: The HPLC procedure is capable of detecting and quantifying various pigments characteristic of different algal groups (e.g., diatoms, cyanophyta, chlorophyta, and dimoflagellates) (Jefferey et al. 1997).

Laboratory Analytical Methods

Detailed methods and references are given in Table 4-1. Some general considerations are presented in the following sections.

Water Column Nutrients

Nitrogen Compounds

Several methods have been used to determine the concentration of N species in the marine environment. Methods presented in this document are relatively easy to use, do not require extensive instrumentation, provide detection limits below those expected in marine environments, and are in general use by many investigators. The most common forms of N in eutrophication evaluation in order of decreasing oxidation state are nitrate, nitrite, ammonia, and organic N. The sum of these is expressed as TN and is not to be confused with total Kjeldahl N (TKN), which is the sum of organic N and ammonia. Total N can be determined through oxidative digestion of all digestible N forms to nitrate, followed by quantitation of the nitrate. Nitrite is an intermediate oxidation state of N, both in the oxidation of ammonia to nitrate and in the reduction of nitrate. Such oxidation and reduction may occur in wastewater treatment plants, water distribution systems, or natural waters. Ammonia is produced largely by deamination of organic N-containing compounds and by hydrolysis of urea. The two major factors that influence selection of the method to determine ammonia are concentration and presence/absence of interferences (e.g., high concentrations of colored organic substances such as humic-like materials or paper mill effluents).

Total N is measured by the persulfate method, which digests all organic and inorganic – containing compounds. All N-containing materials (except nitrogen gas) are measured after sample digestion has occurred. Various organizations have adjusted sample volume or automated the process and produced different ranges of detection. The lowest detectable concentration is ~ 0.7 µM of TN. This is in the range of the measured available N (0.7 to 5.0 µM TN) for studies performed off the continental shelf in the North Atlantic from 1956 to 1958 (Kennish 1989). Kjeldahl N minus the ammonia concentration is the surrogate measurement for all organic N-containing compounds.

Ammonia/ammonium is measured by the indophenol blue (= phenate) or specific ion electrode methods after conversion of ammonia and ammonium to ammonia. This is done by raising the pH of the sample above 11. This method has some essential features (e.g., minimal interference from waters highly stained with humic materials and paper mill effluents); however, the level of detection is relatively high (e.g., 2.0 µM NH_3-N) but adequate for ammonia-rich waters (Flemer et al. 1998). Ammonia electrodes do not work directly in seawater. In the spectrophotometric methods, the ammonia is reduced to monochloramine and then reacted with phenol to form a blue color. In the specific ion electrode method, the ammonium is converted to ammonia using a strong basic solution and partial pressure of ammonia gas (i.e., free ammonia) in solution, which is related to the dissolved ammonia concentrations by Henry's Law.

Nitrates and nitrites are measured in combination using the cadmium reduction procedure of Wood et al. (1967). This colorimetric method determines the concentration of these two materials after reaction of nitrites to produce an azo dye, the color of which is proportional to the concentration of the combined nitrates and nitrites. Total nitrate is determined by subtracting the concentration of nitrite from the combination of the two. The process for measurement of nitrite produces the same azo dye as the combined measure, but without the Cd reduction. The difference in these two measures is the nitrate concentration.

Phosphorus

The target detection limit for measurement of P in seawater is ~ 0.3 µM. The procedures for the measurement of total particulate and dissolved P as well as orthophosphate in seawater provide detection limits that are less than this value (U.S. EPA 1996). These procedures convert the phosphorus-containing compounds to orthophosphate through the digestion of the sample with alkaline persulfate. This treatment is then reacted with ammonium molybdate and antimony potassium tartrate in acidic solution to produce an intense blue complex with ascorbic acid. Interferences with elevated concentrations of Si can be avoided by maintaining an acid concentration in the reagents and analyzing the material at elevated temperatures of ~37°C. The resulting phosphomolybdic acid reduction produces a purple-blue complex that is measured at 885 nm on a spectrophotometer. This method of measuring reactive silicate is recommended in Millero (1996).

Silica

The target detection limit for measurement of Si in seawater is ~0.7 µM. Producing pigmented silicomolybdate complex by procedures contained in U.S. EPA (1996) provides adequate sensitivity after the samples are filtered (0.45 µm GF/F filter) to remove interfering particles and turbidity, and after the interferences of phosphates and arsenates are removed with oxalic acid. The resultant filtrate is treated with a solution containing metol-sulfate (p-methyl-amino-phenol sulfate) to produce a blue color that is evaluated more efficiently than the yellow color recommended for evaluation in U.S. EPA (1996), with a spectrophotometer at 812 nm (Strickland and Parsons 1968). This method of measuring reactive silicate is also recommended in Millero (1996).

Carbon

Total carbon consists of inorganic and organic forms that are in particulate and dissolved size classes. The distinction between total and organic carbon is based on acidifying samples to remove the inorganic forms and filtering through 0.45 µm GF/F filters to remove the particulate forms. Total carbon is measured by burning the sample to release the particles contained on the glass fiber filter. This converts the carbon to CO_2, which is then transported to a thermal conductivity detector for measurement. The carbon left behind in the filtrate is catalytically combusted using a platinum catalyst at ~680°C that is then transported to a nondispersive infrared detector. The EPA methods (U.S. EPA 1996) will provide adequate detection of both dissolved and particulate carbon in the total and organic phases. The difference in total carbon and organic carbon represents the inorganic fractions that are primarily $CaCO_3$ shells.

Sediment Analyses

Bulk Sediment

Cores are collected from field sites to help determine the historical record and sedimentation rate. Short cores, the upper 30 cm of the substrate, can be obtained with a HAPPS core, designed to collect a relatively undisturbed core of surficial sediment (Kannerworff and Nicolaisen 1973) and used to profile sedimentary particulate organic carbon and N. Carbon-N analyses follow the method of Hedges and Stern (1984); samples for dissolved constituents in pore water are extracted either by whole-core squeezing or by centrifugation (Devol et al. 1997, Brandes and Devol 1995, Lambourn et al. 1991). Deep coring devices are used to collect continuous sediment core samples 2 to 3 m into the sediment bed. These deeper cores are used for analysis of ^{210}Pb, carbon and N, sulfide, and biogenic silica in order to determine burial rates of ^{210}Pb and ^{210}Ra.

The sedimentation rate is estimated based on the change in activity of naturally occurring ^{210}Pb radionuclide produced at a constant rate from the decay of ^{210}Ra, using the excess ^{210}Pb inventory method of Anderson et al. (1987). Excess ^{210}Pb is determined from the difference between total ^{210}Pb activity in the sediment and the activity of the background ^{210}Pb being produced from ^{210}Ra. To collect samples for measurement of ^{210}Pb and ^{210}Ra activity at depth with the sediment, cores are sectioned and each section is then homogenized and placed in a precleaned 16 oz jar, with a small subsample removed and placed into a glass vial for particulate C and N analysis (Evans-Hamilton, Inc. 1998).

The excess ^{210}Pb inventory method yields accumulation rates (g/(cm^2/yr)), which are converted to a sedimentation rate (cm/yr) using the bulk sediment density g/cm^3. For evaluation of seasonal trends, the upper cm is subsampled at 0.25 cm intervals, and in 1 cm intervals below the first cm, following the assumption that any seasonal storage of N or carbon would manifest almost entirely at the surface of the sediment.

Pore Water Profiles

Pore water profiles of manganese, iron, nitrate, and oxygen demonstrate that oxidation of iron and magnesium yields less energy than does oxidation of carbon by oxygen or nitrate. Consequently, concentration peaks of these species are located below the depletion depths of oxygen and nitrate. In anaerobic environments, after the supplies of oxygen, nitrate, manganese, and iron are exhausted, sulfate reduction is the dominant mode of organic matter oxidation and nutrient remineralization.

Sulfate reduction rate can be measured with the radiotracer method of Christensen et al. (1987). A significant fraction of the oxygen flux may be consumed by the reoxidation of sulfide produced during sulfate reduction (Canfield 1993).

Sediment traps are used to measure the quantity and composition of the flux of materials settling through the water column to the sediment. There are four materials of interest: chlorophyll as an indicator of planktonic algal remains, pheaopigments as an indicator of degraded plankton that has been consumed by zooplankton, particulate organic carbon (POC), and particulate organic N (PON). Total sedimentation rate is corrected for resuspension materials in order to derive the net flux to sediment. Samples are collected by in situ benthic flux chambers, and measurements of oxygen, silicate, nitrate, ammonium, phosphate, and N gas are made (Evans-Hamilton, Inc. 1998).

Determination of Primary Productivity

Primary productivity refers to the growth rate of the phytoplankton community and is commonly measured using trace amounts of radioactive carbon (as bicarbonate) that label the photosynthetic reaction. Additional variables are measured to support these data: biomass (as estimated by chlorophyll *a*), incoming solar radiation, and nutrient concentrations at depth. Primary productivity, P, is defined as

$$P = \mu \times B$$

where μ is the specific growth rate (growth normalized per cell) and B is the biomass of the phytoplankton population (amount of cells). These variables are > 'compound' = as they in turn depend on other variables. Growth rate depends on light (solar radiation), dissolved nutrients in the water column, and water temperature. The phytoplankton biomass is determined by the net result of growth and loss (grazing, mixing, sinking) processes and reflects enrichment conditions.

To estimate primary productivity, samples are collected at varying depths corresponding to predetermined light levels. Fresh samples at each light level are collected for analysis of chlorophyll *a*, nutrients, and primary productivity in two sets of two clear bottles and one dark bottle; each set is filled for ambient treatment and nutrient spike treatment. Nutrient spiking consists of adding an initial concentration of 10 μM N (NH_4CL) and 1 μM phosphorus (KH_2PO_4) to seawater. Nutrients are monitored from additional samples collected and tested for nitrate, nitrite, ammonium, orthophosphate, and silicate. Samples are inoculated with ^{14}C-labeled sodium bicarbonate and, if appropriate, the nutrient spike, and placed in a screened bag to simulate the light level from which they were collected. Samples are incubated at in situ conditions for 24 hours and then transported to the laboratory for filtration using glass fiber filter paper (Whatman GF/F, nominal pore size 0.7 μm or smaller pore size). The filters are placed into vials containing EcoLume scintillation cocktail. The specific activity of the filtered particulates is measured in a scintillation counter. Primary production is calculated as mg C/(m^3/day) using the basic equations found in Parsons et al. (1984) (Evans-Hamilton, Inc. 1998).

In productive coastal waters, measurements using the light and dark bottle technique with changes in dissolved oxygen often can be used in place of the ^{14}C method (Strickland and Parsons 1968). In some cases, free water gas-based (e.g., DO) methods are possible to measure ecosystem metabolism (Odum 1956; Odum et al. 1959; Kemp and Boynton 1980).

Phytoplankton Species Composition

Samples collected from the field are analyzed to identify and enumerate autotrophic phytoplankton, as well as heterotrophic dinoflagellates and microzooplankton species. From 20 to 50 mL aliquots of samples are settled in separable counting chambers for at least 24 hours before examination under phase-contrast optics with an inverted microscope following the classic Utermöhl technique (Lund et al. 1958). A single transect across the center of the chamber is counted at 390× magnification for flagellates; 150× magnification is used for other organisms. From 25% to 100% of the chamber bottom is examined, depending on cell concentrations in the sample. Appropriate multipliers are used to convert all counts to common units of cells/L (Sournia 1978). Organisms are identified to the lowest taxonomic category possible. Even quite small changes in the physical and chemical parameters and availability of micronutrients can have a significant effect on the growth constants of algae. A difference in doubling time of 25% between two fast-growing organisms can lead to one outnumbering the other by 15 to 1 in a week and quickly lead to alterations in species assemblages (Riley and Chester 1971).

There are numerous algal species in estuarine and open coastal waters that are considered to be harmful (e.g., see Dortch et al. 1998, Anderson and Garrison 1997, Anderson 2000). This is a rapidly changing area of marine ecology and experts should be consulted for specific taxonomic identifications.

Macrobenthos, Macroalgae, and Seagrasses and SAV

Macroinfauna are typically sampled with coring devices or bottom grab samplers and wet-sieved through 0.5 µM mesh sieves to separate the animals from very fine sediments. Stacked sieves can be used to remove larger shell fragments and sand particles. A relaxant (e.g., 0.3% propylene phenoxytol) is applied prior to addition of formalin. Samples are usually preserved in 10% buffered formalin for several weeks and then transferred to 60%-70% isopropanol (Diaz and Rosenberg 1995).

Macroalgae are typically sampled by collecting algal material by hand from a known surface area of the habitat. Various devices may be used (e.g., 0.5 m stainless or plastic hoop).

Both above- and below-ground seagrass and SAV biomass can be collected from a known area of the bed. Various techniques have been used. An often-used method is to shove metal strips along the sediment surface in a square meter pattern and anchor the strips at all four corners by pushing a sharp spike through holes drilled at each end of the strips. Then, the plant material separated to species can be clear-cut with sharp shears and taken to the laboratory and dried in a heated cabinet at 60°C to constant dry weight. A sharp spade is required to collect below-ground roots and rhizomes. This material should be identified and dried to constant weight.

CHAPTER 5

Databases, Sampling Design, and Data Analysis

Developing National/Regional Databases
Sampling Design
Quality Assurance/Quality Control
Statistical Analyses

5.1 INTRODUCTION

Development of national regional numeric nutrient criteria requires that an extensive amount of data from across the country be evaluated. This information can be an invaluable tool to States and Tribes as they develop nutrient criteria. Both existing and historical data may provide considerable information that is specific to the region where criteria are to be set. First the data must be located, then the suitability of the data (type and quality) ascertained before they can be used for analysis of water quality parameters. It is also important to determine how the data were collected to make future monitoring efforts compatible with earlier approaches. Descriptive data that characterize the waterbody are invaluable.

Data may come from existing sources or can be collected from new sampling programs. Nutrient-related data for estuaries and coastal waters, collected by various agencies for many different purposes, exist in numerous databases and have the potential to provide the basis for development of nutrient criteria on a regional level. This chapter presents an overview of existing databases and a general discussion concerning the evaluation of such datasets in terms of their use in the nutrient criteria development process. The list of databases is not all-inclusive—many other data sources exist—but the list provided is intended to represent the kind of information that is available. This chapter also provides a description of existing data resources (e.g., U.S. EPA Legacy STORET and ODES) and how these data may be used to generate preliminary nutrient criteria on regional levels. In addition to discussing the use of existing data, the chapter discusses new data collection, including consideration for sampling design and the types of sampling to be considered as part of data collection activities. The chapter ends with a general discussion of data management, quality assurance, and quality control issues that are integral in the overall discussion of data storage, accessibility, and utilization.

5.2 DEVELOPING REGIONAL AND NATIONAL DATABASES FOR ESTUARIES AND COASTAL WATERS

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

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

Geographic information systems (GIS) are geo-referenced databases that have a geographic component (i.e., spatial platform) in the user interface. Spatial platforms associated with a database allow geographic display of sets of sorted data and make mapping easier. These types of databases with spatial platforms are becoming more common. The system is based on the premises that "a picture is worth a thousand words" and that most data can be related to a map or other easily understood graphic. GIS platforms such as ArcView, ArcInfo, and MapInfo are frequently used to integrate spatial data with monitoring data for watershed analysis.

The EPA National Nutrient Criteria Program initiated the development of a national database application that will be used to store and analyze nutrient data. The ultimate use of these data will be to derive ecoregion- and waterbody-type specific numeric nutrient criteria. Initially, EPA developed a Microsoft Access application that was populated with STORET Legacy data, U.S. Geological Survey (USGS) (NAWQA, NASQAN, and Benchmark) data, and other relevant nutrient data from universities, States/Tribes, and additional data-rich entities. To serve the general public more effectively and efficiently, EPA also developed and maintains a web-accessible nutrient database application in an Oracle™ environment that allows for easy web accessibility, geo-referencing/GIS compatibility, and data analysis on a State/Tribal, regional, and national basis. The total amount of existing nutrient data nationally is large (>20 gigabytes), and it is anticipated that more data will be entered into the system. The Oracle™ application can easily manage large quantities of data and provides ample room for expansion as more data are collected. The Oracle™ database application is being designed for compatibility with EPA's modernized STORET. A key feature of the database design will prevent duplication of effort for users of STORET and the nutrients database application, especially for data updating. Considerable efforts are also being made to ensure compatibility with other database systems (e.g., WQS and RAD) currently being developed by EPA's Office of Water. The Oracle™ application has been online since the fall of 2000.

Data Sources

Potential sources of data include water quality monitoring data from Federal, State, Tribal, and local water quality agencies; university studies; and volunteer monitoring programs. However, the data sources described in this section do not encompass the full extent of available data sources. The data available in the nutrient database can be used to identify reference areas to begin development of potential nutrient criteria. The nutrient data sources for estuaries and coastal waters that will be useful for developing criteria are discussed below. These data sources contain extensive water quality data, however, data; collection should not be limited to these sources. Collection of scientifically sound water quality data from any reliable source is encouraged.

Many of the water quality programs listed here include rivers and streams data or mixed freshwater, estuarine, and coastal water systems data. The rivers and streams information is included in this document because it gives relevant data about nutrient loading from fluvial systems, which is important to estuaries and coastal waters. Generally, in estuaries that have been impaired by nutrients, a database exists, and in less impaired estuaries, the database is often insufficient for comparisons. Nutrient loading information from fluvial systems may provide a basis for comparison between systems if they share important geophysical conditions. Such comparisons would assist in developing trends and extrapolating where insufficient data exist.

EPA Water Quality Data

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

STOrage and RETrieval System (STORET)

STORET is EPA's national database for water quality and biological data. EPA's original STORET system, called the STORET Legacy Data Center (LDC) and operated continuously since the 1960s, was historically the largest repository of water quality data in the Nation. This legacy mainframe-based system was the repository of all data held in EPA's original STORET system as of the end of 1998. This Legacy STORET ceased to exist in the year 2000. In its place, EPA is supporting a modernized database, simply called STORET, designed as a replacement for the original STORET System. While STORET will serve as the major repository for more current data, the nutrient criteria database application will offer major improvements in database content and capabilities that will enable more detailed data analysis.

Interested parties may view both databases on the World Wide Web. For the nutrient database, capabilities exist to produce printed reports and download data files. Queries for data via the web will be designed for use by the general public and will require no special training or software.

STORET is a compendium of data supplied by Federal, State, and local organizations used to evaluate environmental conditions in the field. The data in STORET are organized by both geographic location and data ownership. Every field study site is identified by at least one latitude/longitude and, where appropriate, also by State/Province, county, drainage basin, and stream reach. Monitoring activities recorded include field measurements, habitat assessments, water and sediment samples, and biological population surveys. Records cover the complete spectrum of physical properties, concentrations of substances, and abundance and distribution of species observed during biological monitoring. STORET is designed for maximum compatibility with commercial software, including GISs such as the ESRI ArcView package, and statistical packages such as PC SAS. STORET downloaded files import easily into all standard spreadsheet packages. Further information about STORET may be obtained by e-mailing STORET@epa.gov, or telephoning toll-free at 1-800-424-9067.

Environmental Monitoring and Assessment Program (EMAP)

EMAP is an EPA research program designed to develop the tools necessary to monitor and assess the status and trends of national ecological resources (see EMAP Research Strategy on the EMAP website: www.epa.gov/emap). EMAP's goal is to develop the scientific understanding for translating environmental monitoring data from multiple spatial and temporal scales into assessments of ecological condition and forecasts of future risks to the sustainability of the Nation's natural resources. EMAP's research supports the National Environmental Monitoring Initiative of the Committee on Environment and Natural Resources (CENR). Data from EMAP can be downloaded directly from the EMAP website. The EMAP Data Directory contains information on available datasets, including data and metadata (language that describes the nature and content of data). The status of the Data Directory as well as composite data and metadata files also are available on the EMAP website. EMAP–estuaries data is one of several areas addressed by the program. Most of the estuaries data were collected during a summer index period.

Ecological Data Application System (EDAS)

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

Ocean Data Evaluation System (ODES)

ODES is used for storing and analyzing water quality and biological data from marine, estuarine, and some freshwater environments. The system supports Federal, State, and local decisionmakers associated with marine monitoring programs and managers and analysts who must meet regulatory objectives through the evaluation of marine monitoring information. ODES contains data from the National Estuary Program, the Great Lakes National Program Office, the Ocean Disposal Program, the 301(h) Sewage Discharge Program, the National Pollutant Discharge Elimination System Program, and the 403(c) Program. Records pertain to water quality, fish abundance, bioaccumulation, benthic infauna, fish histopathology, bioassay, and sediment physical/chemical characteristics. Users can examine both spatial and temporal relationships among variables. A quality assurance report describing analytical methods and procedures for each dataset is stored with each dataset.

Chesapeake Bay Program (CBP)

CBP, a cooperative effort between the Federal Government, the States, the District of Columbia, and local governments in the Chesapeake Bay watershed, provides funds to the States of Maryland and Virginia for the routine monitoring of 19 directly measured water quality parameters at 49 stations within the bay watershed. The Water Quality Monitoring Program began in June 1984 with stations sampled once each month during the colder late fall and winter months and twice each month in the warmer months. A refinement in 1995 reduced the number of monitoring cruises to 14 per year. Data are available on the internet at www.chesapeakebay.net/data/.

National Estuarine Programs (NEPs)

Many NEPs have nutrient and related data that could be used for characterization purposes. Presently, there is no national repository of NEP data, but, in the development of regional nutrient criteria, the NEPs may serve as an excellent source for information. Some of these programs have electronic databases and some hard copy data that could be acquired. EPA is attempting to acquire the available NEP data and eventually enter them into the National Nutrient Criteria Program database. A list of NEP estuarine systems can be found online at www.epa.gov/nep.

National Oceanographic and Atmospheric Administration (NOAA)

Water Quality Data in the National Oceanographic Data Center (NODC)

NODC is one of three national environmental data centers operated by NOAA and serves as a national repository and dissemination facility for global oceanographic data. Its primary mission is to ensure that global oceanographic data collected at great cost are maintained in a permanent archive easily accessible to the world science community and to other users. NODC holds physical, chemical, and biological oceanographic data collected by U.S. Federal agencies, including the Department of Defense (primarily the U.S. Navy); State and local government agencies; universities and research institutions; and private industry. NODC does not conduct any data collection programs of its own; it serves solely as a repository and dissemination facility for data collected by others (see website at www.nodc.noaa.gov). NODC provides data management support for major ocean science projects such as Tropical Ocean–Global Atmosphere (TOGA), World Ocean Circulation Experiment (WOCE), and Joint Global Ocean Flux Study (JGOFS). NODC's global holdings of physical, chemical, and biological oceanographic data include substantial amounts of data from coastal ocean areas. For example, the NODC Oceanographic Profile Database holds primarily coastal data (www.nodc.noaa.gov/cgi-bin/JOPI/jopi).

National Estuarine Research Reserve System (NERR)

The NERR Systemwide Monitoring Program was designed to identify and track short-term variability and long-term changes in the integrity and biodiversity of representative estuarine ecosystems and coastal watersheds for the purposes of contributing to effective national, regional, and site-specific coastal zone management. The program has two major goals: (1) to support State-specific nonpoint source pollution control programs by establishing local networks of continuous water quality monitoring stations in representative protected estuarine ecosystems; and (2) to develop a nationwide database on baseline environmental conditions in the NERR system of estuaries. Water quality data collected from phase 1 of the NERR Systemwide Monitoring Program provides data necessary for site and intersite baseline studies, trend analysis, and impact assessment. Data are available for each of the participating NERR systems at http://inlet.geol.sc.edu/cbmoweb/30_minute_data.html.

Rivers and Streams Water Quality Data

Rivers and streams water quality data are potentially useful for estuaries and coastal waters. Because much of the nutrient load to estuaries comes from rivers and streams, it is critical to define nutrient concentrations landward of tidal influence and to calculate fluvial-based nutrient loads to estuaries and potentially to coastal waters. EPA STORET, which was discussed in detail previously, includes data from rivers and streams from across the Nation. Another comprehensive Federal source of river and stream water quality data is USGS. USGS maintains databases on water quantity and quality for waterbodies across the Nation. Many of the data for rivers and streams are available through the National Water Information System (NWIS). The most convenient method of accessing the local databases is through the USGS State representative. Every State office can be reached through the USGS home page on the Internet at URL http://www.usgs.gov/wrd002.html. The USGS data from several national water quality programs covering large regions offer highly controlled and consistently collected data that may be particularly useful for nutrient criteria analysis. Two programs, the Hydrologic Benchmark Network (HBN) and the National Stream Quality Accounting Network (NASQAN), include routine monitoring of rivers and streams during the past 30 years. The USGS National Water Quality Assessment (NAWQA) Program is building a third national database of stream quality data collected and analyzed for more than 50 river basins and aquifer systems across the Nation. More information and data from each of these studies can be found on the USGS website (www.usgs.gov). For additional data sources, the Rivers and Streams Nutrient Criteria document presents an extensive list of related freshwater nutrient-related information (Nutrient Criteria Technical Guidance Manual—Rivers and Streams, 2000, EPA-822-B-00-002).

USGS San Francisco Bay Program

Since 1968, USGS has sustained a research program to understand how coastal ecosystems function and how those functions are altered by human disturbances. One component of this program is directed to following and understanding changes in the water quality of San Francisco Bay. The program includes regular measurements of water quality along a 145-kilometer transect spanning the length of the entire estuarine system, from the South Bay to the Sacramento River. The program studies many different aspects of San Francisco Bay, such as changing land use, hydrology, water currents, nutrients, toxic contaminants, geological structure, and biological communities. The results of water quality measurements, and eventually the full dataset, can be accessed at http://sfbay.wr.usgs.gov/access/index/wqdata.html.

State/Tribal Monitoring Programs

Most States monitor some estuaries and coastal waters within their borders for algal and nutrient variables. Data collected by State/Tribal water quality monitoring programs can be used for nutrient criteria development. These data should be available from the agencies responsible for monitoring.

Sanitation Districts

Massachusetts Water Resources Authority (MWRA)

MWRA has conducted a comprehensive monitoring program in Boston Harbor, Massachusetts Bay, and Cape Cod Bay from 1992 to the present. The program was established to understand baseline conditions and monitor the effects of effluent discharges into Boston Harbor and Massachusetts Bay. This multifaceted monitoring program focuses on water quality, benthic ecosystem health, effluent characterization, and public health issues related to metal, organic, and microbiological contaminants. All data are stored in an Oracle™ Relational Database Management System to support the monitoring program.

New York City–Department of Environmental Protection (NYCDEP)

NYCDEP has conducted extensive monitoring to evaluate viable treatment options for sewage effluents to mitigate conditions that promote eutrophication. The city’s monitoring programs have included point source, water column, sediment, hydrodynamic and atmospheric studies. All data are stored in the NYCDEP databases and have been used in the development and application of a System-Wide Eutrophication Model (SWEM) to enhance the city's ability to evaluate the effectiveness of various treatment options in mitigating conditions that promote eutrophication.

Southern California

The major southern California dischargers of treated sewage effluents into marine waters have conducted applied research and monitoring programs for more than 30 years. These dischargers include the cities of San Diego and Los Angeles and the counties of Ventura/Oxnard, Los Angeles, Orange, and San Diego. The programs are designed to monitor the concentrations and mass emission rates of effluent materials in the treated effluent; the transport and fate of these materials in the receiving waters; the exposure of the contaminants to organisms in the receiving waters; and the effects of that exposure to individuals, populations, and communities of subtidal, intertidal, and water column organisms. Some of this monitoring is performed to comply with NPDES monitoring efforts, and other monitoring addresses specific issues of interest to the districts. The data are retained on a number of local databases, but they are also maintained on EPA’s ODES. In addition to the localized databases managed by the sanitation districts, a research organization (Southern California Coastal Water Research Project) has performed parallel and specialized monitoring and applied research on the effects of treated sewage effluents in this region since the early 1970s. Their data are managed onsite and are provided to national data inventories (e.g., ODES, NODC, STORET).

California and Oregon
Similar monitoring requirements are established for other locations in California and Oregon. The districts that have effluents that are discharged into rivers, streams, or oceans are required, through the NPDES permits, to monitor their treated effluents and the receiving waters. The data are retained locally, but must also be filed with ODES. The State and Regional Water Resources Control Boards of California and the Departments of Ecology in Washington and Oregon administer these permits.

Puget Sound, Washington
The cities and counties on the Puget Sound watershed monitor the treated sewage effluents and the receiving waters in compliance with State water quality parameters. These data are provided to the State in electronic format and are retained on database systems administered by the Department of Ecology that are available to ODES. The data in the receiving water environments are collected in methods that are historically similar to work that has been performed in Puget Sound since the middle 1960s.

Academic and Literature Sources

Many research studies conducted by academic institutions may provide data useful for developing nutrient criteria. Academic research tends to be site specific and span a limited number of years, although data for some systems may span 20 years or more. Academic research data should be available from researchers. However, the scientific literature is likely to be a major source of estuarine and coastal waters data.

Volunteer Monitoring Programs

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

State and local agencies may use volunteer data to screen for water quality problems, establish trends in waters that would otherwise be unmonitored, and make planning decisions. Volunteers benefit from learning more about their local water resources and identifying what conditions or activities might contribute to pollution problems. As a result, volunteers frequently work with clubs, environmental groups, and State/Tribal or local governments to gather information and address problem areas. As with any other data source, whether student, State, Federal, academic, or volunteer based, documented quality assurance procedures are an important consideration.

EPA supports volunteer monitoring and local involvement in protecting our water resources. EPA support takes many forms, including sponsoring national and regional conferences to encourage information exchange among volunteer groups, government agencies, businesses, and educators; publishing sampling methods manuals for volunteers; and providing technical assistance (primarily on quality control and laboratory methods) and regional coordination through the 10 EPA Regional Offices. EPA also produces a Nationwide Directory of Volunteer Monitoring Programs, which is available online at http://yosemite.epa.gov/water/volmon.nsf. This directory lists volunteer organizations around the country engaged in monitoring rivers, lakes, estuaries, beaches, wetlands, and groundwater, as well as surrounding lands. EPA volunteer monitoring activities are coordinated in part through a website that lists many resources at http://www.epa.gov/owow/monitoring/volunteer.

Quality of Historical Data

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

Location Data

STORET and USGS data are geo-referenced with latitude, longitude, and up to Reach File 3 (RF1 & RF3) codes. Geo-reference data can be used to select specific locations or specific USGS hydrologic units. In addition, STORET often contains a site description. Knowledge of the rationale and methods of site selection from the original investigators may supply valuable information. Metadata of this type, when known, are frequently stored within large long-term databases.

Variables and Analytical Methods

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

Laboratory Quality Control

Laboratory quality control of data (blanks, spikes, replicates, known standards, etc.) where available should be reported. Such information may have been infrequently reported in larger data repositories and needs to be identified and coded. Records of general laboratory quality control protocols and specific quality control procedures associated with specific datasets are valuable in evaluating data quality. However, premature elimination of lower quality data can be counterproductive because the increase in variance caused by analytical laboratory error may be negligible compared with natural variability or sampling error, especially for nutrients and related water quality parameters. However, data of uncertain quality should not be accepted unless no other data are available.

Data Collecting Agencies

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

Time Period

Long-term records are critically important for establishing trends. Determining if trends exist in the time series database is also important for characterizing reference conditions for nutrient criteria. Length of time series data needed for analyzing nutrient data trends is discussed in the Sampling Design section (Section 5.3).

Index Period

The index period for estimating average concentrations can be established if nutrient and water quality variables were measured through seasonal cycles. The index period may be the entire year or the summer season. The best index period is determined by considering water quality characteristics for the region, the quality and quantity of data available, and estimates of temporal variability (if available). Additional information and considerations for establishing an index period are discussed in Section 5.3.

Representativeness

Data may have been collected for specific purposes. Data collected for toxicity analyses, effluent limit determinations, or other pollution problems may not be useful for developing nutrient criteria. Furthermore, data collected for specific purposes may not be representative of the spatial scale of interest. The investigator must determine if the spatial scale for the data included in the database is representative of the area to be characterized. If a sufficient amount of data for the appropriate scale cannot be found, then new surveys will be necessary (see Section 5.3).

Gathering New Data

New data should be gathered following the sampling design protocols discussed below. New data collection activities for developing nutrient criteria should focus on filling in gaps where data are particularly needed for high-risk systems. Data gathered under new monitoring programs should be imported into databases or spreadsheets and merged with the existing nutrient database for criteria development.

5.3 SAMPLING DESIGN

This section discusses issues surrounding sampling nutrients, response variables, and related environmental variables in estuaries and coastal waters. Where appropriate data are unavailable or insufficient to derive numerical nutrient criteria, efforts must be made to collect new data to fill those gaps. New sampling programs should be scientifically based and statistically rigorous while maximizing available management resources. Such programs are used to better define nutrient and algal relationships within an ecosystem framework. At the broadest level, sampling efforts should detect or contribute to the following objectives:

- Identify the reference condition, that is, existing, most natural, least culturally impacted locations and their relative enrichment status

- Identify whether nutrient concentrations or loads are increasing, decreasing, or remaining the same

- Characterize seasonal patterns in nutrient levels and their relationship to primary productivity

- Help assess the assimilative capacity of the system, that is, contribute to the determination of how much nutrient loading can be assimilated without causing unacceptable changes in water quality or algal biomass and composition. (Note: In estuaries and coastal waters, this objective will likely require application of a computer-based hydrodynamic and nutrient-coupled water quality model. The intention here is to recognize particularly susceptible waters, not to lower expectations relative to historical antecedents and the reference condition.)

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

Sampling Protocol

Success of nutrient criteria development requires that consideration be given to sampling design. Initially, the relationship between critical response variables and nutrient concentrations, or in some cases, nutrient loads, needs to be established. Next, reference sites should be sampled, if feasible, in an attempt to establish reference conditions within classes of systems or subsystems. Classification should be linked to the reference condition activity. Nutrient concentrations/load and algal biomass relationships should help define the ecological state that can be attained if impaired systems are restored. As discussed in the following sections, this is not a straightforward exercise; it is very difficult to predict water transport/mixing in estuaries and coastal waters. The physics of these waters plays a major role in determining the observed patterns in nutrient and chlorophyll *a* concentrations, turbidity, and bottom water dissolved oxygen deficiency as well as transport. Variability in time and space further complicates empirical analysis as pointed out in Chapter 2.

Nutrient concentrations, chlorophyll *a* production, and system respiration represented as biological oxygen demand (BOD) are biochemical processes. How these processes are expressed in terms of dissolved oxygen, especially in bottom waters, has a great deal to do with the variability in freshwater flows, density stratification, advection, and mixing. The forcing factors include wind setups, changes in barometric pressure gradients, freshwater gravitational circulation, and the added complexity of bottom bathymetry. Interactions of these factors may create "flow jets," flow reversals, three-layered circulation, and other physical complexities that suggest any monitoring scheme planned for estuaries and coastal waters would be advised to have a physical oceanographer as part of the team.

Sampling Technique

A reasonable and representative method is to profile the general physical character of the site by a CTD hydrocast. Water samples then may be collected from the surface 1 meter, mid-depth, and bottom 1 meter of the water column. Sample station activities should be coordinated as much as possible with the same tidal and current phase each time data are collected. If turbidity is measured by Secchi depth, the disc should be lowered from the shaded side of the vessel and depth determined from as close to the water level as practicable. Secchi depth measurements should be made only during periods of full daylight. The data for each station and sampling event should be recorded for each depth interval. This permits assessment for surface as well as bottom conditions. Where satisfactory, the results for each sampling episode can be combined into a mean or median measure representing all depths at that site. Temporal and spatial medians of the sites then can be determined to establish the representative values for that reference site.

During sampling visits, the candidate reference stations also should be examined to confirm whether they actually meet the reference site requirements. This may include looking for nearby discharges into the waters or tributaries and a quick survey of the shoreline to determine if new modifications may have changed the site. If an area appears to have been significantly impacted, measurements should be made for nutrient concentrations and biological response variables such as chlorophyll concentration and fish, macroinvertebrate, macrophyte, and planktonic community variables. Sites that do not meet the physical reference requirements should be excluded from the reference dataset. However, a high nutrient concentration present in an otherwise minimally developed area is not justification alone for exclusion. This may be part of the natural background level to be identified by the reference condition process.

Initial Considerations

Variability is inherent in sampling, which means that accuracy (how well the measure reflects actual conditions) and precision (how consistent the measurement is) must be assessed. Precision in ecological samples and measurements is more easily characterized than accuracy. Replicate samples from an experimental unit provide the basis for precision analysis. Standard statistical textbooks focus on precision (i.e., various ways to assess the nature of variability) and especially inferences regarding null hypotheses. In analytical chemical analyses, accuracy can be assessed by including samples of known purity and/or amount. Accuracy often refers to systematic errors in a method, whereas precision refers more or less to random errors. Outliers can be detected with statistical methods, but the so-called outlier may actually prove to be more accurate than the remainder of the data.

Some key questions to consider are the following: Does the method reflect more on how the assessor or analyst uses the method if the technician is imprecise, and Does the method actually do what it is purported to do? For example, is the arsenic concentration high enough that it interferes importantly with the phosphate analysis? Does the Loran system accurately place one on station, especially when electronic interferences are highly probable? Does the temperature setting on the dissolved organic carbon analyzer allow for accurate measurement or a biased one, or does the chlorophyll *a* method accurately correct for interferences from other pigments? For these examples, the precision may be very high (low variance) but the measurement may be in substantial error so the results are inaccurate. The following discussion expands on these ideas and provides additional information on statistical concepts and procedures relative to sample design. It is worth remembering that environmental data may not conform to the assumptions of normality required for statistical inferences; adjustments often can satisfy parametric assumption, but when they do not, the analysts must resort to distribution-free methods.

Specifying the Population and Sample Unit

Sampling is statistically expressed as a sample from a population of objects. Finite populations may be sampled with corresponding natural sample units, but often the sample unit (say, an estuary) is too large to measure in its entirety, and it must be characterized with one or more second-stage samples of the sampling gear (bottles, benthic grabs, quadrats, etc.). Each sample unit is assumed to be independent of other sample units. The objective of sampling is to best characterize individual sample units in order to estimate some attributes (e.g., nutrient concentrations or dissolved oxygen) and their statistical parameters (e.g., mean, median, variance and percentiles) of a population of sample units. The objective of the analysis is to be able to say something (estimate) about the population. It is critical to distinguish between making an inference about a population of many estuaries (e.g., “lagoonal estuaries around northern Gulf of Mexico are shallow and mesotrophic”) versus an inference about a single estuary or coastal water (e.g., “estuary ABC has fewer fish species than unimpaired reference estuaries or salinity zones within estuaries”). These two kinds of inferences require different sampling designs: the first requires independent observations of many waterbodies and does not require repeated observations within sample units (pseudoreplication; Hurlbert 1984), while the second often does require repeated observations within a waterbody. Examples of sample units include:

- A point in an estuary or coastal water (may be characterized by single or multiple sample device deployments). The population then would be all points in the waterbody, an infinite population.

- A constant area (e.g., square meter, hectare). The population could be all square meters of a coastal water surface area in a State or region.

- An estuary or a definable subbasin or salinity zone of an estuary as a single unit. Because salinity often specifies population distributions in estuaries, these zones most often are discrete environments, at least in the short term, and this is likely to be the most common sample unit. The population would be all salinity zones in a State or region, a finite population.

Specifying the Reporting Unit

It is also necessary to specify the units for which results will be reported. Usually, this is the population (e.g., all estuaries), but it also can be subpopulations (e.g., estuaries within a given nutrient ecoregion) and even individual locations (e.g., estuaries or coastal waters of special interest). To help develop the sampling plan, it is useful to create hypothetical statements of results in the way that they will be reported, for example:

- Status of a place: "The estuary ABC is degraded."

- Status of a region: "An estimated 20% of the estuary area in State XYZ has an elevated trophic state, above reference expectations"; "Approximately 20% of estuaries in State XYZ have an elevated trophic state."

- Trends at a place: "Nutrient concentrations in estuary ABC have decreased by 20% since 1980."

- Trends of a region: "Average estuary trophic state in State XYZ has increased by 20% since 1980"; "Average trophic state index values in 20% of estuaries of State XYZ have increased by 15% or more since 1980."
- Relationships among variables: "A 50% increase of N loading above natural background is associated with decline in taxa richness of benthic macroinvertebrates, below reference expectations."; "Coastal waters receiving runoff from large nonpoint sources have 50% greater probability of elevated trophic state above reference conditions than coastal waters not receiving such runoff."

Sources of Variability

Variability of measurements has many possible sources, and the intent of many sampling designs is to minimize the variability due to uncontrolled or random effects, and conversely to be able to characterize the variability caused by experimental or class effects. For example, estuaries may be classified by soil phosporus content of their surrounding watersheds so that estuaries within a class are likely to have similar water column concentrations in current or historical reference areas. The population of estuaries is stratified so that observations (sample units) from the same stratum will be more similar to each other than to sample units in other strata.

Environmental measures vary across different scales of space and time, and sampling design must consider the scales of variation. In coastal waters, measurements of some variables such as total nitrogen or chlorophyll concentrations are taken at single points in space and time (center of the deep depression, 20 m depth, 10 a.m. on 2 July). If the same measurement is taken at a different place (littoral zone, 1 m), or coastal waters, or time (30 January), the measured value may be different. A third component of variability is the ability to accurately measure the quantity of interest, which can be affected by sampling gear, instrumentation, errors in proper adherence to field and laboratory protocols, and the choice of methods used in making determinations.

The basic rule of efficient sampling and measurement is to sample so as to minimize measurement errors, to maximize the components of variability that have influence on the central questions and reporting units, and to control other sources of variability that are not of interest, that is, to minimize their effects on the observations. In the example of chlorophyll concentrations, variability could be reduced by sampling each of several coastal waters in the deepest part, with multiple depth samples or a vertically integrated pump sample taken in early spring before stratification appears. Many coastal waters are sampled to examine and characterize the variability due to different coastal waters (the sampling unit). Each coastal water is sampled in the same way, in the same place, and in the same timeframe to minimize variability due to location, depth, and season, which are not of interest in this particular study.

In the above example, chlorophyll concentrations vary with location within a coastal water, among coastal waters, and time of sampling (day, season, year). If the spatial and temporal components of variability within coastal waters are large (e.g., measurements of chlorophyll concentrations typically vary more between spring and fall samples within a coastal water than they do among coastal waters), then it may be best to use an index period. For this reason, coastal water chlorophyll concentrations often are estimated as a growing season average, estimated from several determinations (e.g., monthly) during the growing season.

In statistical terminology, there is a distinction between sampling error and measurement error that has little to do with actual errors in measurement. Sampling error is the error attributable to selecting a certain sample unit (e.g., a coastal water or a location within a coastal water) that may not be representative of the population of sample units. Statistical measurement error is the ability of the investigator to accurately characterize the sampling unit. Thus, measurement error includes components of natural spatial and temporal variability within the sample unit as well as actual errors of omission or commission by the investigator. Measurement error is minimized with methodological standardization: selection of cost-effective, low-variability sampling methods; proper training of personnel; and quality assurance procedures to minimize methodological errors. In analytical laboratory procedures, measurement error is estimated by replicate determinations on some subset of samples (but not necessarily all). Similarly, in field investigations, some subset of sample units should be measured more than once to estimate measurement error.

Analysis of variance (ANOVA) can be used to estimate measurement error. All multiple observations of a variable are used (from all coastal waters with multiple observations), and coastal waters are the primary effect variable. The root means square error (RMSE) of the ANOVA is the estimated variance of repeated observations within coastal waters. Note that a hypothesis test (F-test) is not of interest in this application, only the RMSE of the analysis.

Natural variability that is not of interest for the questions being asked, but which may affect the ability to address them, should be estimated with the RMSE method above. If the variance estimated from RMSE is unacceptably large (i.e., as large or larger than variance expected among sample units), then it is often necessary to alter the sampling protocol, usually by increasing sampling effort in some way to further reduce the measurement error. Measurement error can be reduced by multiple observations at each sample unit, for example, multiple ponar casts at each sampling event; multiple observations in time during a growing season or index period; depth-integrated samples; or spatially integrated samples.

A less costly alternative to multiple measures in space is spatially composite determinations. In nutrient or chlorophyll determinations, a water column pumped sample, where the pump hose is lowered through the water column, is an example of a spatially composite determination. Spatial integration of an observation and compositing the material into a single sample is almost always more cost-effective than retaining separate, multiple observations. This is especially so for relatively costly laboratory analyses such as organic contaminants and benthic macroinvertebrates, but the price of this economy is loss of information about the water column or about distribution over an area.

Statistical power is the ability of a given hypothesis test to detect an effect that actually exists and must be considered when designing a sampling program (e.g., Peterman 1990, Fairweather 1991). The power of a test (1-b) is defined as the probability of correctly rejecting the null hypothesis when the null hypothesis is false (i.e., the probability of correctly finding a difference [impairment] when one exists). For a fixed confidence level (e.g., 90%), power can be increased by increasing the sample size or the number of replicates, except in cases where the variance is proportional to the mean. To evaluate power and determine sampling effort, an ecologically meaningful amount of change in a variable must be set.

Optimizing sampling design requires consideration of tradeoffs among the measures used, the effect size that is considered meaningful, desired power, desired confidence, and resources available for the sampling program. Every study requires some level of repeated measurement of sampling units to estimate precision and measurement error. Repeated measurement at 10% of sites is common among many monitoring programs.

Alternative Sampling Designs

Sampling design is the selection of a part of a population to observe the attributes of interest to estimate the values of those attributes for the whole population. Classical sampling design makes assumptions about the variables of interest; in particular, it assumes that the values are fixed (but unknown) for each member of the population until that member is observed (Thompson 1992). This assumption is perfectly reasonable for some variables, say, length, weight, and sex of members of an animal population, but it seems less reasonable for more dynamic variables such as nutrient concentrations, loadings, or chlorophyll concentrations of estuaries. Designs that assume that the observed variables are themselves random variables are model-based designs, where prior knowledge or assumptions are used to select sample units.

Probability-Based Designs (Random Sampling)

The most basic probability-based design is simple random sampling, where all possible sample units in the population have the same probability of being selected; that is, all possible combinations of n sample units have equal probability of selection from among the N units in the population. If the population N is finite and not excessively large, a list can be made of the N units, and a sample of n units is randomly selected from the list. This is termed list frame sampling. If the population is very large or infinite (such as locations in an estuary), one can select a set of n random (x,y) coordinates for the sample.

All sample combinations are equally likely in simple random sampling, thus there is no assurance that the sample actually selected will be representative of the population. Other unbiased sampling designs that attempt to acquire a more representative sample include stratified, systematic, multistage, and adaptive designs. In stratified sampling, the population is subdivided or partitioned into strata, and each stratum is sampled separately. Partitioning is typically done so as to make each stratum more homogeneous than the overall population; for example, estuaries could be stratified on ecoregion or coastal waters by dominant current structure. Systematic sampling is the systematic selection of every k^{th} unit of the population from one or more randomly selected starting units, and it ensures that samples are not clumped in one region of the sample space. Multistage sampling requires selection of a sample of primary units, such as fields or hydrologic units, and then selection of secondary sample units, such as plots or estuaries within each primary unit in the first-stage sample.

Estimation of statistical parameters requires weighting of the data with inclusion probabilities (the probability that a given unit of the population will be in the sample) specified in the sampling design. In simple random sampling, inclusion probabilities are by definition equal, and no corrections are necessary. Stratified sampling requires weighting by the inclusion probabilities of each stratum. Unbiased estimators have been developed for specific sampling designs and can be found in sampling textbooks, such as Thompson (1992).

Model or Goal-Based Designs

Use of probability-based sampling designs may miss relationships among variables (models), especially if there is a regression-type relationship between an explanatory and a response variable. As an example, elucidation of estuary response to N loading with the Vollenweider-type model; that is, chlorophyll *a* concentration regressed against a depth-normalized N concentration (Vollenweider 1968) requires a range of trophic states from ultra-oligotrophic to hypereutrophic. A simple random sample of estuaries is not likely to capture the entire range (i.e., there would be a large cluster of "mesotrophic" estuaries with few at high or low ends of the trophic scale), and the random sample therefore may be biased with respect to the model.

In model-based designs, sites are selected based on prior knowledge of auxiliary variables, such as estimated phosphorus loading, estuary depth, and elevation. Often, these designs preclude an unbiased estimate of the population response variable (e.g., trophic state), unless the model can be demonstrated to be robust and predictive, in which case the population value is predicted from the model and from prior knowledge of the auxiliary (predictive) variables. Selection of unimpacted reference sites is an example of samples for a model (index development; response of index variables to measures of anthropogenic influence) that cannot later be used for unbiased estimation of the biological status of estuaries. Ideally, it may be possible to specify a design that allows unbiased estimation of both population and model. Statisticians should be consulted in developing the sample design for a nutrient criteria program.

Sampling and Analytical Designs for More Complex Ecological Questions

Complex ecological questions may not be required to develop numerical nutrient criteria. However, the manager may require a biological and an ecological assessment of resources at risk to establish management goals—that is, focus on biological resources of high social and economic value (e.g., the Chesapeake Bay Program includes biological variables as part of the goal setting process). Questions on how to sample different levels of biological organization (e.g., populations, communities, and ecosystems), indicators of stress, diversity and similarity measures, and biotic indices may become important. Criteria that go beyond the core variables will likely address one or more of these ecosystem or community elements. Publications are available that provide conceptual and statistical guidance for monitoring biological/ecological systems, including multivariate analytical approaches (e.g., Spellerberg 1991, Luepke 1979, Digby 1987, Clark and Warwick 1994a,b, Ott 1995, Ludwig and Reynolds 1988, Eckblad 1991).

Monitoring Programs

The purpose of monitoring is to obtain data that can be used not only to determine reference condition, but to help classify estuaries and coastal waters, or portions thereof, into groups (see Chapter 3). Classification should aid in the determination of reference sites or stations that are representative and have the lowest possible variability.

In some cases, a problem may exist where monitoring data indicate that the system has been greatly impaired from nutrient enrichment over the period of record. This is analogous to the so-called "corn-belt" problem in some lakes in the upper mid-West of the United States (see Lakes and Reservoirs Nutrient Guidance Document). This problem suggests that a meaningful reference condition may no longer exist. Or, the system has been greatly disturbed and it is not clear to what extent the impairments are due to nutrient enrichment. In this case, both historical information and diagnostic sampling may be required to clarify the reference condition and subsequent nutrient criteria.

Where data are insufficient, several approaches can be tried, for example, running a mathematical nutrient model "backwards"; use of biostratigraphic approaches, including changes in algal dominance and composition, loss of submerged aquatic vegetation (SAV), and pyritization of iron in sediments to detect earlier anoxia (Brush 1984, 1986, 1992, Cooper 1995) or reference to old written accounts (e.g., newspapers, diaries). For example, there are accounts of water clarity in the mouth of the Patuxent River estuary, Chesapeake Bay, in the late 1930's where engineers sitting in a "Beebe-like Bathysphere" on the estuary bottom could see horizontally approximately 20 to 30 feet. The methods discussed here are often qualitative to semiquantitative, but such information can be useful, especially if marked nutrient increases have evidently occurred over historical conditions but ambient data are insufficient. Older aerial photographs and other forms of watershed land use information and human population density trends can help make extrapolations regarding the system's response to nutrient loading.

Parameters to Survey

Each of the core variables discussed in Chapter 4 must be included in the survey (e.g., concentrations of TN, TP [total phosphours], chlorophyll *a*, and a measure of water clarity, such as Secchi disc, submersible PAR meter, or spectral radiometer). It is also appropriate to measure salinity, water temperature, flow and direction, tide phase, pH, and nutrient load to help better interpret the core variables. This is a much different problem than usually experienced in rivers and lakes. It may be desirable in some circumstances to include secondary variables, for example, vertical dissolved oxygen profiles, distribution and abundance of SAV/seagrasses, distribution of tidal emergent marshes, distribution and density of benthic filter feeders (e.g., oysters), water color, dissolved organic carbon (especially if humic-like materials are abundant), and particulate organic carbon. This more complex array of variables would require a diagnostic justification.

Sampling Frequency

A single grab sample from an estuary or coastal water will be grossly inadequate. Estuaries are near the bottom of watersheds, which makes them prone to episodic rainfall events. Coastal waters are also subject to seasonal storms that churn the waters and physically disturb shallow sediments, and these events may be seasonally highly variable. If information is available to set expectations when possible seasonal pulses of freshwater occur, then it should be used to help schedule the sampling of wet and dry periods. In north temperate estuaries, where winter to early spring is the dominant freshet period, this interval should be included in the sampling scheme. A lag of hours, days, or several weeks to one or more months is usually required to detect the system’s response to the nutrient load, depending on magnitude of the freshet relative to volume of the estuary or mixing zones of coastal waters. This also may capture any spring blooms of diatoms if such occur. A midsummer and early fall survey should give a first-order picture of the nutrient concentration and response variable pattern suitable for classification. In the event of variable summer freshwater flows, then more frequent sampling may be required. Because different patterns of rainfall exist around the coasts, regional considerations should weigh heavily in the design of sampling schedules.

Long-term datasets have well-documented ecological value (Likens 1992, Wolfe et al. 1987, Livingston 2001a); however, all too frequently resources constrain longer term sampling which can average out short-term variability. Recent data connected to long-term trends provide the strongest case for classification, reference condition determination, and other criteria development. By measuring the nutrient load, especially during freshet and low flow periods and concurrently with ambient water quality and hydrographic sampling, one can get an estimate of the load and salinity/nutrient and response variable relationships while keeping in mind the precautions noted above. For comparative purposes, it is important to compare core monitoring variables under similar salinity conditions.

If tidal elevation is large (e.g., greater than 2.0 m), then this component of estuarine flushing probably dominates over nontidal gravitational flows (Monbet 1992), and eutrophication symptoms are likely to be of a small magnitude. In some estuaries (e.g., York River estuary of the Chesapeake Bay), spring tides may break down density stratification, and the system responds differently to nutrient supplies than during periods of relatively strong stratification (Hass 1977). Thus, for estuaries with tidal elevations less than 2.0 m, it is important to note that they are likely to be quite vulnerable to nutrient enrichment.

A general rule of thumb regarding freshwater run-off events to estuaries is that a large freshet may displace the nutrient supply and responses will be detected seaward of the focal area. A modest freshet may not deliver enough nutrients or physically affect the density stratification to make an estuary vulnerable to nutrient enrichment. But an intermediate freshet may cause the focal area to receive a significant nutrient load and establish a strong vertical density gradient so maximum responses will be detected (e.g., high average chlorophyll *a* concentrations and minimum Secchi disk readings). This rule is less easily applied to coastal waters.

Sampling Locations

Sampling locations depend on the size (and especially the length of an estuary), bathymetry, nutrient source inputs, and hydrography (especially the longitudinal and vertical salinity profiles). In estuaries, consideration should be given to tidal freshwater, the turbidity maximum (if one is present), mesohaline and polyhaline regimes, as well as water below zones of density stratification.

In large tidal freshwater riverine systems, it is important to employ several stations because this portion of the estuary may “store” a large supply of nutrients that later advect into the saline reach of the estuary (e.g., the Hudson River system) (Lampman et al. 1999). Enough samples should be taken to detect nutrient concentration gradients along the salinity gradient from tidal river to the estuary receiving waters. Typically, this will require from five to seven stations at a minimum. If the estuary is relatively wide (e.g., lagoonal systems such as Pamlico Sound, NC, or Pensacola Bay, FL) or has large tributary creeks, then these features may need independent sampling. Where salinity gradients are distinct both horizontally and vertically, composite sampling may have severe limitations. Depth variability also should be considered, for example, main channel, shelf samples, and samples in shallow water near SAV/seagrass meadows or in emergent marsh channels should be included. Emergent marsh creeks should be sampled in the summer during high and low tides, because high system respiration may cause hypoxia/anoxia in these tidal creeks that may be largely natural. Where SAV meadows are poorly developed, resuspension of bottom sediments may be more common and not represented by open channel samples.

Serious consideration should be given to some replicate sampling within salinity zones to estimate variability; however, resources may require a broad picture where gradients become equally or more important than the physical salinity “zones.” In most cases, analytical levels of detection should be a trivial aspect of data acquisition for reference characterization. This does not free one from application of good laboratory quality asurance/quality control (QA/QC) practices, which must be maintained with appropriate blanks, reference samples, and other considerations to standard analytical measurements.

Citizen Monitoring Programs

Citizen monitoring programs have greatly increased, especially since the early 1980's. Where there is adequate technical oversight either from within the group expertise or from the outside, such monitoring efforts can play an important role in assessing trends, identifying “hotspots,” and locating likely sources of nutrients, especially in smaller estuaries where larger research vessels are not required. Many Federal, State, Tribal, and local agencies assist citizen monitoring efforts, and these agencies contribute to training and direction, development, and implementation of QA/QC procedures, act as a data repository; and perform analyses on environmental samples collected by citizen groups. Citizen monitoring groups often can provide more frequent observations, such as visiting a gauging station, than can State personnel. Citizens also can identify those property holders or resource users not following best management practices or operating within permit limits. See also Volunteer Monitoring Programs, above.

5.4 QUALITY ASSURANCE/QUALITY CONTROL

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

Representativeness

Sampling program design (when, where, and how you sample) should produce samples that are representative or typical of the environment being described. Sampling designs for developing nutrient criteria are discussed earlier in this chapter.

Completeness

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

Comparability

To compare data collected under different sampling programs or by different agencies, sampling protocols and analytical methods must demonstrate comparable data. The most efficient way to produce comparable data is to use sampling designs and analytical methods that are widely used and accepted such as Standard Methods for the Examination of Water and Wastewater (APHA, AWWA, WEF, 1998) and EPA methods manuals.

Accuracy

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

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

Variability

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

5.5 STATISTICAL ANALYSES

Statistical analyses are used to identify variability in data and to elucidate relationships among sampling parameters. Several statistical approaches for analyzing data are mentioned here. We advocate simple descriptive statistics for initial data analyses, that is, calculating the mean, median, mode, ranges, and standard deviation for each parameter in the system of interest. The National Nutrients Database discussed above calculates simple descriptive statistics for queried data. Specific recommendations for setting criteria using frequency distributions are discussed in Chapter 7.

Data Reduction

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

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

Frequency Distributions

Frequency distributions can be used to aid in the setting of criteria. Frequency distributions do not require prior knowledge of individual waterbody conditions before setting criteria. Criteria are based on and, in a sense, developed relative to the population of systems in the Region, State, or Tribal jurisdiction.

Data plotted on a scale of mean nutrient concentration versus frequency of occurrence for a specific estuary, portion of an estuary, or coastal reach produces a frequency distribution of mean or median nutrient concentration. Plots of frequency distributions of median TP, median TN, median chlorophyll *a*, and Secchi depth for the index period (discussed in Chapter 4) should be examined to determine the normality of the data in the distribution and to determine the potential for further subdivision of the waterbody under investigation. Data that are not normally distributed often are transformed into a distribution more approximating the normal distribution by taking the logarithm of each value. Analysis of outliers may assist in explaining variability in small datasets; additional analysis can be conducted to identify the statistical significance of population differences.

Correlation and Regression Analyses

The relationship between two variables may be of use in analyzing data for criteria derivation. Correlation and regression analyses allow the relationship to be defined in statistical terms. A correlation coefficient, usually identified as r, can be calculated to quantitatively express the relationship between two variables. The appropriate correlation coefficient is dependent on the scale of measurement in which each variable is expressed (whether the distribution of data is continuous or discrete) and whether there is a linear or nonlinear relationship. Results of correlation analyses may be represented by indicating the correlation coefficient and represented graphically as a scatter diagram that plots all of the collected data, not just a measure of central tendency. The statistical significance of a calculated correlation coefficient can be determined with the t test. The t test is used to determine if there is a true relationship between two variables. Therefore, the null hypothesis states that there is no correlation between the data variables measured within the population. A critical α value is chosen as a criterion for determining whether to reject the null hypothesis. If the null hypothesis is rejected, the alternate hypothesis states that the correlation at the calculated r value between the two variables is significant.

Regression analysis provides a means of defining a mathematical relationship between two variables that permits prediction of one variable if the value of the other variable is known. In contrast to correlation analysis, there should be a true independent variable (a variable under the control of the experimenter) in regression analysis. Regression analysis establishes a relationship between two variables that allows prediction of the dependent variable (predicted variable) for a given value of an independent variable (predictor variable). However, scientists (other than statisticians) apply regression analyses to field data when a relationship is known to exist, even when there is no true independent variable (e.g., cell counts of algae and chlorophyll concentration; nutrient concentrations and chlorophyll concentration) (Ott 1988, Campbell 1989, Atlas and Bartha 1993, Ott 1995).

Tests of Significance

Various statistical tests are used to assess the hypotheses being tested. Statistical tests of significance differ in their applicability to the dataset of interest and the power of the test (the ability of the test to detect a false null hypothesis). A parametric test of significance assumes a normal distribution of the population. Nonparametric analyses are valid for any type of distribution (normal, log-normal, etc.) and can be used if the data distribution is not normal or unknown. A parametric test has more power than a nonparametric test when its assumptions are satisfied. Two types of errors can be made when testing hypotheses: Type I—where a correct null hypothesis is mistakenly rejected, and Type II—when there is a failure to reject a false null hypothesis. The parametric test is less likely than a nonparametric test to make a Type II error, when the assumptions are met. Therefore, if given a choice, the parametric test should be used rather than the nonparametric test when the assumptions of the parametric test are fulfilled. Less powerful, nonparametric tests of significance must be used in cases where the data do not fit the assumption of a normal distribution (Ott 1988, Campbell 1989, Atlas and Bartha 1993). Parametric tests include the student *t* test, analysis of variance, multivariate analysis of variance, and multiple range tests. Nonparametric tests include chi square, Mann Whitney U test, and the Kruskal–Wallis test (Ott 1988; Campbell 1989; Atlas and Bartha 1993). Detailed descriptions of these and other relevant statistical tests can be found in standard statistical texts.

CHAPTER 6

Determining the Reference Condition

Introduction and Definition
Significance of Reference Conditions
Paucity of Similar Estuarine and Coastal Marine Ecosystems
Approaches for Establishing Reference Conditions

6.1 INTRODUCTION AND DEFINITION

A reference condition is the comprehensive representation of data from several similar, minimally impacted, "natural" sites on a waterbody or from within a similar class of waterbodies, i.e., median values of TN, TP, chlorophyll *a*, or Secchi depth. However, in cases where severe degradation has occurred, surrogate values for reference site data may be required as described in this chapter. There are two basic approaches for their determination: (1) analysis of in situ estuarine and coastal data, and (2) analysis of watershed nutrient loading to estuaries and, through advective transport, nutrient loading to the coastal environment. These approaches reinforce each other, but one may be preferred or even required depending on comparative or site-specific data. Reference conditions are a primary element of nutrient criteria development, but should be used in conjunction with the other elements described in Chapter 1 and Chapter 7. Classification of estuaries and coastal waters should facilitate development of reference conditions but, as pointed out in Chapter 3, further research is essential to bring classification of these systems to the level of practical utility comparable to that of most freshwater systems. Models of estuarine susceptibility to nutrient overenrichment are at an early stage of development, and even less may be known about coastal ecosystems (Chapter 3).

6.2 SIGNIFICANCE OF REFERENCE CONDITIONS

The reference condition is made explicit through several environmental measures. This manual focuses on TN and TP as principal causative agents, but their relative roles depend on individual watershed/estuary and conditions. There are two response variables: chlorophyll *a*, a measure of algal biomass; water clarity, linked to algal biomass through chlorophyll *a*; and often a third, dissolved oxygen deficiency, particularly in estuaries. These explicit measures are indicators of nutrient enrichment but are linked conceptually to a continuum of biological resources and recreational opportunities (Figure 6-1). These linkages show considerable variability and apparent elasticity because ecological processes are not "hardwired" as are some physically engineered systems (e.g., cogs or pulleys that drive a machine). States and authorized Tribes are encouraged to employ additional response variables.

EPA assumes that ecosystems will support natural assemblages of aquatic life and high-quality recreational activities if a nutrient supply is achieved and maintained at a level to support the natural biological system. This can happen, of course, only if other environmental conditions are compatible.

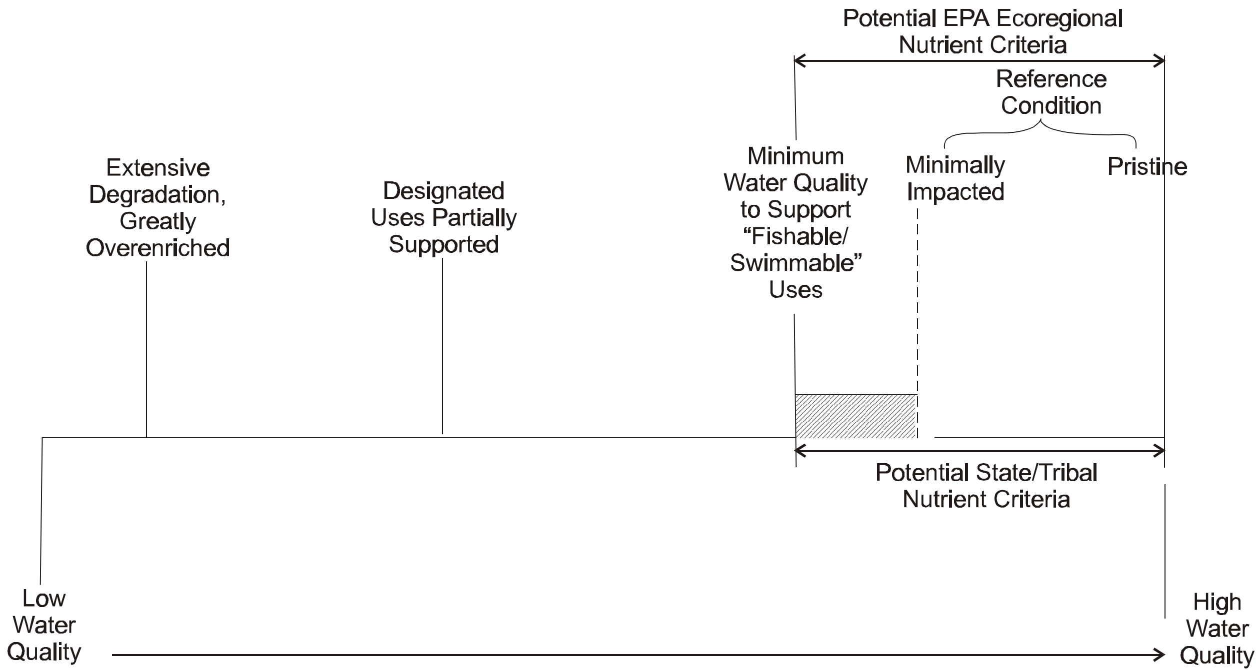

Figure 6-1. Environmental quality scale representing reference conditions and potential nutrient criteria relative to designated uses.

Conditions that support minimal or unimpaired aquatic resources are typically associated with very low human population densities and limited land use activities in watersheds that would otherwise be a source of increased human-mediated nutrient supplies to estuaries and coastal waters. This is a plausible association because once a watershed is moderately to heavily developed, it is practically impossible to control all nutrient inputs to the aquatic system. State and national preserves may have relatively high-quality environmental conditions, but even in these systems atmospheric nitrogen deposition likely has caused some nutrient enrichment. Ideal conditions may still exist in some estuaries and coastal waters, and they should be identified. However, where these ideal nutrient conditions no longer exist, especially in the 50% to 60% of moderately to heavily enriched estuaries (Bricker et al. 1999), the reference condition should be sought from comparative analyses of similar systems and/or the historical record, which provides an implied reference condition.

In this manual, a distinction is made for estuaries and coastal waters between “pristine” and minimally impacted waters. The precolonial period may have had water quality and habitat conditions, including nutrient loading, that were pristine. This would approximate the ideal of “restoration and protection of physical chemical and biological integrity” that is the ultimate goal of the Clean Water Act. But this ideal is largely hypothetical, because methods to estimate it invariably contain a relatively high degree of uncertainty. However, “fishable and swimmable” conditions are commonly used as an interim goal of the act and represent a goal of the Nutrient Program, exemplified by reference conditions associated with minimal human-mediated nutrient enrichment.

The term “fishable and swimmable” is not easily quantified in any waters because of their inherent natural variability. A lake example helps provide a perspective. When a lake shifts from an oligotrophic to a mesotrophic fishery because of nutrient overenrichment, at some point the change becomes demonstrable. Likewise, in an estuary when nutrient enrichment shifts phytoplankton production and algal species composition toward microbial dominance and away from oyster production, the change becomes demonstrable. What is needed are early warning indicators of the impending change. This is the role of nutrient criteria. Although it is clear that additional research is called for, there is sufficient symptomatic knowledge such as changes of species with enrichment to merit setting such indicator criteria.

Identifying a reference condition in degraded waters should start with an analysis of the best existing estuarine or marine waters within a watershed or coastal area, or as commonly stated, “the best of what’s left.” Because of the difficulties in identifying reference sites in some overenriched estuaries and nearshore coastal waters, it may be necessary to derive an “implied” reference condition by comparing the “best of what’s left” with “what used to be” as established by the historical record. In any case, it is still important to identify the best remaining sites in the waterbody of concern. Where the the “best” sites are known to be severely degraded, more emphasis must be placed on the historical record, but some knowledge of the continuum from past to present is necessary to establish protective criteria.

Nutrient enrichment-based impacts are a function of the concentrations and supply of nutrients as well as the ecological conditions and nutrient processes characteristic of the system. Nutrient enrichment effects may be exacerbated in estuaries where the dominant grazing populations (e.g., menhaden and oysters) have been lost through human causes, natural causes, or a combination of the two over the past century or the past several decades. For this reason, it is important to assess the factors that may have modified the “assimilative capacity” of coastal waterbodies. Reference conditions are not threshold values (a concentration less than some specified value). For example, in the lower Potomac River estuary, Chesapeake Bay, the average N load caused a larger phytoplankton bloom than either drought or flood conditions (Chapter 2). This response involved system hydrodynamics. The point is that reference conditions should be interpreted in an ecosystem context, especially when the system has experienced significant nutrient overenrichment and/or is subject to periodic natural disruption such as hurricanes, winter storms, or droughts.

Some argue that one of the earliest symptoms of impairment involves a nutrient stimulation of harmful algae relative to beneficial algae. Many types of algal blooms that become a nuisance or harmful clearly are a form of pollution. There is mounting evidence that many harmful algal blooms are associated with nutrient enrichment (NRC 2000). For example, substantial loss of seagrass habitat due to algal shading or bottom habitat due to hypoxia are associated with negative effects on living resources. Seagrasses also stabilize shorelines and provide cover from predators.

This manual emphasizes the importance of reference conditions to address nutrient problems in a timely manner. They serve as the best initial measure for identifying nutrient loads that could cause use impairments. Statistical and computer-based modeling can improve site-specific estimates of the load and response relationships. Classification may assist in extrapolating nutrient effect relationships between systems in the same class, although classification of coastal waters probably has less predictability and utility than it does for lakes and streams. As pointed out in Chapter 2, for coastal waters, the relationships between nutrient loading (e.g., TN and TP) and the response indicators of chlorophyll *a* and water clarity normalized to chlorophyll *a* can be less than straightforward. However, an understanding of the reference condition will help prevent resource managers from being blindsided by complications associated with cause-effect relationships.

6.3 PAUCITY OF SIMILAR ESTUARINE AND COASTAL MARINE ECOSYSTEMS

Estuaries and coastal marine ecosystems tend to be relatively individualistic in their sensitivity and response to nutrient overenrichment. Susceptibility to nutrient enrichment ranks as a premier research need (Hobbie 2000). The lack of physically similar waterbodies may severely limit grouping (classifying) waterbodies as recommended for lakes, reservoirs, rivers, and streams where frequency distributions are used to derive reference conditions (e.g., upper 75th percentile of a priori nutrient-unimpaired waterbodies or lower 25th percentile of all waterbodies; EPA 2000a,b). As mentioned in Chapter 1, an exception may be coastal embayments that form behind barrier islands. If several relatively similar embayments can be identified within a given geographic area, then one or more may serve as benchmarks against which the others may be compared.

6.4 APPROACHES FOR ESTABLISHING REFERENCE CONDITIONS

Three primary estuarine approaches may provide considerable flexibility to meet the diversity of conditions encountered by resource managers. A fourth approach focuses on nutrient loading from the watershed. A fifth approach is described for coastal marine waters. There are situations where light and estuarine flushing limit the expression of nutrient enrichment effects. In such cases, downstream effects may nonetheless be a problem requiring attention. Consequently, we have proposed a variety of criteria development approaches. Reference conditions in any case should be defined with due consideration of salinity gradients and seasonal and interannual variability. Table 6-1 summarizes the five approaches to establishing reference conditions in estuaries and coastal waters.

The alternatives for reference condition determination presented in the following text and table represent two general approaches to developing baseline nutrient quality measurements. The reference conditions approach per se acknowledges that the system response to overenrichment can be extremely variable, as described earlier in this manual. For that reason, measuring the nutrient characteristics of minimally impacted sites provides a reliable nutrient goal regardless of how those nutrients may or may not be assimilated. This is the approach upon which the National Nutrient Criteria Program is predicated, and reference sites should always be sought when designing nutrient criteria protocols.

Table 6-1. Summary of estuarine and coastal nutrient reference condition determinations

Degree of apparent estuarine degradation	Method recommended	Criterion measure
A. In Situ Observations as the Basis for Estuarine Reference Condition		
1. Recognized unique excellent condition	Median ambient concentration. Fig. 6-2.	Concentration of TP, TN, chlorophyll *a*, Secchi depth (m)
2. Some degradation, but reference sites exist	Upper quartile. Fig. 6-2.	Same as above
3. Significantly degraded, including all potential reference sites	Intercept value on a regression or distribution curve as illustrated in Fig. 6-3 and 6-4 or by use of a comparable comparative regression model.	Same as above
B. Watershed-Based Approaches for Estuarine Reference Condition		
4. Same as approach 3 above, but insufficient historical data	Ref. sites along each trib. and calculate delivery. Summation is reference condition. Fig 6-5. Model required to back-calculate load where all tribs. are degraded	Load of TP and TN; model is required to convert load to estuarine concentration
C. Coastal Reference Condition		
5. Applicable to all coastal reaches - Estuarine plumes - Coastal areas	Index site approach; models may help distinguish anthropogenic contribution See also Appendix H.	Concentrations

However, many estuaries are so degraded and/or exhibit such short retention times that investigators cannot determine reference sites with any degree of confidence. In this instance, dose-response curves and similar approaches as described below may be more appropriately applied in determining historical reference conditions, i.e., conditions before degradation was first exhibited. Although these approaches may not provide the real-time affirmation of existing reference sites, their strength is the documentation of system decline coincident with the overenrichment. The investigator is expected to assess the characteristics of the given estuary and select the most responsive option for reference condition determination from those presented.

In all cases, historical information is important either as an alternative reference condition for heavily enriched systems or as one of the five elements of criteria development essential to providing a status and trend perspective important to data evaluation before criteria are established.

In Situ Observations as the Basis for Estuarine Reference Condition

These approaches require nutrient and relevant hydrographic data within an estuary.

Recognized Unique Excellent Condition

Typically, this condition is based on an extensive spatial and temporal scientific database. If an existing excellent condition is agreed to by the RTAG and stakeholders, then the State, authorized Tribe, or appropriate government agency may establish reference conditions based on data that document the condition and address the remaining elements of criteria development. With limited data, in a very small number of cases it may be possible to document that the watershed is unimpacted (e.g., has very little human development, is distant from the influence of local population centers, adjacent land uses are relatively undisturbed, and is outside of major atmospheric deposition of nitrogen). It is necessary to document, or augment with additional data, to ensure the original hypothesis is confirmed. Segmenting the estuary by salinity zones, typically based on estuarine circulation, may be required to reflect nutrient conditions associated with salinity gradients. However, the geographic scale of the nutrient overenrichment problem suggests that this a priori approach likely will be limited to a relatively small number of estuaries and coastal waters (Chapters 1 and 2). The data can be summarized as either medians of the indicator endpoints or frequency distributions (Figure 6-2). Areas that meet or come very close to minimally impaired conditions include Plum Island Sound and Blue Hill Bay, ME, and lower Narragansett Bay, RI.

Some Degradation Exists But Reference Sites Can Be Identified

Two situations may be applicable for characterizing reference conditions where some degradation occurs.

A. Some Minimally Impaired Sites Are Available

Reference sites should be representative of the system (e.g., a branched estuarine system where one branch is unimpaired by nutrients and otherwise relatively similar ecological conditions prevail). Comparisons should be made among similar salinity zones. This example may apply only rarely because in atmospheric nitrogen deposition and land use practices likely will have so altered the landscape that truly undisturbed conditions are unavailable. In those cases where minimal biological resource uses are impaired by nutrient overenrichment, then reference conditions for nutrients should be deemed to occur. Clearly, point and nonpoint source discharges must be at a minimum. Land cover in the watershed should be very close to natural for the ecoregion or, if modified in the past, then recovery must be well along (e.g., forests should be near the anticipated climax condition for the region). The reader is referred to Chapter 4 of “Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance,” December 2000 (EPA-822-B-00-024), for additional information (U.S. EPA 2000b). Although the upper estuary likely does not qualify, reference-quality sites probably occur in lower Delaware Bay, especially under sustained periods of low Delaware River input. Under these conditions, because the measurement values likely will not exhibit a trend or show high variability, the data can be summarized in terms of median values or a frequency distribution.

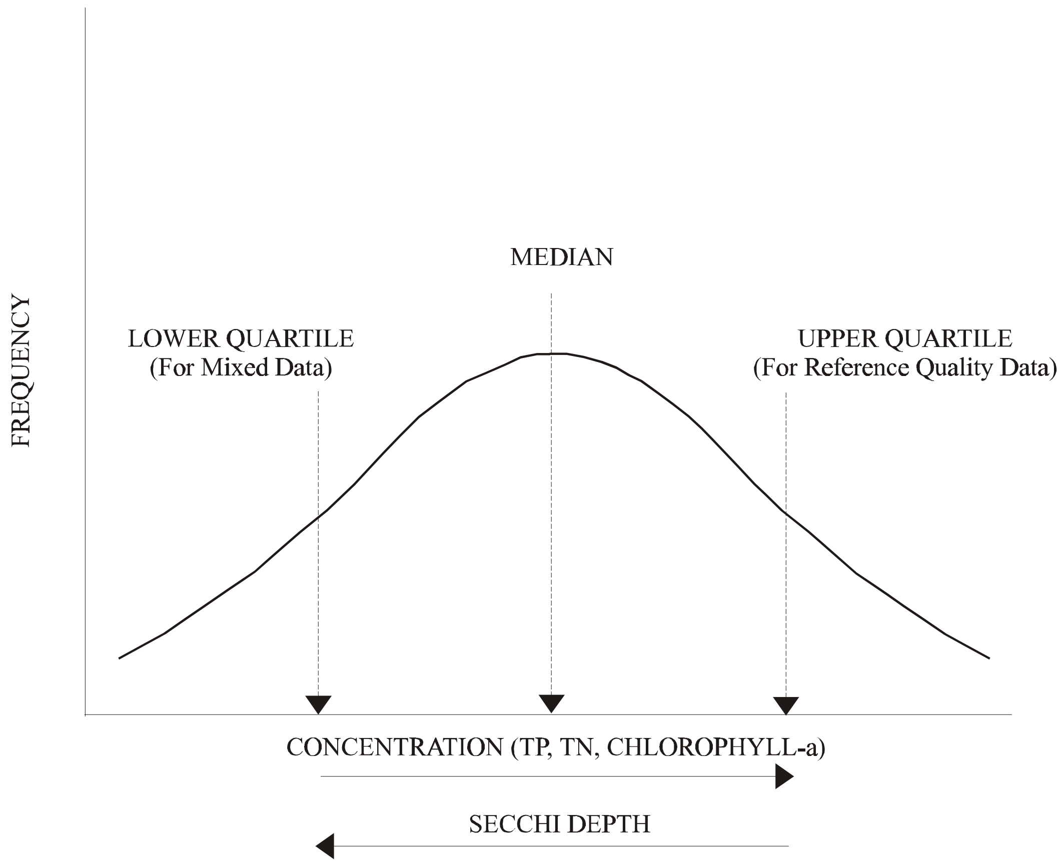

Figure 6-2. Hypothetical frequency distribution of nutrient-related variables showing quantities for reference or high-quality data and mixed data (all data incuded).

B. Reference Condition Derived From A Priori Selection of a Subset of Reference Estuaries or Coastal Waters Within a Class of Systems

This approach may apply to a series of coastal embayments located within a physically similar reach of coastline (e.g., some estuaries along the Maine coast and possibly New England salt ponds). Where freshwater inputs contribute to a strong salinity gradient, salinity zones may be compared. The goal is to establish frequency distributions for similar systems. If a population of 15 or more estuaries or embayments occurs within the class, then a frequency distribution may be applicable (see Chapter 6, Nutrient Criteria Technical Guidance Manual–Lakes and Reservoirs, April 2000, EPA-822-B00-001). A sample size of fewer than 15 systems will not likely have enough predictive power to justify application of this approach. However, if only a small number exists, then one or two embayments may serve qualitatively or possibly semiquantitatively as references for the remaining systems. Seasonality and similar freshwater inputs are important to ensure that physical processes are not masking potential nutrient effects. To date, it is unknown whether there are enough reference embayments to make a frequency distribution approach feasible. Other examples may be difficult to identify but tributaries to North Inlet, SC, may qualify.

Significant Degradation Exists—Reference Sites Cannot Be Identified From Current Monitoring Data

Historical Analysis of Estuarine Data

Because of nutrient overenrichment, it is highly likely that no minimally impacted estuarine waters are available for many coastal areas. Under these circumstances, the "best" of the existing waters are clearly degraded. An alternative is to establish reference conditions from the historical record. Three approaches are recommended: (1) analysis of historical ambient nutrient and hydrographic data; (2) analysis of sediment cores to reveal the historical record, including the paleoecological record; and (3) model hind-casting. The analysis of ambient data is likely available only for a small number of estuaries (e.g., Chesapeake Bay and some tributaries, San Francisco Bay, Narragansett Bay, Tampa Bay), because many estuaries were impaired before nutrient data were initially collected.

Empirical In Situ Data Analysis

The first approach depends on the availability of an adequate database. Such a database will need considerable scientific judgment even for systems with a relatively rich database. For example, the ambient nutrient-based historical record for the entire main-stem Chesapeake Bay becomes much less abundant during the 1970s and earlier (Harding and Perry 1997). In addition, the spatial coverage was much less widespread in earlier years. In many systems, early studies focused primarily on the deep channels because of the interest in hypoxia. Analysis of ambient trends in chlorophyll *a* may be complicated by the interaction with freshwater flow, as reported by Harding and Perry (1997) and Hagy et al. (2000). This co-linear effect may affect assessment of trends in water clarity and cause a nonlinear relationship between a limiting nutrient and hypoxia (Chapter 2). Hypoxia is the result of an extended temporal effect of nutrient loading (lag effect), so published empirical relationships usually are based on seasonal or annual nutrient loads (Chapter 2), not on short-term concentration data.

An important aspect of the historical approach is the selection of a period when nutrient loading caused minimal use impairments. Often the nutrient monitoring data are more continuous for past decades than is documentation of use impairments (e.g., visual pollution and reduction in fish productivity) due to nutrient overenrichment. The economics and technology that affect fishery statistics in estuaries and coastal waters are not easily translated into quantitative fish population abundance data. In some systems (e.g., Tampa Bay, Janicki and Wade 1996, Greening et al. 1997), seagrasses were monitored to demonstrate recovery but not necessarily prior to nutrient-based losses. The effects of increased nutrient loads may be confounded by increased suspended sediments. Sediments can cause light limitation and impair benthic habitats, with potential negative effects on living resources.

Historical ambient nutrient-related data will encompass seasonal and interannual variability. It is important that the key variables (TN, TP, chlorophyll *a*,[1] and water clarity—usually Secchi depth) be distributed in a representative spatial and temporal manner. Although the Secchi disc is inexpensive and has been widely used, it is recommended that future measurements of water clarity employ submersible

[1]It is recommended that future algal pigment analyses consider some high-performance liquid chromotography (HPLC) measurements that can also quantify algal groups.

PAR meters as a minimum and that submersible spectral radiometers will be used more frequently. Some precautions are relevant. For example, comparing nutrient-related variables in the turbidity maximum, if light limitation dominates algal net production, would seem to provide little insight into the nutrient problem. Another consideration is to ascertain where on the long-term hydrograph the reference period falls (e.g., a wet or dry decade). These past measurements can then be used as a basis for comparison to present condition (Figure 6-3). If a spring bloom is evident in the data, a judgment is required as to whether remineralization of the spring bloom helps fuel a summer algal bloom, and whether the spring bloom contributed directly to summer hypoxia or other use impairments. Therefore, it is recommended that two averaging periods, a spring and summer period, be used.

The magnitude and duration of historical algal blooms are expected to be much lower than those of current blooms in nutrient overenriched waterbodies under similar physical conditions. Some evidence indicates that the magnitude of algal blooms (e.g., coefficient of variation) may increase as systems become more enriched. At some point light limitation may limit variability. The historical data should be aggregated within a physical classification, such as salinity zones, similar to present data analysis. Several options exist for data summation. A point-in-time estimate may not capture the anticipated seasonal and interannual variability in the data. In some cases, data for one or more key variables may exist before an inflection, indicating worsening conditions. Such variables may include increase in concentrations or loads of TN or TP, increase in chlorophyll *a*, decrease in Secchi disk values, and increase in hypoxic volumes. If this more ideal case prevails, then seasonal medians (medians are less sensitive to outliers or extreme values in a distribution than are means) or medians of seasonal index periods (month of highest chlorophyll *a*) should be calculated. If available, the median of seasonal medians for one or more years is desirable and may be essential to capture interannual variability.

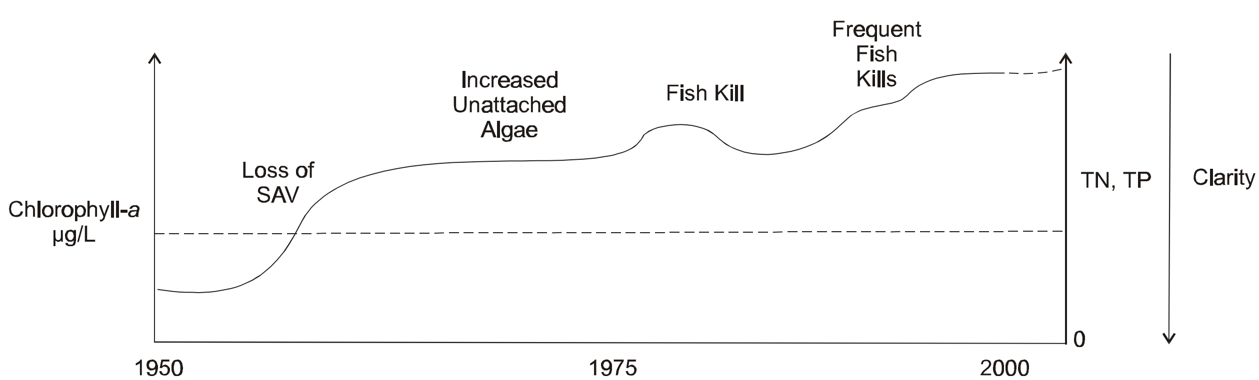

Figure 6-3. Hypothetical example of load/concentration response of estuarine biota to increased enrichment. Dashed line represents the selected reference condition level.

Years may need to be excluded from the reference period if the reference period includes increasing problem-related values (e.g., chlorophyll *a* and extinction coefficients). Nutrient and chlorophyll *a* concentrations and Secchi disk values often will show a gradient within a given salinity zone. In data-rich cases within similar salinity zones, one may express the values as a frequency distribution (e.g., Figure 6-4; see EPA 2000a, Chapter 6, EPA-822-B00-001, for more details). However, attention needs to be paid to potential confounding effects (co-linear) of freshwater input, even when comparing similar salinity zones, because vertical density stratification may be a dominant controlling influence and be dissimilar over time. When the above precautions are addressed and confounding factors are poorly understood, it may be appropriate to set the reference condition at the median between the historical median and the median for present data (Figure 6-4). This simple procedure reflects the magnitude of the departure from minimally impacted waters and is, in part, a function of the length of the historical database, addresses inherent variability, and is a realistic approximation of a reference condition over the time span.

Sediment Core-Based Reconstruction

Sediment core analysis is becoming widely used in assessment of historic biogeochemical and climatological conditions in lake and marine environments (Brush 1984, 1986; Cooper 1995). The approach is applicable in sediment depositional areas where bioturbation and other forms of sediment disturbance are minimal. Improved sediment dating techniques (e.g., lead-210, cesium-154, carbon-14) have contributed to the understanding of historical conditions when sediments were deposited. Certain

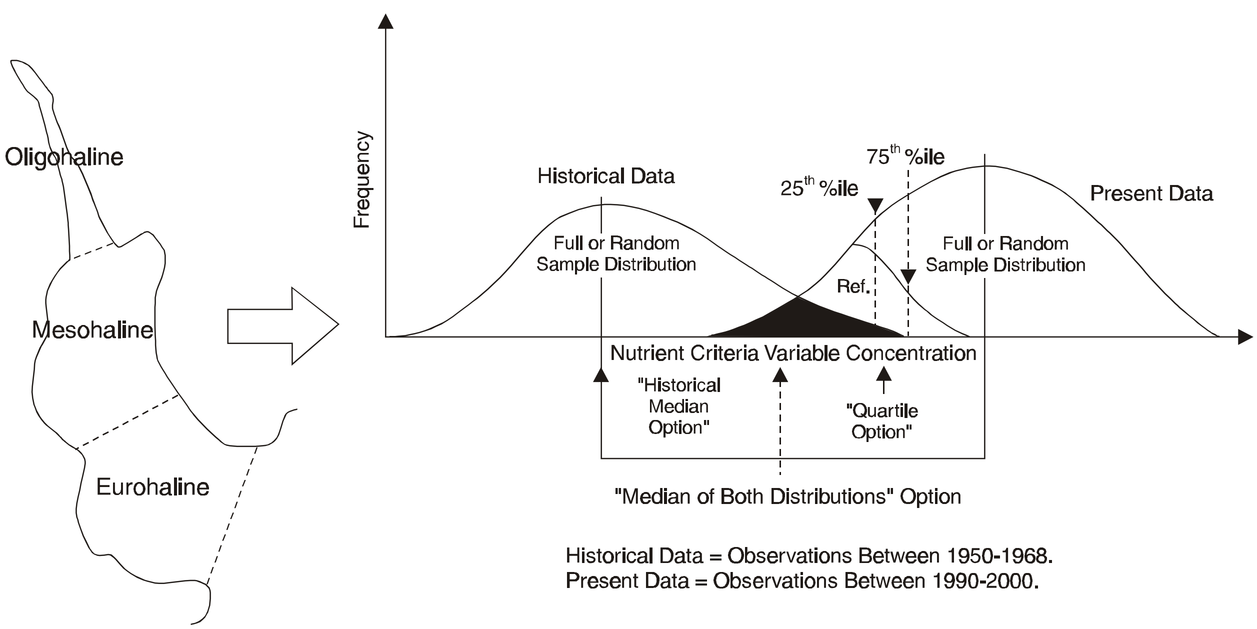

Figure 6-4. An illustration of the comparison of past and present nutrient data to establish a reference condition for intensively degraded estuaries. The option of selecting the distributions from both time periods is compared to an expected frequency distribution if the observations were available.

other chemicals (e.g., plasticizers) began to leave signatures when developed in the 1930s. Sediment cores may provide information about metals and organic chemicals with long half-lives. Many environmental indicators of past sediment conditions now are widely reported in the literature. These include pyrite formation related to anoxia, history of submerged aquatic vegetation, diatom composition related to nutrient enrichment, presence and composition of certain dinoflagellate species, distribution and abundance of indicator species of *Foraminifera*, hard body parts of molluscs and crustaceans, land use based on oak and ragweed pollen, total nitrogen and phosphorus profiles, and stable isotope analysis of carbon, nitrogen, and sulfur in organic materials. Hind-casting with sediment cores provides a means to infer reference conditions at a time when nutrient concentrations were much lower than present. In shallow estuaries where depositional areas are not present or questionable, sediment cores have limited applicability.

Model Hind-Casting

Hind-casting is a controversial approach because it is difficult to calibrate and verify a model when the data difficult to quantify, as with past nutrient and hydrographic conditions. Running a computer-based environmental process model backwards involves many uncertainties; however, where ambient data are inadequate, sediment cores are not applicable, and a model is available and in the hands of experienced scientists, then with expert guidance it seems reasonable to estimate reference conditions on a first-order basis. Use of geographic-based land use models coupled to estuarine hydrodynamic-algal growth models is one approach to hind-casting. Chapter 9 describes several models that may be useful.

Watershed-Based Approaches for Estuarine Reference Condition

An alternative to using in situ coastal marine data to establish reference conditions is to base estimates on watershed nutrient loading characteristics. One can, in some situations, use watershed loading estimates to define conditions in the watershed where nutrient loads would represent minimally impaired waters downstream in estuaries. This assumes that in an undisturbed estuary and its watershed, the nutrient load would historically represent the most natural condition. In some cases, the watershed above the confluence of major freshwater streams may contain a relatively low level of human development, but the nearshore estuarine area may contain considerable development (e.g., Elevenmile Creek, Perdido Bay, AL/FL, and development near lower Perdido Bay distributary systems; see Livingston 2001a). Some estuarine watersheds may still contain a tributary, or segment thereof, whose nutrient load represents a minimally impaired stream system because human development is minimal. The minimally impaired stream nutrient loading to head of tide may be used to estimate the nutrient load for the entire watershed if certain assumptions are met. Application of a watershed stream segment to estimate nutrient loading is based on seven assumptions, as described in Table 6-2, including atmospheric deposition of nitrogen as a constant.

Areal Load Approach to Identification of Reference Condition

The nutrient load is measured for the minimally disturbed subtributary or segment. State and national preserves, when available, typically offer appropriate conditions (Clark et al. 2000, Fulmer and Cooke 1990). A decision is required as to whether the geology is approximately homogeneous across the watershed to extrapolate from the reference tributary to the entire suite of tributaries. If not, then a

Table 6-2. Requisite assumptions for establishing watershed-based reference conditions

Assumptions	Description
Estuarine systems are usually unique rather than being readily divided into similar classes of estuaries.	There are some instances of coastal bays that can be classified together so that one or more of them may be designated as reference sites that collectively may comprise a reference condition against which other similar estuaries in a given area can be compared. However, most often each estuary must be addressed individually and the reference condition must be derived from data within that system.
The tidal factor and large volume shifts make regional subdivision of each overenriched estuary difficult.	In the instance of biological criteria development, it was possible to subdivide the estuary by salinity habitat regimes and thus reduce the portions of the estuary and its water column dynamics to a manageable level, especially for assessments of nonmobile benthic invertebrates. This is not viable with nutrient-related planktonic organisms and dissolved or suspended water column materials.
We can estimate "pristine" or natural loads from estimations of concentration-flow relationships, and therefore loading estimations in unaltered subestuarine watersheds. That loading can be extended to an entire estuary for reference condition development.	Loading estimations are an established practice in water resource management. The universal soil loss equation (USLE) is the best known example, and it has been extended to other unit loading estimations. This robust and rather straightforward concept estimates most of the nutrients that enter an estuary regardless of how effectively the system processes or assimilates nutrients.
Most of the measurable load to an estuary is tributary based, and atmospheric deposition is a constant over the system.	Admittedly, there is an atmospheric source of nitrogen to and from estuarine waters, but this is variable and, generally, is much less than runoff from streams and rivers. The effect is somewhat mitigated because the atmospheric deposition to the stream surfaces is incorporated in the loading estimation. Phosphorus and suspended material as well as algal responses to nutrients are most certainly more tributary related than atmospheric. Shoreline sheet runoff can be incorporated into the loading approach.
Loading from coastal marine waters is usually negligible compared with anthropogenic watershed loads to the estuary	While many estuarine ecologists are properly concerned with this aspect of estuarine dynamics, from the practical criteria development approach, a large portion of the marine load may be presumed to have originated in the previous outgoing estuarine tidal water. The remainder is to some extent a part of a natural process inherent to estuarine systems. This assumption may not prevail when estuaries enter deep, upwelling, oceanic waters.
The predevelopment nutrient loading rates expressed as yield per watershed land area are similar within a single geographic region (e.g., province, ecoregion, or subecoregion). Local regional uniformity of geography is assumed.	This is a reasonable expectation in the absence of extensive land development with attendant anthropogenic discharges and runoff. The geographic subdivisions of a natural landscape can be expected to be homogeneous by definition (i.e., similar soil type, topography, and vegetative cover).
Because the National Nutrient Criteria Program assumes that groundwater influence is a separate loading factor to surface water eutrophication, groundwater-dominated estuaries should be treated separately for the development of nutrient criteria.	The groundwater factor can be highly significant in some localities such as coastal Florida. This generalized approach does not address that factor, but Regional Technical Assistance Groups should be aware of the groundwater contribution and account for it in their estimations.

second minimally disturbed subtributary should be sought to represent a different geology; this logic should be applied if additional subwatersheds differ substantially. Nutrient loads are then extrapolated in a simple proportional manner from the reference tributaries to the entire tributary system within the watershed or subunit and the aggregate load is calculated (Figure 6-5). This load represents the "reference load" reflecting in situ reference conditions for the four primary indicators (TN, TP, chlorophyll *a*, and water clarity).

Extrapolation from a reference tributary can be augmented by application of geographic-based nutrient erosion and transport models. This nutrient load would become the target load for the downstream estuary or coastal waterbody. Consideration should be given to the representative nature of the freshwater flow over the average hydrograph. It is desirable and may be necessary to obtain a measure of the average multidecadal freshwater flows at the head of tide. Nutrient loads based on a drought period would not accurately represent conditions in terms of nutrient-based ecological impairments. Extremely high flows are important, but they are likely to fall outside of resource managers' capabilities to solve a nutrient problem. Therefore, this approach has its limits. Large fluvial streams do not necessarily transport the most upstream load to the lowest fluvial portion of the stream tributary; in-stream ecosystem processes modulate the load. In summary, a sequence of steps is outlined in the following box to complete the areal load watershed approach for reference condition determination.

Coastal Reference Conditions

Following is an example of a nutrient loading assessment from a very large watershed, the Mississippi River system. In 1997, the EPA Gulf of Mexico Program, through a Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, asked the White House Office of Science and Technology Policy to conduct a scientific assessment of the causes and consequences of Gulf hypoxia through its Committee on Environment and Natural Resources (CENR). The National Oceanic and Atmospheric Administration (NOAA) was asked to lead the assessment. The assessment included various computer-based modeling approaches to characterize the nutrient delivery to coastal Louisiana and Texas.

Nutrient load reduction within the various watersheds to meet resource management objectives is an analogue of the reference condition approach (see NOAA website for details: www.nos.noaa.gov/products/pubs_hypox.html). Two conditions are considered: coastal estuarine plumes and shelf waters.

Coastal Estuarine Plumes

The foregoing example for the Mississippi River watershed is a large-scale effort to assess nutrient conditions associated with a large coastal nutrient plume. For large estuarine watersheds with the potential to cause nutrient overenrichment on the continental shelf, it is important to extend the reference concept beyond the local area. In many cases, the concerns will be large enough to warrant development of a hydrodynamic model with coupled nutrient-phytoplankton growth kinetics (Chapter 9). Smaller estuarine plumes along the coast may be addressed through a well-designed research and monitoring plan with expert input on design features. An initial EPA-sponsored sampling effort in the Mid-Atlantic is currently underway to provide range-finding data to assist in development of a more comprehensive

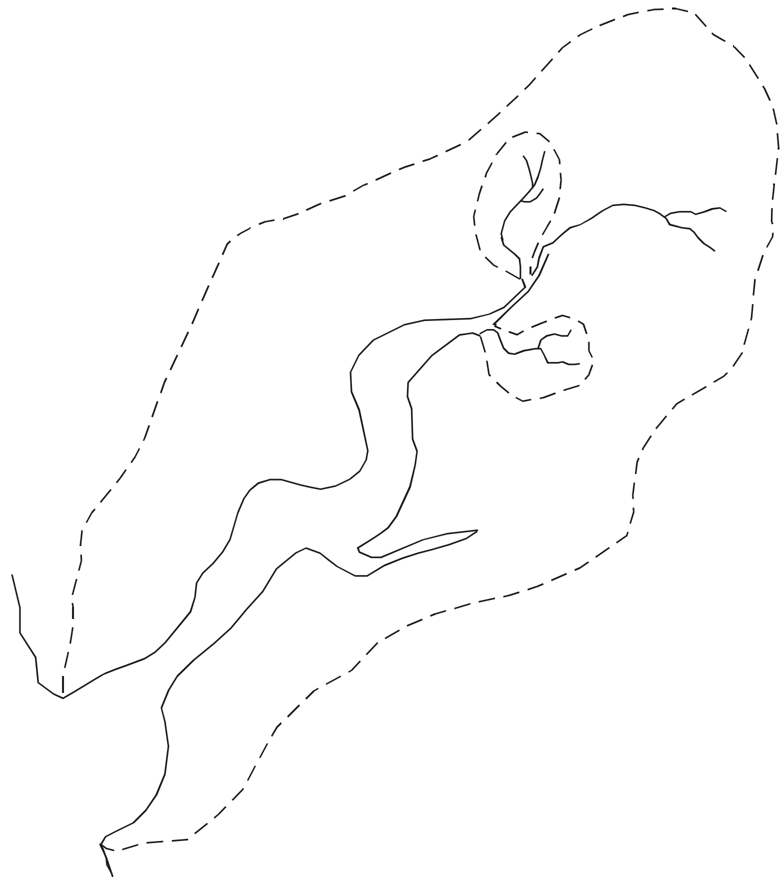

Figure 6-5. Areal load estimate approach to nutrient reference condition determination. The watershed is estimated to be approximately 368 square miles; the reference tributary streams representative of above head of tide systems in the watershed are approximately 20 square miles combined. The median load estimate at the mouths of the tributaries could therefore be multiplied by 18.4 to approximate a reference condition load for the river.

monitoring plan. The first step is to assess the shelf's mixing and dispersive capabilities to attenuate the negative effects of nutrient enrichment. Bloom development along physical discontinuities should be assessed. If dispersive mechanisms are large enough to attenuate phytoplankton blooms, then concerns are given a lower priority. In coastal estuarine plumes where the physical processes may not attenuate the nutrient enrichment effects to acceptable levels, then an appropriate level of research, monitoring, and modeling may be required to assess nutrient reduction from upstream nutrient sources as well as from seaward upwelling of nutrients and atmospheric deposition (Paerl and Whitall 1999, Vitousek et al. 1997, Howarth et al. 1996).

Open Coastal Water

The NRC (2000) publication (Chapter 6) recommends an index site approach for estuaries and coastal waters. The index site approach in particular has merit because the continental shelf is very extensive and too large for the Nation to conduct comprehensive studies of all sites potentially affected by nutrient enrichment. A priority-setting rationale should be based on a physical classification system that arrays coastal water masses relative to their susceptibility to nutrient enrichment, especially waters that are likely to respond similarly to nutrient enrichment. This would allow resource managers an opportunity to apply unique “dose-response” curves to a particular coastal water class. Because such a system likely is relatively crude, the best professional judgment of experts should augment physical classification systems. Because this approach has a significant research element, it would be appropriate to begin it in the competitive research forum with oversight provided by some mix of Federal, State, and university participation. Chapter 7 also discusses the roles of monitoring and modeling, which offer useful insights applicable to the coastal ocean. This manual recommends that the index site approach be given serious consideration, especially for coastal waters including estuarine plumes. The coastal ocean is large and oceanographically dynamic and complex. Thus, assessment of individual States’ contribution to nutrient overenrichment will in most cases require a Federal and States’ partnership. Even in large estuaries whose watersheds or tidal waters are shared by more than one State, a multi-State agreement is probably required (e.g., Chesapeake Bay Program and the New York and Connecticut TMDL Agreement).

The Coastal Research and Monitoring Strategy, an element of the Clean Water Action Plan, contains approaches that can help determine reference conditions. The strategy provides for a coordinated effort among Federal, State, and private agencies.   Clearly, an approach that coordinates use of aerial surveillance tools (e.g., satellite-based water quality sensors), data buoys, and ship-based measurements (especially ships of opportunity such as the North Carolina Albemarle Sound ferryboat monitoring program; H. Paerl, personal communication) within an index site to underpin a cause-and-effect framework is highly desirable.

On a provisional basis, any additional monitoring might include a stratified random approach (e.g., EMAP), because this provides an opportunity to address known ecological structure and functional processes and unbiased trend monitoring. It is important to continue monitoring that involves identified relationships. A challenge will be to design a program that can distinguish the effects of natural coastal processes (e.g., nutrient upwelling) from anthropogenic influences (atmospheric nitrogen deposition, fluxes of nutrient from estuaries, and possible expansion of mariculture activities).

One such investigation is the design presently being tested by the National Nutrient Criteria Program in the Mid-Atlantic Bight. This near-coastal marine nutrient sampling protocol is intended to identify inshore (within the 3-mile limit) reference sites based on land use and physical coastal characteristics together with comparisons to offshore nutrient water quality. A stratified-random approach is used, and the compiled data from reference sites establish the reference condition for that portion of the coastal marine waters. Riverine and estuarine plumes or other discharges can then be evaluated relative to this minimally impacted condition. Sampling is recommended for spring and summer conditions with multidepth collections. The technique is being tested in a variety of State waters. A description of the design and preliminary results is presented in Appendix H.

1. Identify major tributaries to the estuary. Classify similar tributaries by physical size and freshwater delivery and compare similar geological features of the subwatersheds for classification.

Three nutrient loadings can be estimated:

A. *Existing (actual current load)*. Existing load is estimated from the known tributary loads as measured at the mouth above head of tide and extrapolated to similar land units in the watershed, plus any shoreside runoff and discharges that directly enter the estuary, plus direct atmospheric deposition. This is the present status of the waterbody.

B. *Best existing load*. Find the **best existing** tributary(ies) or **best subtributary**(ies) in the region and calculate the nutrient yield. Extrapolate this value to the rest of the watershed tributaries or subtributaries as appropriate.

C. *"Pristine" or unimpacted load*. Identify a regional reference streams in an undisturbed watershed having little or no development, such as State or national preserves, that can be used to estimate areal yield for the region approximating an entire unimpaired, undeveloped condition as though there were no significant cultural impairments.

2. Find the tributary(ies) with the least impaired status and minimal disturbed lands contributing to nutrient loads.
These are B and C-type conditions from above. Each of these tributaries is monitored enough to establish a nutrient load. Seasonal and interannual variability should be assessed. Cross-sectional sampling at the head of tide is required because of nutrient flux variability. The USGS has established protocols for this type of stream monitoring. Where available, dams on rivers, if located near the head of tide (e.g., Conwingo Dam on the lower Susquehanna River, MD), make very desirable locations to measure nutrient loads.

3. Estimate the annual areal nutrient yield for TN and TP.
Extrapolate from the unimpaired or minimally impaired watersheds above head of tide to all other similar tributaries in the watershed and apply the load estimate to direct runoff portions of the tributary. Do this for each monthly or seasonal increment throughout the year. Do the same for each major tributary.

4. Extrapolate the nutrient yield to the entire watershed land area within the region.
For example, generate total best existing nutrient loading. This will, in many cases, represent the best attainable loads.

5. Repeat for other regions if the estuary watershed covers more than one major geological landform.
This is necessary to comply with the assumption that regional homogeneity within the watershed covers only part of the entire estuarine watershed.

6. Sum the nutrient yield for all tributaries within the estuarine watershed.
Do not factor in atmospheric nitrogen loads, as they were incorporated into the tributary loads. The atmospheric nitrogen loads may need to be reduced to achieve an acceptable nutrient condition in the estuary. The summation of tributary loads becomes the estuarine reference condition. Computer modeling may be required, especially in larger watersheds, because in very long tributaries some nutrients, especially phosphorus, may become embedded in the stream bottom, and some nitrogen and phosphorus is potentially lost on the scale of years to decades in the floodplain. The chlorophyll *a* concentrations in the estuary will need to be modeled from the reference nutrient loads and some measure of central tendency of freshwater inflows to the estuary. When a larger estuarine system may dominate the lower tributary estuarine hydrodynamics (e.g., the mainstem of the Chesapeake Bay dominates the hydrodynamics in the lower Patuxent River estuary), then as a minimum, a box modeling approach may be required to account for the two-estuarine interactions of the freshwater inflow and lower estuarine interactions (Hagy et al. 2000).

CHAPTER 7

Nutrient and Algal Criteria Development

Role of RTAGs
Classification
Descriptive Background Information
Elements of Nutrient Criteria
Hypothetical Examples of Nutrient Criteria Development Determinations
Evaluation of Proposed Criteria
Nutrient Criteria Interpretation Procedures
Criteria Modifications
EPA, State, or Tribe Responsibility Under the CWA
Implementation of Nutrient Criteria

7.1 INTRODUCTION

This chapter addresses the details of developing scientifically defensible quantitative criteria for nutrients, algae, and measures of water clarity (referred to hereafter as nutrient criteria). The chapter is divided into eight sections: (1) role of the RTAG; (2) classification; (3) descriptive background information; (4) approaches to criteria development; (5) evaluation of proposed criteria; (6) criteria modification; (7) EPA, State, and/or Tribal responsibilities under the Clean Water Act (CWA); and (8) implementation of nutrient criteria into water quality standards. The five elements of criteria development described in the Executive Summary and Chapter 1 are integrated herein, and additional information relevant to criteria development is provided.

As explained in Chapters 2 and 3, estuaries and coastal waters are especially complex ecosystems where it is often difficult to distinguish the effects of anthropogenic nutrient enrichment from natural variability primarily because of intervening physical processes and their interaction with biological (e.g., grazing; Cloern 1982) and chemical (e.g., flocculation and sedimentation) processes (Malone et al. 1996, NRC 2000). The individual nature of many of these ecosystems presents a particular challenge for criteria development. The ideal goal is to establish nutrient criteria that are protective during periods when estuaries and coastal ecosystems are most vulnerable to nutrient enrichment and that protect designated uses. It is important to understand that designated uses may be met but some nutrient-based impairment may have occurred. In such cases, it is also desirable to have restoration goals in mind whose objective is to restore the original ecological integrity, at least as represented by the reference conditions and criteria. This information helps determine if new designated uses are appropriate.

If a shallow estuary is dominated by point sources of nutrients, then low freshwater flow periods might be times of greatest vulnerability because of limited flushing. For a deep estuary under the same situation, weak density stratification may set up conditions where the algae spend considerable time below the euphotic zone and, hence, bloom development is minimized. Now, consider the situation where these two estuaries are dominated by nonpoint sources of nutrients. The low flow period would likely contribute lower nutrient loads to both types of estuaries and with weaker density stratification the algal bloom potential might be substantially lowered (see Chapter 2). It is also extremely difficult to establish nutrient criteria for episodic events, either hurricanes or drought periods. The reference condition and consistent reference site records, however, make it possible to adjust the criteria accordingly to address these intervals.

In this manual, the elements of criteria development are sequenced and emphasized somewhat differently from those described in the two published freshwater manuals. Because of the relatively high individual nature of estuaries and coastal waters, the role of the regional technical assistance groups (RTAGs) is enlarged and historical data become especially important in development of nutrient criteria. Figure 7-1 provides a visual perspective of the elements that should be integrated to arrive at a criterion.

7.2 ROLE OF REGIONAL TECHNICAL ASSISTANCE GROUPS

Expert evaluations are important throughout the criteria development process. The role of the RTAG in criteria development for estuaries and coastal waters has an added dimension over that applicable to lakes and reservoirs and rivers and streams. In the latter case, most of the data used in development of criteria recommendations resided in national electronic data sets (e.g., STORET) collected by State and Tribal agencies. The RTAGs helped review these data for duplication and outliers. They also encouraged States and authorized Tribes to submit additional data to STORET. Under these circumstances, EPA developed criteria recommendations based on frequency distributions (U.S. EPA 2000a,b). However, for most estuaries and coastal waters, the majority of relevant data have been collected by universities and other organizations (e.g., NOAA, Minerals Management Service, USGS); many of their data are not entered into STORET. It is expected that regional RTAGs will likely be considerably more knowledgeable about the data veracity and applicability for criteria development in their region. In addition, it is anticipated that the regional RTAGs will have the knowledge and access to other local scientific expertise to assist in the sampling design and collection of additional data. These considerations lead to the expectation that the RTAGs will have a larger role in the development of protective nutrient criteria for estuarine and coastal waters.

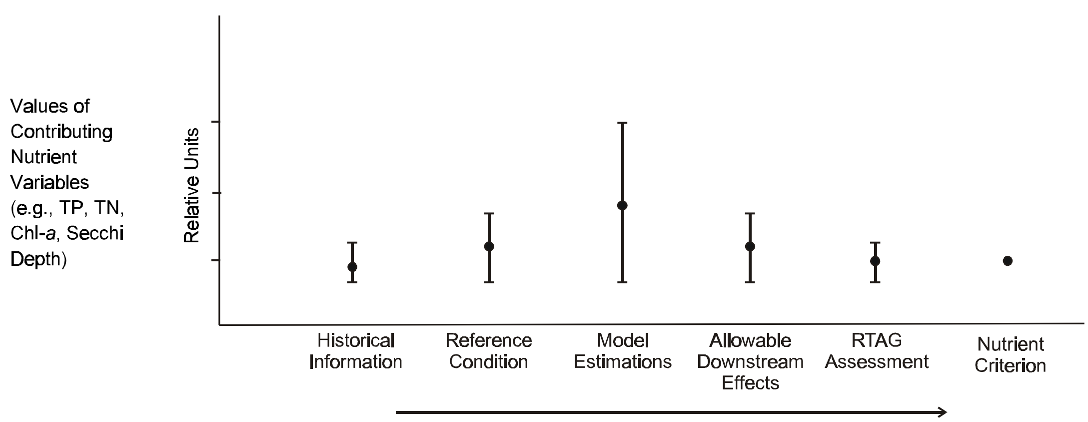

Figure 7-1. Generalized progression and relationship of the elements of a nutrient criterion.

Because of the tendency of estuaries and coastal waters to exhibit a high degree of individuality relative to nutrient susceptibility, little predictive success has been demonstrated in using published values of nutrient concentrations or nutrient impairments. However, experience with an individual estuary or coastal water system may demonstrate a general range of algal biomass accumulation that leads to hypoxia at a known level of enrichment. This type of assessment often requires several to many years of observation and measurement and is an example of incorporating the RTAG experience in the criteria development process.

EPA expects to continue to lead the effort to identify potential estuarine and coastal waters for the development of nutrient criteria and fund the overall data collection and analysis.

7.3 CLASSIFICATION

Classification is a pivotal step in the process of developing criteria. Physical classification of waterbodies for nutrient criteria development reduces variability in ambient measurements as the reference conditions and nutrient criteria represent and reflect a relatively similar natural state. Classification helps ensure that appropriate comparisons are made among comparable waterbodies so that the variables measured are influenced as little as possible by dissimilar inherent characteristics. This will facilitate appropriate application of criteria through their implementation.

In contrast to rivers and lakes, physical classification of estuarine and coastal waters is scale-sensitive (Giller et al. 1994) and may not be as predictable of nutrient enrichment effects or be as useful for generalizations about effects among estuarine systems. Classification nonetheless can provide improved understanding of the processes that contribute to ecosystem susceptibility and variability in the expression of nutrient effects. Classification may have valuable applicability at smaller physical scales within larger estuarine and coastal ecosystems (e.g., embayments; subestuaries and estuaries discharge plumes). Classification based on salinity gradients, circulation patterns, depth, and flushing within larger estuarine and coastal systems should also prove useful, especially in correlating the different biological communities at risk to nutrient overenrichment.

7.4 DESCRIPTIVE BACKGROUND INFORMATION

Estuarine Watershed Characterization

One of the keys to understanding nutrient enrichment problems in waterbodies is an environmental characterization of the watershed from a historical perspective. Such investigations may provide insights into the potential for confounding cause and effect relationships (e.g., nutrient, not herbicides as the primary cause of the SAV decline in Chesapeake Bay.) Historical changes in land use, land cover, and population demographics may correlate with increased anthropogenic-based nutrient enrichment and major changes in downstream water quality and biological community structure.

Farm acreage, crop types, and fertilizer application rates also provide useful information to assess the potential historical magnitude in nonpoint source nutrient loading to estuarine systems. Atmospheric plumes that result in nitrogen deposition should also be assessed as they may help explain an increased nitrogen load to coastal waters beyond that attributed to local land use activities. This also is a situation where reference sites, if available, will help the manager distinguish atmospheric from local anthropogenic causes of overenrichment. Conversely, increases in forest coverage and soil banking of agricultural lands may help explain potential decreases in nitrogen and phosphorus loading to estuaries and coastal waters.

Within Estuarine System Characterization

Changes within estuarine systems that influence basic hydrography should not be overlooked. For example, opening new passes, and later deepening them in several Gulf of Mexico estuaries during the early part of the 20th century (e.g., Perdido Bay, Alabama/Florida, and Choctahatchie Bay, Florida) apparently modified estuarine circulation resulting in strengthened density stratification leading to enhanced potential for hypoxia (Livingston 2001a). Also, major changes in freshwater supplies should be considered as a potential factor that can modify estuarine susceptibility to nutrient enrichment.

The use of marine sediment cores is another tool to assist in assessment of nutrient enrichment patterns in coastal waters (Brush 1984). These analyses are relatively expensive to perform but appear more frequently in the literature because of the numerous important insights they provide. They can provide estimates of sedimentation rates and initiation of anoxia, changes in algal community structure, initiation of the loss of SAV and other responses to nutrient enrichment. This temporal picture is important in setting approximate timelines when nutrient enrichment may have been a major cause of biological impairment. However, correlation does not necessarily equate to causality (Havens 1999).

It is also important to attempt to collect long-term (e.g., multiple decades) fisheries landings data as many stakeholders need to be appraised of whether such landings data are associated with nutrient enrichment. Such analyses often prove to be difficult because information on catch per unit effort that helps normalize for variable fishing pressure, is difficult to obtain. A change in fishery yield may be confounded by overfishing, as well as the role of increased or decreased primary productivity. Increased bottom water hypoxia related to overenrichment may explain the loss of benthic habitat for bottom-dwelling marine life (e.g., flounder and croaker and benthic infauna that serve as fish food).

Historical decreases in water column visibility from nutrient-driven algal blooms (phytoplankton and macroalgae) may explain reductions in water-borne recreation (e.g., swimming); however, human perceptional responses are often subjective and other factors such as user conflicts may also be involved. Reduced visibility may also be related more to inorganic suspended sediments then nutrients. So, both chlorophyll *a* records and total suspended sediment concentrations, preferably measured from the same local water mass, may be required to establish nutrient enrichment (Gallegos 1994). Such analyses need to be assessed with freshwater flow records because of potential co-linear effects that complicate interpretation of cause and effect relationships.

The above measurements performed as part of a general characterization may help relate nutrient enrichment effects and thresholds to existing and designated uses and identify any reference systems that are minimally impacted by nutrient pollution. Such information may enter directly into predictive water quality models or serve as indirect collaborative information contributing to a “weight of evidence” analysis.

7.5 ELEMENTS OF NUTRIENT CRITERIA

Reference Condition

Chapter 6, beginning with Table 6-1, describes several approaches that can be employed to determine the ambient minimally impacted nutrient condition of the water resource. The significance of this reference condition to nutrient criteria development cannot be overstated. It represents the determination of the existing, presently attainable nutrient water quality of the estuarine or coastal waters of concern.

The selection of the method for reference condition determination in this manual involves more options than previous guidance manuals because so many estuaries are reported to be unique and/or severely degraded, thus requiring an array of alternative approaches to approximate reference conditions in the absence of acceptable reference sites. In selecting from the different approaches, the resource manager should strive for the most direct measurement of the resource and with the least number of intermediate interpretative steps to a determination. Of equal importance in this process are in situ reference sites and supporting data showing the system response to the nutrient increases. The best of both these worlds is a set of reference sites documenting an optimal nutrient condition as well as response data confirming that system degradation occurs at levels beyond this measure, which also corresponds to the EPA regional reference condition for that area and class of waters. Failing this, the manager should seek the greatest approximation possible and a sufficient understanding of the divergence to be confident of the reference values determined.

Even though the reference condition is salient to nutrient criteria development, it should not be interpreted as the only necessary element. It should be interpreted in light of the historical condition of the resource and projections of its future potential.

Historical Information

Knowledge of antecedent conditions is particularly important in the case of estuarine waters, where causal relationships are often confounding and existing reference sites compromised. In such cases the historical data may not only qualify present information, they may, in fact, be the requisite reference condition demonstrating to the manager and RTAG not only previous nutrient quality, but least impacted conditions as well.

Models

In addition to computer modeling to help determine reference conditions, models may be used both to estimate nutrient loads and load reductions to achieve a targeted nutrient regime in the receiving estuarine and coastal waters systems. Various models are available (Chapter 9) that estimate nutrient erosion from different land uses, riverine transport of nutrients, and estuarine effects including hypoxia and chlorophyll *a* concentrations. These models are typically expensive to calibrate and verify, but in large estuaries and coastal ecosystems their application can be cost-effective when costly nutrient control decisions are involved. A coupled nutrient transport and hydrodynamically coupled algal nutrient uptake and growth model provides the ability to address “what if scenarios” of nutrient load versus level of biological impairments (see Chesapeake Bay Case Study and web site).

Statistical models can be used to help separate effects of nutrient loading from estuarine physical processes as determinants of increases in system response variables (e.g., chlorophyll *a* concentrations; Harding and Perry 1997). Effects of nutrient enrichment are inferred by a process of elimination when the suspected physical forcing functions are de-trended in the analyses and are inferred not to explain the variability. Box models, using salinity as a tracer of water masses, are useful in assessing net non-tidal physical circulation, hydraulic residence times, the effect of river flow on residence time versus seaward higher salinity processes, and extent that nutrient sources are conserved within an estuarine region or transported seaward (Hagy et al. 2000). Dettmann (in press) used a regression modeling technique to compare the degree of nitrogen export of a variety of estuarine systems. Boynton et al. (1996) used a box model to mass balance nitrogen and phosphorus in Chesapeake Bay and calculated net transport from the Bay of nitrogen and phosphorus, sedimentation, amount tied up in plants and the amount of nitrogen lost from the Bay through denitrification. Properly applied box and regression models are relatively inexpensive to construct and can provide useful information to the scientist and water quality manager.

Antidegradation Policy and Attention to Downstream Effects

A critical requirement for the use of reference conditions associated with nutrient criteria is the EPA antidegradation policy, which protects against incremental deterioration of waterbodies and reference conditions. An observed downward trend in the conditions of reference sites cannot be used to justify relaxing reference expectations, reference conditions, and the associated nutrient criteria. Once established, nutrient criteria should only be refined in a positive direction in response to improved conditions. Without antidegradation safeguards, even the establishment of reference conditions and nutrient criteria could still allow for continual deterioration of water quality.

To combat this, the States should implement an effective antidegradation policy that promotes continually improving conditions. As an example, Maine has an antidegradation policy that requires that waterbodies remain stable or improve in trophic state (Courtemanch et al. 1989, NALMS 1992). The RTAG should assume a comparable sense of antidegradation responsibility.

Estuaries that supply nutrients to relatively static coastal waters may require more stringent nutrient criteria, not only to protect estuarine designated uses (e.g., “fishable and swimmable” conditions), but the water quality of coastal shelf waters. At present, there are little data to assess whether U.S. estuaries are supplying nutrients to coastal shelf waters at levels that are causing widespread harm, except in the case of world-class rivers (e.g. Mississippi River Plume on LA/TX shelf). Locally, river-dominated estuaries with open passes to the coastal shelf supply nutrients at levels that may increase secondary productivity of valued fisheries (Sutcliffe et al. 1977), but the potential threshold for overenrichment effects in such cases is generally still poorly understood. If the RTAG determines that estuarine nutrient criteria may be expected to fall between the existing present nutrient concentrations or load and the reference condition determined from similar unimpaired systems or from a historical load and response relationship. It is then up to the States and Tribes to adopt the criteria into 303(c) water quality standards.

The RTAG

Assimilation of all of the above information is the responsibility of the RTAG when developing ecoregional nutrient criteria and when reviewing State or Tribal nutrient criteria as part of its role to assist EPA.

The RTAG should work with the States to develop a monitoring program that would evaluate the status of the reference systems, the possible future negative anthropogenic nutrient effects, and the condition of the estuary and coastal receiving water. A well-designed monitoring program should provide data to assess whether there is incremental deterioration of the subject waterbodies and reference conditions.

7.6 HYPOTHETICAL EXAMPLES OF NUTRIENT CRITERIA DEVELOPMENT DELIBERATIONS

To help illustrate the role and responsibility of the RTAGs, an abbreviated hypothetical illustration of nutrient criteria development follows for a river-dominated estuary that has a relatively deep channel with moderate density stratification, and well-developed seagrass meadows located in the shallow waters. The estuary is located in the northern Gulf of Mexico. The estuary often borders on both nitrogen and phosphorus limitation with nitrogen limitation occurring more frequently during the summer. The nitrogen sources have been and continue to be primarily nonpoint sources in headwater streams and point sources near the pass to the Gulf. Tidal action is minimal. The focus for this illustration is on nitrogen criteria.

Scenario

In the subject nitrogen-limited estuary no existing areas qualified as a reference condition and no meaningful analogs of the estuary were available to apply the spatial-based frequency/percentile approach to reference conditions applied to lakes (e.g., the 25/75 percentile approach; see Lakes and Reservoirs Nutrient Guidance Manual, U.S. EPA 2000a). The historical nutrient concentrations were plotted over time by classifying the summer estuarine salinity zones based on the 30-year average into tidal fresh and brackish (0-5 psu), mesohaline (5-18 psu) and polyhaline (18-30 psu). The concentrations were consistent with a calculated nutrient load and estuarine physical hydrodynamic model. Additional analysis demonstrated that the system was most vulnerable to average summer freshwater inflows. Flushing in the estuary was determined to be on the order of one month on average during the summer under average freshwater flow conditions so the physical potential was high for phytoplankton bloom buildup.

A 50 µM TN reference condition was determined by plotting TN concentrations for the mesohaline zone from 1970 to 2000. Because nutrient data were few for the estuary prior to 1970 and most of the data were collected for the mesohaline zone, a decision was made to select the 1970 summer average TN value as the reference condition with application for the mesohaline zone. Only an occasional bottom channel hypoxia occurred in the summer of 1970 and seagrass meadows were growing to a depth of 1 to 2 meters. By 1980, the average summer TN concentrations had increased to 60 µM when it became apparent from field monitoring that major loss of seagrass acreage was documented, hypoxia volumes had doubled, and hypoxia-laden water had consistently reached the deep-channel/shallow water shelf break. By 1980, seagrasses grew on average only to 0.5 meter depth. Some sediment core evidence indicated that seagrasses in 1950 grew at a depth of 2 to 3 meters but the nutrient concentrations and loading information were much weaker than for the 1970-2000 period. The reference condition of 50 µM TN suggests a gradual rise upward in ambient TN, from an estimated average 20 µM TN 100 years ago, with a projected further upward trend indicated by use of demographic, land use, and hydrological models. It was determined that a significant loss of ecological integrity had likely occurred prior to 1950. The RTAG, therefore, concluded that setting a criterion any higher than the present reference condition would eventually lead to an unacceptable trend in water quality degradation due to expected development increases for this part of the estuary. If the model projections are accurate, any increased load implied by raising the criteria above reference levels will hasten nutrient overenrichment problems. Some on the RTAG argued that the criterion should be set at 54 µM TN but the final consensus was to be somewhat conservative. The RTAG therefore concludes that it will be prudent to set the criterion at 50 µM TN (see Figure 7-2).

7.7 EVALUATION OF PROPOSED CRITERIA

The RTAG will provide expert assessment of proposed criteria and assure that criteria protect designated uses. Criteria will need to be verified in many cases by comparing criteria that apply across State and Tribal borders. In addition, attention will need to be paid to downstream effects and designated uses,

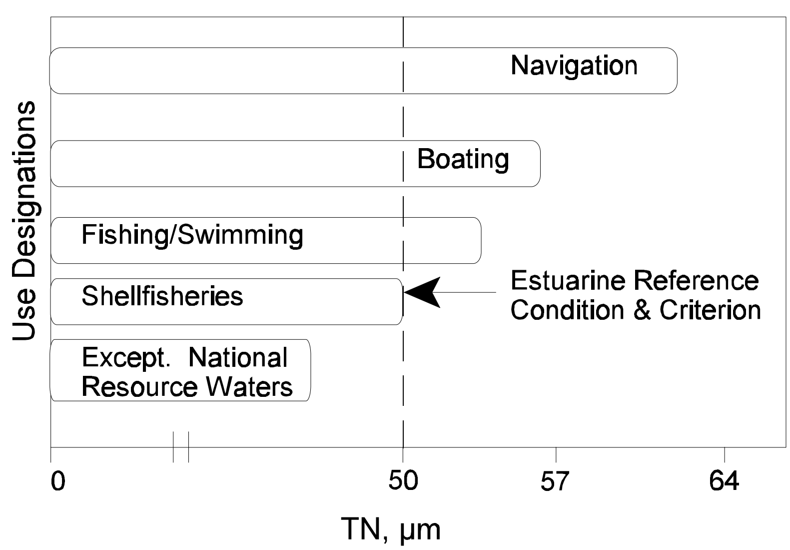

Figure 7-2. Hypotetical illustration of developing a TN criterion in an estuary.

especially in large estuaries that are shared by two or more States or Tribes. Criteria recommended by the RTAG can be adopted by the State or Tribe and approved by EPA if evidence is presented that assures no adverse effects will result downstream (e.g., criteria developed for tidal freshwaters may not be stringent enough to protect uses located at higher salinities.) In estuaries, a consideration, not typical of streams and lakes, is one where the life cycle of anadromous and semi-anadromous fish species must be considered as well as marine spawners that utilize estuaries as nursery areas. The RTAG may need to consult with neighboring Regional RTAGs in regions where estuaries are shared.

At present, EPA's Office of Water is developing a policy to address effects from nutrient transport that causes water quality problems at downstream estuarine sites including river systems that deliver nutrients from far inland to coastal tidal systems. If downstream designated uses are not protected by a proposed criterion, then the river or stream criterion must be modified accordingly.

Guidance for Interpreting and Applying Criteria

A critical step in the criteria development process is to assess how realistically criteria can be implemented into standards that are accepted by the public. It should be realized that today's designated uses are not those that would be applicable in many estuaries at the turn of the century or in some cases even several decades ago. Many estuaries have lost important fisheries that may not be easily recovered if at all. For example, sturgeon are rare in many estuaries today when they were abundant decades ago in several east coast estuaries. It is doubtful that the nutrient relationship for sturgeon growth and survival is adequately known except for obvious factors such as hypoxia. The RTAG should make some judgements about designated uses as exemplified by the sturgeon example that significantly improves nutrient-based degraded water quality in terms of "fishable and swimmable" but maintains an important degree of realism.

Do the Criteria Protect Designated Uses?

Section 303(c) of the CWA as amended (Public Law 92-500 [1972], 33 U.S.C. 1251, et seq.) requires all States and authorized Tribes to establish designated uses for their waters. EPA's interpretation of the CWA requires that wherever attainable, standards should provide for protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the water (section 101(a)). Note: this is the secondary goal of the Act; the primary goal being the protection and restoration of the physical, chemical, and biological integrity of the Nation's waters, and zero discharge of pollution. Other uses identified in the Act include industrial, agricultural, and public water supply. However, no waters may be designated to be used as repositories for pollutants (see 40 CFR 131.10(a)). Each waterbody must have criteria that protect and maintain the designated use of that water.

Also discussed below are general guidelines for developing criteria to protect selected designated uses. The values included here are *not* intended to represent proposed EPA or State estuarine and coastal waters nutrient criteria. Rather, they are simply guidance and illustrate ranges of parameters associated with the impairment of some designated uses in some States and Tribes. Criteria to protect these uses should be developed on a site-specific basis when the individual nature of the estuary or coastal waters require such specificity.

Outstanding National Resource Waters

Some estuarine and coastal waters of the State may require special criteria based on unique characteristics of that waterbody. Such characteristics might include undisturbed or unique fjords or subestuaries or stretches of coastline that are markedly different from other coastal waters in the State. Some areas may include threatened or endangered species that need to be protected. Such waters are the very best of the reference set and are most in need of protection by rigid State and Tribal antidegradation policies and procedures.

Aquatic Life Uses

Aquatic life uses, including fisheries and shellfisheries, are heavily dependent on the initial high quality condition of the resource. Species will change as a function of trophic state, and it may be difficult to defend why one species is necessarily "better" than another. The use of reference areas and their accompanying biota is one measure that can be used to predict the species that should be expected in a region.

Fisheries

Developing criteria to protect a specific fishery may be somewhat difficult because in open estuarine and coastal waters fish species shift with seasonal migrations and salinity changes. However, basic response variables such as available DO and turbidity can be incorporated to protect all seasonal fish and crustacean communities and resident molluscan populations. Consultation with fisheries managers, the recreational public, and commercial fishermen should help resolve any issues of targeted species management through nutrient abatement.

Although our knowledge of the dynamics of change in the biota as a function of eutrophication requires further development, there is sufficient evidence to conclude that eutrophication will bring species changes. If an area has an existing aquatic life use, then that use must be maintained. (See 40 CFR §131.12(a) (1).) Eutrophication will cause some species to change in relative abundance and cause others to disappear; therefore, nutrient enrichment may be incompatible with the maintenance of a specific biota. The ultimate extension of this concept is in the use classification of outstanding natural resource waters.

Recreation

Swimming/Primary Contact Recreation

Criteria to protect a contact recreation use may be associated with the occurrence (or appearance) of certain phenomena that affect certain types of recreation. For example, in general, swimmers will not be affected by the trophic state of the estuary, but resulting changes in transparency or change in species may be important. Dense planktonic or macrophyte growth may inhibit swimming opportunities and help promote seasonal densities of sea nettles or "jelly fish." Excess nutrients feed not only nuisance algae growth, but potentially health-endangering bacteria, especially when human or animal waste may be involved such as when sewerage discharges are in the general area and may impact swimming when wind and tide coordinate. This risk is not unusual for coastal ocean beaches where development promotes sewerage expansion and offshore discharges.

Boating and Secondary Contact Recreation.
It might be expected that the transparency of the water or the presence of algal scums would not deter boating, unless water skiing were involved. However, boating may be affected by the presence of dense inshore beds of tall or floating macrophytes.

Restoration Goals

As described in the introduction to this chapter, designated uses may be protected but some nutrient-based ecological degradation already may have occurred. The public deserves to know what nutrient conditions existed before anthropogenic nutrient enrichment initiated a shift from the natural nutrient regime toward conditions of nutrient impairment within the limits of scientific knowledge or reasonable scientific inference.

Sampling for Comparison to Criteria

Once criteria have been selected for each indicator variable (e.g., as a minimum, TN, TP, chl *a*, or macroalgal biomass as AFDW and a measure of water clarity associated with chl *a* and, where appropriate, the addition of dissolved oxygen), States and Tribes will want to develop implementation procedures to assess the estuarine and coastal water with the criteria. Sampling to evaluate attainment with criteria and adopted standards should be compatible with the procedures to establish the criteria in the first place. If the criterion was developed for a particular season, then sampling should be compatible with that season. In some cases, it is plausible that nutrient concentrations will not correlate predictably with response variables because of estuarine hydrodynamics. In such cases, the useful relationship should be directed toward nutrient loading. Many published estuarine nutrient relationships are based on nutrient load, often normalized to estuarine surface area, not nutrient concentrations. In such cases, the sampling of load relative to response variable should be scientifically based on the appropriate season with consideration for appropriate time lags (see Chapter 2). In many cases, relationships will need to be sought through application of a computer model (see Chapter 9).

Questions will arise about the size of area, depth of sample, frequency, and duration of any exceedances. These are difficult questions but the RTAG must be prepared to address them. It is expected that scientific judgment will be required in numerous cases that pushes the state of the science and in some cases it may be necessary to make risk management decisions extend beyond the current state of science. Some illustrative examples are provided but should not necessarily be interpreted literally. If an area is small and does not limit life cycle completion of important species (e.g., deepwater hypoxia that may serve as a bottleneck to estuarine species migration), then some tolerance is accepted. However, if the duration and magnitude of noncompliance of a criterion lasts long enough to affect the distribution and abundance or recruitment of an important species or a key food web component at a designated level (e.g., 15% reduction in the population of a harvestable size class is estimated based on the best available judgment), then the criterion needs to be adjusted. In some cases, empirical or computer models will be required to address many of the more complex relationships.

The question arises, how many replicate samples are needed to obtain an acceptable precision of data in order to detect differences between sites and changes over time? This depends on the nature of the variability in the variable of interest. Several approaches are available. However, this question involves both statistical and practical considerations (e.g., cost). General experience suggests that field water quality sampling will often vary by 20%. With this “rule of thumb,” it may not be cost-effective to try to achieve a lower percent difference. Eckblad (1991) provides some guidance on statistical considerations in sampling power. The Kendall test with Sen slope estimate (Hirsch et al. 1982) allows the determination of the number of replicate samples needed to detect a certain percent change in annual means of a variable or a certain percent trend over a period such as 10 years (see Rivers and Streams Nutrient Guidance Manual (U.S. EPA 2000b, Appendix A).

7.8 NUTRIENT CRITERIA INTERPRETATION PROCEDURES

However done, a State’s or Tribe’s nutrient criteria should include a procedural protocol to implement the newly adopted nutrient criteria. The criteria and procedures should be reviewed by the RTAGs for concurrence and are subject to further EPA review and approval if submitted as part of State or Tribal standards.

The initial criteria variables include two causal variables (TN and TP) and two response variables (algal biomas, e.g., chlorophyll *a* for phytoplankton and AFDW for macroalgae) and water clarity (e.g., Secchi depth), and where hypoxia occurs, dissolved oxygen may be added as a third response variable. Failure to meet either of the causal criteria should be sufficient to indicate a criteria “excursion,” and usually the biological response, as measured by chlorophyll *a* and Secchi depth, will follow this nutrient trend. However, if the causal criteria are met but some combination of response criteria are not met, then there should be some means of determining if the waters in question meet the nutrient criteria. Two suggested approaches are described below.

Decisionmaking Protocol

One option is to establish a decisionmaking procedure equation all of the criteria. Such a rule might state: “Both TN and TP causal nutrient criteria must be met, and a least three out of five response criteria (e.g., water clarity, algal biomass as chlorophyll *a* or macroalgal biomass as AFDW, DO, seagrass or SAV biomass, and phytoplankton species composition) must be met for three out of four sampling events during the June through August survey period over 2 consecutive calendar years of sampling. No sampling events may be less than 3 weeks apart [to avoid clustering sampling activities near a particular flow condition or runoff event], and flow and tidal conditions must be recorded as well so that watershed base flow and runoff events are evident and can be factored into the data assessment process.”

Multivariable Enrichment Index

The second option is to establish an index that accomplishes the same result by inserting the data into an equation that relates the multiple variables in a nondimensional comprehensive score much the same way an index of biotic integrity (Karr 1981) does. An example of an enrichment index approach is presented in Table 7-1.

Table 7-1. Example of an enrichment index using the middle portion of a hypothetical estuary

Variable	Criterion	Hypothetical estuary or salinity zone of estuary	
		Median measured value	Enrichment Index (EI) score*
Causal variables			
Total P (mg/L)	≤2.0	2.5	4
Total N (mg/L)	≤68	70	5
Primary response variables			
Secchi depth (M)	≥1.0	0.6	3
Chlorophyll *a* (mg/L)	≤60	75	3
Secondary response variables			
Dissolved oxygen (mg/L in hypoliminion)	≥6.0	3.5	4
Enrichment Index Value: = 19**			

* Each of the eight variables receives an EI score. The scoring procedure is: 0 = meets criterion; 2 = fails to meet criterion by 10%; 3 = fails to meet criterion by 25%; 5 = fails to meet criterion by 50% or more.
** Enrichment Index Value is the sum of the EI scores. The maximum i.e. worst score achievable is 25.

If necessary, the scoring process can be weighted by seasons. Thus, different emphasis can be given to the results of winter surveys as compared with summer surveys, and year-round work can be conducted if necessary or desired. For example, greater weight perhaps by a factor of 2 could be given to the primary response variables in winter for north temperate waters because these variables would normally be expected to be improved at this time of year. Similarly, the criteria for TP and TN might both be changed to lower concentration for winter because less runoff or fewer fertilizer applications are expected in the watershed. In the example, the estuary or region thereof fails anyway because it failed the criterion for either TP or TN (in fact it failed both). With a score of 19 out of a possible 25, it is also a prime candidate for extensive remediation management.

Such enrichment index scores are not intended at this time to be surrogate nutrient criteria. They may, however, serve as a “translator” to implement multiparameter criteria. However, like biological criteria index scores such as the Index of Biotic Integrity, the enrichment index may be a useful assessment tool relating several parameters. This helps the resource manager plan the distribution of effort and funds over the entire estuarine or coastal resource base in one procedure.

Frequency and Duration

Frequency and duration are important concerns when evaluating any water with respect to meeting criteria. This is a difficult process at this initial phase of the program because the data sources for criteria development are presently so diverse. In general, however, the method of data gathering for compliance should be as near as possible to that used to establish the criteria, especially keeping in mind tidal phase and salinity. Once consistency is established, excursions from the criteria based on frequency and duration can be evaluated whether based on a decision rule or a multivariable index.

Frequency of "excursion" from a criterion is a decision that can be best established by the State or Tribe on the basis of their knowledge of the local water resources. An excursion that occurs less than 10 percent of the times when sampling is conducted (at regularly spaced or random intervals) may be considered acceptable. Duration of the excursion may be stipulated as a set period of time (e.g., 2 weeks, or as to not exist over more than two consecutive sampling intervals, whichever is the lesser period). The State or Tribe in consultation with EPA will need to specifically define these terms as appropriate to the region and should also determine the combination of these factors that constitutes an "excursion."

7.9 CRITERIA MODIFICATIONS

Some situations may require site-specific criteria because of unique environmental conditions. In such situations, the general criterion is a starting point and it must be modified to protect designated uses in a unique situation. Such criteria can be adopted into State or Tribal water quality standards and reviewed by EPA.

7.10 EPA, STATE, OR TRIBE RESPONSIBILITY UNDER THE CLEAN WATER ACT

The Clean Water Act as amended (Pub. L. 92-500 (1972), 33 U.S.C. 1251, et seq.) requires all States to establish designated uses for their waters (Section 303(c)). Designated uses are set by the State. EPA's interpretation of the CWA requires that wherever attainable, standards should provide for the protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the water (Section 101(a)). Other uses identified in the act include industrial, agricultural, and public water supply. However, no waters may be designated for use as repositories for pollutants (40 CFR 131.10 [a]). Each waterbody must have criteria or measures of appropriate water quality that protect and maintain the designated use of that water. It is recommended that the EPA nutrient guidance be followed. However, States and Tribes may follow other guidance to adopt water quality criteria as long as the criteria are based on scientifically sound methods and protect designated uses.

7.11 IMPLEMENTATION OF NUTRIENT CRITERIA INTO WATER QUALITY STANDARDS

Nutrient criteria adopted into water quality standards by States and Tribes are submitted to EPA for review and approval (see Section 40 CFR 131). EPA reviews the criteria (40 CFR 131.5) for consistency with the requirements of the CWA and 40 CFR 131.5, which requires that water quality criteria be sufficient to protect the designated use (40 CFR 131.6 (c) and 40 CFR 131.11). The procedures for State/Tribal review and revision of water quality standards, EPA review and approval of water quality standards, and EPA promulgation of water quality standards (upon disapproval of State/Tribal water quality standards) are found at 40 CFR 131.20-22. The Water Quality Standards Handbook (U.S. EPA 1994) provides guidance for implementation of these regulations.

CHAPTER 8

Using Nutrient Criteria To Protect Water Quality

Managing Point Source Pollution
Managing Nonpoint Source Pollution
Comprehensive Procedure for Nutrient Management
Resources

This chapter provides an introduction to the applications of nutrient criteria. Chapter 1 described the ways in which nutrient criteria are used to (1) identify problems, (2) develop management plan, (3) assess regulations, (4) evaluate projects, and (5) determine the status and trend of the water resource. In this applications chapter, some of these are discussed in further detail. Section 8.1 addresses the management of point source pollution, in the context of standards development, National Pollutant Discharge Elimination System (NPDES) permits, and total maximum daily loads (TMDLs). Section 8.2 focuses exclusively on nonpoint source management programs. Although some material is not directly related to estuarine and coastal marine resources, it is included here because the coastal waters are the ultimate recipients of all drainage, both coastal and inland, and the information may be useful to a manager addressing various sources within a watershed. Coastal waters may, of course, be waters of the United States (see 40 CFR 122.2) and may thus be subject to the requirements of the Clean Water Act. Section 8.3 sets out a comprehensive planning, application, and evaluation procedure for estuarine and coastal marine nutrient quality management, and Section 8.4 lists publications on coastal/estuarine and watershed management and protection.

8.1 MANAGING POINT SOURCE POLLUTION

The term "point source" means any discernible, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged (CWA § 502(14)). This term does not include agricultural storm water discharges and return flows from irrigated agriculture. This section describes some of the programs relevant to point source discharges into rivers and streams.

The Clean Water Act and Water Quality Standards

The goals of the CWA are to achieve, wherever attainable, water quality that provides for protection and propagation of fish, shellfish, and wildlife and recreation in and on the water. The CWA further specifies that States adopt, and EPA approve, water quality standards consisting of designated uses, criteria to project those uses, and an antidegradation policy (CWA section 303(c)). The criteria must be based on a sound scientific rationale and must contain sufficient parameters or constituents to protect the designated uses (40 CFR § 131.11(a)). For waters with multiple use designations, criteria must support the most sensitive use (Id.). Finally, in designating uses and establishing water quality criteria, States must ensure attainment of standards in downstream waters (40 CFR § 131.10(b)). With regard to nutrient criteria, Section 304(a) of the CWA directs EPA to develop and publish criteria that reflect the latest scientific knowledge of the effects of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication for varying types of receiving waters. In establishing water quality criteria, States should establish numeric values to protect designated uses based on EPA's Section 304(a) criteria guidance, modifications of the guidance recommendations reflecting site-specific conditions, or criteria based on other scientifically defensible methods (40 CFR § 131.11(b)(1)).

As illustrated in Figure 8-1, States adopt water quality standards for waters of the United States that comprise designated uses, criteria to protect those uses, and an antidegradation policy to protect existing water quality. Additionally, States develop implementation procedures to describe how the water quality standards will be applied. Once water quality standards are adopted and approved, they become the basis for legally enforceable NPDES permit limitations and a variety of assessment activities under the Clean Water Act.

Protecting Designated Uses

It has been amply demonstrated that nutrients are a major contributor to use impairment in waters of the United States. Because States are required to designate uses in consideration of the goals of the CWA and adopt criteria that contain sufficient parameters to protect designated uses, and because it is EPA's responsibility to make related recommendations, the Agency is developing and publishing Section 304(a) criteria for nutrients that provide for protection and propagation of fish, shellfish, and wildlife and recreation in and on the water.

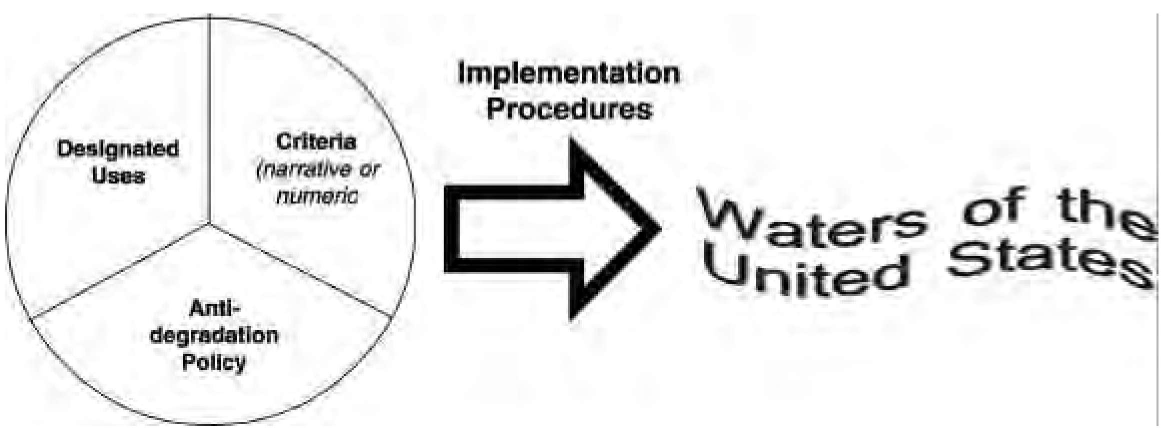

Figure 8-1. Components of water quality standards.

EPA's Section 304(a) criteria for nutrients are issued on the basis of ecoregion and waterbody type. This approach to nutrient criteria development provides a sound, scientifically defensible approach that accounts for the characteristics of different types and locations of waterbodies. EPA's ecoregional nutrient criteria are intended to represent enrichment conditions of surface waters minimally affected by human development. These criteria may be developed and further refined on the basis of the five elements described in this technical guidance manual.

Water quality criteria incorporating minimally affected (i.e., reference) conditions should provide for protection and propagation of aquatic life and recreation and reflect conditions that will not adversely affect the biological community. The parameters addressed in the Ecoregional Nutrient Criteria Documents are total phosphorus, total nitrogen, chlorophyll *a*, and turbidity (e.g., Secchi depth for lakes; turbidity for rivers and streams). These are the parameters that EPA considers important in nutrient assessment because the first two (nitrogen and phosphorus) are the main causal agents of enrichment, whereas the two response variables (chlorophyll *a* and turbidity) are indicators of system overenrichment for most surface waters.

Maintaining Existing Water Quality

Antidegradation

State and Tribal water quality standards include an antidegradation policy and methods through which the State or Tribe implements the policy. An antidegradation policy is required in State water quality standards to protect existing water quality. At a minimum, States must maintain and protect the quality of waters to support existing uses Antidegradation implementation procedures address the measures used by States and Tribes to ensure that permits and control programs meet water quality standards and antidegradation requirements. The water quality standards regulation sets out a three-tiered antidegradation approach for the protection of water quality (40 CFR § 131.12).

Tier 1

Maintains and protects existing uses and the water quality necessary to protect these uses (40 CFR 131.12[a][1]). An existing use can be established by demonstrating that fishing, swimming, or other uses have actually occurred since November 28, 1975, or that the water quality is suitable to allow such uses to occur, unless there are physical problems, such as substrate or flow, that prevent the use from being attained (U.S. EPA 1994).

Tier 2

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

Tier 3

Outstanding national resource waters (ONRWs) are provided the highest level of protection under the antidegradation policy. The policy provides for protection of water quality in high-quality waters that constitute an ONRW by prohibiting the lowering of water quality. ONRWs are often regarded as highest quality waters of the United States: That is clearly the thrust of 131.12(a)(3). However, ONRW designation also offers special protection for waters of "exceptional ecological significance." These are waterbodies that are important, unique, or sensitive ecologically, but whose water quality, as measured by the traditional parameters such as dissolved oxygen or pH, may not be particularly high or whose characteristics cannot be adequately described by these parameters (such as wetlands).

The regulation requires water quality to be maintained and protected in ONRWs. EPA interprets this provision to mean no new or increased discharges to ONRWs and no new or increased discharge to tributaries to ONRWs that would result in lower water quality in the ONRWs. The only exception to this prohibition, as discussed in the preamble to the Water Quality Standards Regulation (48 FR 51402), permits States to allow some limited activities that result in temporary and short-term changes in the water quality of ONRW. Such activities must not permanently degrade water quality or result in water quality lower than that necessary to protect the existing uses in the ONRW. It is difficult to give an exact definition of "temporary" and "short-term" because of the variety of activities that might be considered. However, in rather broad terms, EPA's view of temporary is weeks and months, not years. The intent of EPA's provision clearly is to limit water quality degradation to the shortest possible time. If a construction activity is involved, for example, temporary is defined as the length of time necessary to construct the facility and make it operational. During any period of time when, after opportunity for public participation in the decision, the State allows temporary degradation, all practical means of minimizing such degradation shall be implemented. Examples of situations in which flexibility is appropriate can be found in the Water Quality Standards Handbook (U.S. EPA 1994).

General Policies

The water quality standards regulation allows States and Tribes to include implementation in their standards policies and provisions, such as mixing zones, variances, and low-flow exemptions (40 CFR § 131.13). Such policies are subject to EPA review and approval. These policies and provisions should be specified in the State’s or Tribe's water quality standards document. The rationale and supporting documentation should be submitted to EPA for review during the water quality standards review and approval process.

Mixing Zones

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

Low-Flow Provisions

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

Water Quality Standards Variances

Variance procedures involve the same substantive and procedural requirements as removing a designated use (see 40 CFR 131.10 (g)), but unlike use removal, variances are both discharger and pollutant specific, are time-limited, and do not forego the currently designated use.

A variance should be used instead of removal of a use where the State believes the standard can ultimately be attained. By maintaining the standard rather than changing it, the State will assure that further progress is made in improving water quality and attaining the standard. With a variance, NPDES permits may be written such that reasonable progress is made toward attaining the standards without violating section 402(a)(1) of the Act, which requires that NPDES permits must meet the applicable water quality standards.

State variance procedures, as part of State water quality standards, must be consistent with the substantive requirements of 40 CFR 131. EPA has approved State-adopted variances in the past and will continue to do so if:

- Each individual variance is included as part of the water quality standard

- The State demonstrates that meeting the standard is unattainable based on one or more of the grounds outlined in 40 CFR 131.10(g) for removing a designated use

- The justification submitted by the State includes documentation that treatment more advanced than that required by sections 303(c)(2)(A) and (B) has been carefully considered, and that alternative effluent control strategies have been evaluated

- The more stringent State criterion is maintained and is binding upon all other dischargers on the stream or stream segment

- The discharger who is given a variance for one particular constituent is required to meet the applicable criteria for other constituents

- The variance is granted for a specific period of time and must be rejustified upon expiration but at least every 3 years (Note: the 3-year limit is derived from the triennial review requirements of section 303(c) of the Act.)

- The discharger either must meet the standard upon the expiration of this time period or must make a new demonstration of "unattainability"

- Reasonable progress is being made toward meeting the standards

- The variance was subjected to public notice, opportunity for comment, and public hearing. (See section 303(c)(1) and 40 CFR 131.20.) The public notice should contain a clear description of the impact of the variance on achieving water quality standards in the affected stream segment.

Providing Flexibility in Implementation

Abundant flexibility is built into the criteria-setting process and water quality standards regulations to allow States to (1) develop their own criteria to protect specific uses or reflect local conditions, (2) use different techniques to develop criteria as long as they are protective and scientifically defensible, and (3) conduct use attainability studies and refine their use designations.

States also have the flexibility to adopt numeric criteria to protect designated uses or adopt methods and procedures that "translate" narrative criteria into numeric values. Narrative criteria statements, often referred to as "general criteria" in States' standards regulations, usually take the form of a description of desired water quality condition or a preclusion of certain types of pollution or undesirable conditions (i.e., the "free from" provisions). Narrative criteria are considered critical backstops for designated use protection and are a powerful means of achieving desired water quality if they are interpreted in a clear and consistent manner. In water quality standards parlance, a "translator" is a process, methodology, or guide that States or Tribes use to quantitatively interpret narrative criteria statements. Translators may consist of biological assessment methods (e.g., field measures of the biological community), biological monitoring methods (e.g., laboratory toxicity tests), models or formulas that use input of site-specific information/data, or other scientifically defensible methods. Translators are particularly useful in describing water quality conditions that require a greater degree of sophistication to assess than typically can be expressed by numerical criteria that apply broadly to all waters with a given use designation. The translator may be either directly incorporated into State or Tribal water quality standards or incorporated by reference. In either case, specific limits or values for a measurable pollutant derived using a translator that interpret a narrative criterion statement should be attached to the State or Tribal regulations to ensure public review, as would be required of any site-specific numerical criterion.

States have the flexibility under current law to adopt appropriate nutrient criteria. If a State determines that its water quality criteria cannot be met, that State may consider refining use designation, adopting site-specific criteria, or issuing a variance to ensure that the appropriate uses and criteria to protect the uses are established. For example, if a regulated source faces expensive treatment to comply with a new or revised requirement, the State or Tribe can authorize a variance, based on a justification using one of the six factors at 131.10(g), to allow time for the discharge to come into compliance with a permit limit for the criteria and/or weigh options on treatment technologies, use reclassification, or site-specific development criteria as appropriate.

Water quality criteria guidance published by EPA under Section 304(a) of the CWA, such as for nutrients, serve as primary sources of information to States and Tribes as they develop numeric criteria as part of their water quality standards. Under the CWA and EPA's implementing regulations, States and Tribes also may use other information, including local water quality conditions, as they develop standards. Typically, EPA uses its own water quality criteria guidance as the principal basis for proposing and promulgating a replacement water quality standard when a State or Tribe fails to adopt an acceptable standard. In doing so, EPA commits to a process that includes public review and comment. EPA will solicit information from the public to determine if any such proposed Federal nutrient criteria for State waters are sufficiently protective of uses. This public process will help ensure that any promulgated Federal water quality standards are appropriately protective.

NPDES Permits

The CWA requires wastewater dischargers to have a permit establishing effluent limits on pollutant discharges. The regulations at 40 CFR 122.41 et seq. require these permits to specify monitoring and reporting requirements. More than 200,000 sources are regulated by the NPDES permits nationwide. These permits regulate household and industrial wastes that are collected in sewers and treated at municipal wastewater treatment plants. Permits also regulate industrial point sources and concentrated animal feeding operations that discharge into other wastewater collection systems or that have the potential to discharge directly into receiving waters. Permits regulate discharges with the goals of protecting public health and aquatic life and ensuring that every facility treats wastewater. Typical pollutants regulated by NPDES are "conventional pollutants" such as fecal coliforms or oil and grease from the sanitary wastes of households, businesses, and industries; and "toxic pollutants" including pesticides, solvents, polychlorinated biphenyls (PCBs), dioxins, and heavy metals that are particularly harmful to animal or plant life. "Nonconventional pollutants" are any additional substances that are not conventional or toxic that may require regulation, including nutrients such as N and P.
[Source: http://www.epa.gov/owm/gen2.htm]

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

The NPDES Storm Water Permitting Program

Storm water runoff is one of the remaining causes of contaminated lakes, streams, rivers, and estuaries throughout the country. Pollution in storm water runoff is responsible for closing beaches and shellfish harvesting areas, contaminating fish, and reducing populations of water plants and other aquatic life.

High flows of storm water runoff cause flooding, property damage, erosion, and heavy siltation. The 1987 Congressional Amendments to the Clean Water Act required EPA to control pollution from storm water discharges. In 1990, EPA promulgated Phase I regulations to control storm water discharges from municipal separate storm sewer systems (MS4s) serving populations of more than 100,000, construction activities disturbing more than 5 acres, and industrial facilities through issuance of NPDES storm water permits. EPA promulgated Phase II of the program in 1999 to control storm water discharges from MF4s less than 100,000 and small construction sites between 1 and 5 acres in size. The Phase II regulations also expanded the exemption for industrial facilities that do not have exposure of industrial activities and materials to storm water.

Construction Permits

The 1987 Congressional Amendments to the CWA required EPA to control pollution from storm water discharges. EPA issued a general NPDES permit for construction sites disturbing 5 or more acres in 1992. General permits provide EPA with an effective mechanism to regulate discharges from tens of thousands of construction sites, thus protecting and improving surface water quality across the Nation. Several general permits for construction activity have been issued/reissued since that first permit in 1992. EPA Regions 1, 2, 3, 7, 8, 9, and 10 have a general permit that authorizes the discharge of storm water associated with construction activity disturbing 5 or more acres and smaller sources as designated by the Agency on a case-by-case basis. This multiregional permit is known as the "Construction General Permit" (CGP).

Region 4 has issued a separate CGP for the State of Florida and Indian Country lands in Florida, Mississippi, Alabama, and North Carolina. Region 6 has also issued its own CGP for the States of Texas and New Mexico; Indian Country lands in Texas, New Mexico, Oklahoma, and Louisiana; and construction activity at oil, gas, and pipeline facilities in Oklahoma.

As used in these construction permits, the term "storm water associated with construction activity" refers to category (x) of the definition of "discharge of storm water associated with industrial activity," which includes construction sites and common plans of development or sale that disturb 5 or more acres (see 40 CFR 122.26 [b][14]). The CGP applies only to areas for which EPA is the permitting authority (certain States, Federal facilities, and Indian Lands). The majority of the country (i.e., 44 States and the Virgin Islands) has been granted authority for permitting storm water discharges and as such, each of these States is required to develop permits to control discharges from construction activities. In response to the Phase II regulations, permit applications from construction activities between 1 and 5 acres is required by March 2003.

Combined Sewer Overflows (CSOs)

Combined sewer overflows, or CSOs, can be a significant water pollution and public health threat in urban areas. EPA's 1994 CSO Control Policy is a comprehensive national strategy to ensure that cities, NPDES authorities, water quality standards authorities, and the public engage in a comprehensive and coordinated planning effort to implement cost-effective CSO controls that meet the objectives and requirements of the Clean Water Act.

During dry weather, combined sewer systems transport wastewater directly to sewage treatment plants. In periods of rainfall or snowmelt, however, the wastewater volume in a combined sewer system can exceed the capacity of the collection system or treatment plant. When this happens, combined sewer systems overflow and discharge untreated wastewater directly to streams, rivers, lakes, or estuaries.

It provides guidance to municipalities and State and Federal permitting authorities on meeting pollution control goals of the CWA in a flexible, cost-effective manner. Information on EPA's CSO Control Policy, accompanying guidance documents, and other elements of the national CSO control program can be found on the following website: http://www.epa.gov/npdes.htm.

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

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

Total Maximum Daily Load (TMDL)
States, territories, and authorized Tribes establish section 303(d) lists of impaired waters based on information contained in their 305(b) reports as well as other relevant and available water quality data. The section 303(d) list is a prioritized list of waters not meeting water quality standards. States are required to submit lists biennially (40 CFR § 130.7(d)). States must develop TMDLs for waters and pollutants on their section 303(d) lists (CWA § 303(d)(1)(C)). State section 303(d) lists and TMDLs must be submitted to EPA for approval or disapproval. If EPA disapproves a list or TMDL submission, EPA must identify waters for the list or establish the TMDL itself (CWA § 303(d)(2)).

A TMDL is a written, quantitative plan and analysis for attaining and maintaining water quality standards in all seasons for a specific waterbody and pollutant. Specifically, a TMDL is the sum of the allowable loads of a pollutant from all contributing point, nonpoint, and background sources (40 CFR § 130.2(i)). TMDLs may be established on a coordinated basis for a group of waterbodies in a watershed. TMDLs identify the loading capacity of the water, wasteload allocations (for point sources), load allocations (for nonpoint sources), and a margin of safety (U.S. EPA 1999) (40 CFR § 130.2), and are calculated at levels necessary to achieve applicable water quality standards (CWA § 303(d)(1)(C)).

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

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

Continuing Planning Process (CPP)

Each State is required to establish and maintain a continuing planning process (CPP) as described in section 303(e) of the CWA. A State's CPP contains, among other items, a description of the process that the State uses to identify waters needing water quality-based controls, the process for developing TMDLs, and a description of the process used to receive public review of each TMDL (40 CFR § 130.5 & 130.7(a) & (c)). Descriptions may be as detailed as the Regional office and the State determine is necessary to describe each step of the TMDL development process. This process may be included as part of the EPA/State Agreement for TMDL development.
[Source: http://www.epa.gov/owowwtr1/tmdl/decisions/dec4.html]

Look to the Future ... Pollutant Trading

Point and nonpoint source pollutant trading involves financing reductions in nonpoint source pollution in lieu of undertaking more expensive point source pollution reduction efforts. A trading program is intended to produce cost savings for point source dischargers while improving water quality. In order for a trading program to be viable, there should be a waterbody identifiable as a watershed or segment, as well as a measurable combination of point sources and controllable nonpoint sources. In addition, point source dischargers can be expected to trade for nonpoint source reductions if they perceive this as an alternative to upgrade facility treatment capabilities. In addition, there should be significant load reductions for which the cost per pound reduced for nonpoint source controls is lower than the cost for upgrading point source controls.

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

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

8.2 MANAGING NONPOINT SOURCE POLLUTION

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

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

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

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

Nonpoint Sources of Nutrients

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

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

Agricultural Runoff

- Erosion and sediment control
- Control of facility wastewater and runoff from confined animal facilities
- Nutrient management planning on cropland
- Grazing management systems
- Irrigation water management

Urban Runoff

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

Silvicultural Runoff

- Streamside management
- Road construction and management
- Forest chemical management (includes fertilizers)
- Revegetation
- Preharvest planning, harvesting management

Marinas and Recreational Boating

- Siting and design
- Operation and maintenance
- Storm water runoff management
- Sewage facility management
- Fish waste management
- Pollution prevention education (e.g., proper boat cleaning, fish waste disposal, and sewage pumpout procedures)

Hydromodification (i.e., channelization, channel modification, dams)

- Minimize changes in sediment supply and pollutant delivery rates through careful planning and design
- Erosion and sediment control
- Chemical and pollutant control (includes nutrients)
- Stabilization and protection of eroding streambanks or shorelines

Wetlands, Riparian Areas, Vegetated Treatment Systems

- Protect the NPS abatement and other functions of wetlands and riparian areas through vegetative composition and cover, hydrology of surface and groundwater, geochemistry of the substrate, and species composition
- Promote restoration of preexisting function of damaged and destroyed wetlands and riparian systems
- Promote the use of engineered vegetated treatment systems if they can serve a NPS pollution abatement function

Efforts To Control Nonpoint Source Pollution

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

Nonpoint Source Management Programs

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

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

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

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

For additional information on the Nonpoint Source Management Program and distribution of Section 319 grants in your State, contact your State's designated nonpoint source agency. For many States, the nonpoint source agency is the State Water Quality Agency. However, in several instances, other agencies or departments are given nonpoint source responsibility (see Table 8-1).

National Estuary Program

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

Table 8-1. States for which the nonpoint source agency is not the water quality agency

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

- Albemarle-Pamlico Sounds, NC
- Barataria-Terrebonne Estuarine Complex, LA
- Barnegat Bay, NJ
- Buzzards Bay, MA
- Casco Bay, ME
- Charlotte Harbor, FL
- (Lower) Columbia River Estuary, OR and WA
- Corpus Christi Bay, TX
- Delaware Estuary, DE, NJ, and PA
- Delaware Inland Bays, DE
- Galveston Bay, TX
- Indian River Lagoon, FL
- Long Island Sound, NY and CT
- Maryland Coastal Bays, MD
- Massachusetts Bays, MA
- Mobile Bay, AL
- Morro Bay, CA
- Narragansett Bay, RI
- New Hampshire Estuaries, NH
- New York-New Jersey Harbor, NY and NJ
- Peconic Bay, NY
- Puget Sound, WA
- San Francisco Estuary, CA
- San Juan Bay, PR
- Santa Monica Bay, CA
- Sarasota Bay, FL
- Tampa Bay, FL
- Tillamook Bay, OR

Atmospheric Deposition

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

National Ambient Air Quality Standards

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

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

Atmospheric Deposition Initiative

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

Great Waters Program

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

Coastal Nonpoint Pollution Control Programs

In November 1990, Congress enacted CZARA. These amendments were intended to address several concerns, a major one being the impact of nonpoint source pollution on coastal waters.

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

States with Coast Nonpoint Pollution Control Programs are required to include measures in their programs that are "in conformity" with the 1993 CZARA guidance, as discussed previously. A listing of States with Coastal Nonpoint Pollution Control Programs is presented in Table 8-2. For additional information on the programs in these States, contact the State water quality agency.

Table 8-2. States and Territories with coastal nonpoint pollution control programs

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

Farm Bill Conservation Provisions

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

Environmental Conservation Acreage Reserve Program (ECARP)

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

Conservation Reserve Program (CRP)

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

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

Wetlands Reserve Program (WRP)

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

Environmental Quality Incentives Program (EQIP)

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

Wildlife Habitat Incentives Program (WHIP)

This program is designed for parties interested in developing and improving wildlife habitat on private lands. Plans are developed in consultation with NRCS and the local Conservation District. USDA will provide technical assistance and share up to 75% of the cost of implementing the wildlife conservation practices. Participants generally must sign a 5- to 10-year contract with USDA that requires they maintain the improvement practices.

Forestry Incentives Program (FIP)

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

Conservation of Private Grazing Land

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

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

8.3 COMPREHENSIVE PROCEDURE FOR NUTRIENT MANAGEMENT

Numeric water quality criteria adopted by States into their water quality standards can also serve as effective scientific tools for comprehensive water and land resource management. Effective programs incorporate aspects of prevention and maintenance as well as restoration. It is important that existing high-quality waters be managed wisely as a public resource, and waters whose uses are not yet threatened or impaired, but nonetheless are at risk from ongoing pollution, should be identified and managed so that designated uses are maintained in the future. The following 10-step management program that originated with the Wisconsin Inland Lakes Program (Gibson et al. 1983) has since been refined to become a natural resources management approach with utility for any water resource. States and Tribes can use this approach in addition to established regulatory protocols. Though intended to protect or enhance coastal marine or estuarine waters, the program does not, however, establish or replace any mandated procedures as part of a regulatory requirement.

Management of tidal and marine waters may be approached as a rational progression of actions beginning with a statement of major stressors and symptoms and progressing logically to a course of action and final assessment to determine the relative success of the effort. The following steps illustrate this management approach. States or communities are encouraged to adapt this technique to suit their particular needs and expectations. Where considerable information is already available, some of these steps may be skipped, but the methodology is presented here in detail for consideration.

Step 1: Status Identification
Data used during the preliminary nutrient criteria development process and the application of the criteria will present the resource manager with the general status of the estuary or coastal area and the need for responsive action. The information associated with these efforts, however, usually indicates a broad status condition, for example, high nutrient concentrations, algal blooms, fish kills, loss of seagrasses, or low-dissolved oxygen. Available data should be evaluated carefully to tease out potential connections to land use practices or recent changes in practices (e.g., development, fishing pressure, stocking or lack thereof). In particular, previous investigations should be reviewed to make a preliminary determination of anthropogenic cause(s) as opposed to natural cycling of the system. Trend assessments on the Chesapeake Bay, for example, are based on 10-year intervals in an effort to distinguish seasonality from cultural impacts. Essentially, a preliminary evaluation of readily available data is necessary to ascertain that there is indeed a problem or potential problem brought on by nutrient overenrichment and that the sources of the threat probably can be addressed to the betterment of these waters and the public good.

Step 2: Background Investigation

Given that the initial information reveals a viable management concern, it then becomes necessary and justifiable to gather as much background information as possible about the waters in question. There are three primary sources of such information.

Literature Searches

The initial effort here should be a search of the "gray literature" (often internal regional State and Federal agency reports that provide specific information about the relevant waters). Sources of such information include natural resource and fisheries agencies, forestry services, water quality administrations, hydrological and geological survey offices, planning offices, multi-State or county commissions, and community or environmental groups. A second source to comb through is peer-reviewed professional literature journals and related publications such as proceedings of conferences and symposiums, which may include specific studies of the estuary or coastal region of concern. The primary value of this source, however, probably will be discussions of methods and techniques of investigation and management. As the management investigation progresses, these sources of information become more pertinent.

Questionnaires

In preparing a list of agencies from which reports may be solicited, the names of key personal contacts should emerge. These contacts are the biologists, chemists, specialists, academics, resource managers, and citizen activists most familiar with these waters. As the literature and baseline data are reviewed, particular questions should develop, the answers to which will provide a fuller understanding of the resource and lend direction to the investigation and eventual management plan. Particularly helpful will be an understanding of the historical antecedents of the present status of the waters.

A standardized questionnaire can be prepared listing concerns such as the availability of any reports or data or understanding of the history of development in the watershed, perhaps including industries, agricultural practices, or development and structures associated with the resource. Particular episodes may be noted for comment in the questionnaire, such as hurricanes, fish kills, algal blooms, or spill events, as well as historical problems, such as marina development, port facilities, spoils disposal sites, agricultural runoff, erosion problems, or development concentrations.

All discharge sites should be documented, such as industrial discharges, "Superfund sites," wastewater treatment plants, drains, concentrations of housing with onsite wastewater treatment, marinas, major road crossings, and tributaries potentially bearing large loadings of sediments or nutrients. Problem land use areas along the shore also should be noted, for example, degraded wetlands, embayments where blooms or fish kills regularly occur, or areas where seagrasses recently have contracted. In addition, it is helpful to include a large, fairly detailed line drawing of the estuary or portion of the estuary or coastal reach and its proximal watershed that the respondent may use to reference particular observations.

If at all possible, the questionnaire should be limited to no more than two pages of questions, including space for answers plus the line drawing. Questions should be direct and concise. Determine exactly what you wish to learn and ask questions specifically related to this information. Opportunities for additional comments should include an open-ended question at the end of the questionnaire.

To get the best response to a questionnaire, the potential respondents should be called first to confirm their mailing addresses and availability. They should be advised of the nature of the study and their cooperation then requested. Other potential respondents may be identified through these calls. If a large survey is necessary, this preliminary step may not be possible. Most regional inquiries, however, are made usually to no more than 50 specialists, and the additional information gained is well worth the telephone calls.

Interviews

By this point in the background investigation process, the key people to contact for detailed information should be evident. Their names will have come up in conversations and on reports, and they will be the people providing the most helpful responses on the questionnaires.

Other valuable contacts are Basin Commissions; Interstate and State management agencies; EPA, NOAA, and U.S. Fish and Wildlife Service specialists; the USDA Cooperative Extension Service agents for the area; State, county, and municipal planners; and university faculty. Long-term residents and commercial as well as sport fishermen and their organizations should be contacted also. Anecdotal information can be invaluable and helps add perspective to other sources of data.

The interviews should assess the questionnaire data gathered already; they should clarify and elaborate on the basic information generated. The interviews also are the means by which apparent contradictions in perceptions or observations may be resolved at least partially. It should be noted that many people are uncomfortable with recorded interviews; note-taking often is less intimidating. In either case, immediately after each interview, a record of answers and observations should be prepared while the impressions of the interviewer are still fresh.

The information compiled from each background investigation should clarify further the initial problem stated. It should help resolve any ambiguities about the dynamics of the system and the human community. In addition, the compilation should identify areas where more definitive, primary data collection is required to clearly understand the nutrient problems of a particular waterbody and provide direction for the subsequent management project.

Step 3: Data Gathering and Diagnostic Monitoring

Data obtained during the nutrient criteria development process are the mainstay of the database to be prepared for any subsequent investigation. The intent of such a process is to develop a reasonable image of the status of the estuary, bay, or coastal region. Diagnostic monitoring should expand on that structure and extend the understanding from status of the resource to a diagnosis of causes of the overenrichment. For example, where several tributaries of an estuary or salinity zone have been sampled and two are identified as being of concern, they and other higher order feeder streams must now be sampled to further target the locations of probable loadings. Though earlier sampling was done to portray the enrichment state of the waters, subsequent sampling should focus on near-shore areas of potential loadings, tributaries, and portions of the tributaries where loadings may originate.

Diagnostic monitoring supports the identification of water quality problems and helps to develop an appropriate management plan. General guidelines for conducting diagnostic monitoring are as follows.

Parameters To Sample

Diagnostic monitoring is conducted after nutrient criteria have been established. It might not be necessary, therefore, to sample some parameters that are not related to the criteria.

- Diagnostic sampling for nutrients requires an estimation of nutrient loading and sources. Major potential sources of nutrients (e.g., tributary streams, groundwater flow, runoff, illegal discharges, atmospheric deposition) should be identified and sampled in such a way as to obtain an estimate of annual loads from each source.

- The variables and techniques employed in the preliminary survey should be reviewed for adequacy and either repeated or augmented. A manager should not eliminate the basis of the original classification by dropping any variables or stations at this point. Documenting potential success or failure of the subsequent management program will require "before" and "after" databases, and the initial survey design should be modified only after careful consideration and due attention to reestablishing the baseline survey.

Flow measurements also are an essential part of this survey. Perhaps more important than in any other water resource investigation, attention to tidal state and amplitude and seasonal hydrologic characteristics is imperative to determining the extent and source of nutrient loadings to estuaries and coastal marine waters. If nutrient concentrations are to be compared meaningfully and loading estimates made, cross-sectional areas and flow rates for all tributary streams and discharges also must be included in the survey design. These measurements must be made or extrapolated whenever water quality samples are collected. Without this information, assigning priorities to various loading sources identified in the investigation will be difficult or impossible.

Sampling Frequency

Sampling frequency will increase for diagnostic monitoring because the sample population is now a particular area of the waterbody. Sampling should occur repeatedly during the growing season to precisely characterize individual areas as well as discharges and loadings. Statistical power analysis can be used to determine the appropriate sample size based on the purpose of the sampling and the acceptable error (see Chapter 5).

In addition to expanding the number of stations and parameters to accommodate diagnostic determinations, the survey design should address temporal variables by sampling these stations during each season of the year at times calibrated to the particular climate and locale. Accommodation may be needed for periods of base flow, maximum runoff, turnovers, periods of maximum and minimum productivity, and, in some instances, migratory patterns of fish or waterfowl. Seasonal changes in land use such as peak summer or winter vacation periods, agricultural applications and harvests in the watershed, and seasonal commercial or industrial activities also should be addressed.

To separate signals from seasonal "noise," it may be necessary to gather survey data for 2 or more consecutive years to strengthen data assessment (as noted above, some estuary management programs survey over as many as 10 years, requiring an extensive operating budget). Such assessments will require a robust statistical evaluation of the data; this element should be incorporated into the study design at the outset. As with the initial survey design, the preliminary statistical tools chosen may be carried into this subsequent design as well. Care should be taken to replicate sample collections to ensure representative sample design and confidence in the results obtained. Early inclusion of a skilled environmental statistician on the management team is advisable.

Sampling Location

If turbidity, nutrients, and algae are known to be variable across the surface of an estuary or salinity zone, then multiple sample sites within that zone are required. The exact number of sampling sites in a zone is determined by the spatial variability of nutrients, turbidity, and chlorophyll and the desired precision. In general, within a basin or zone, variation in time is larger than variation in space (Knowlton and Jones 1989). Thus, chlorophyll samples taken 2 weeks apart may differ severalfold, but samples taken on the same day 500 meters apart are likely to differ much less. Depending on the questions being addressed in the investigation, spatially composite samples may be more cost-effective than separate samples from several sites.

The design and placement of these sample stations will rely heavily on the proximal and watershed land use information garnered from the background investigation. The overall objective should be to bracket suspected sources of nutrient loadings in the tributaries and near-bank areas so parcels can be either selected or eliminated as potential candidates for management attention.

Step 4: Source Identification

The cumulative information gathered should now provide a clear image of the state of the estuarine or coastal segment, the most likely sources of nutrient loadings or related degradation, and their relative contributions to the problem. It is important to note that this process reveals only local sources of the overenrichment. Atmospheric deposition of nitrogen compounds and other broad-scale impacts beyond the watershed scale are not specifically addressed and must be assumed as essentially an environmental constant. With all the risks this constant entails, it is probably not an undue assumption; remediation of such depositions is probably beyond the scope of most nutrient management projects employing this guidance.

The problems to be identified are likely to be as diverse as the geology, hydrology, and land use practices of the waterbody and watershed. Typical developments include sediment resuspension and nutrient re-release; biotic imbalances affecting nutrient utilization caused by overfishing or stock mismanagement; discharge of excess nutrients directly to the waters by wastewater treatment plants, storm water runoff, or failing septic systems; and runoff from municipalities, subdivisions, farms, commercial enterprises, and industrial activities. Other problems have included concentrations of migratory and resident waterfowl contributing to an excess of nutrients, removal or filling of bank areas and wetlands that once intercepted nutrient runoff, runoff of herbicide applications that killed macrophytes and promoted nuisance algal blooms, and chronic, low-dissolved oxygen problems attendant to overenrichment and vegetative imbalances.

Any combination of these in situ and land use problems can cause a cumulative overenrichment problem. Management planning requires identifying first the loading sources and, second, of those sources, the ones that are most significant. Proximity of a source to a lake or reservoir (or in some cases, the ubiquitous nature of a source throughout a watershed such as subdivision or farm runoff), the relative loading estimate of that source, and the likelihood of successful remediation are the key factors in deciding which problem sources are priorities for inclusion in a management plan.

Loading estimation models are valuable for estimating the relative significance of various nutrient sources in the watershed with respect to the likely response of the waters. Chapter 9 describes many of these models and their relative utility. Modeling permits a manager to try out various scenarios and combinations of techniques to estimate their likely effectiveness. Some of these options for a nutrient management plan are discussed in the next step.

Step 5: Management Practices for Nutrient Control

Once the major sources of concern are identified and agreed on by the management planners, remedial measures appropriate to these sources must be identified. Management practices are well defined and documented for a variety of land uses in EPA guidance documents, USDA manuals, U.S. Forest Service manuals, and urban land use planning guides. Resource managers should study these references for likely approaches to consider and then consult regional experts in each of the subject land uses for qualification and other suggested management practice recommendations. Bringing these specialists together as a small workgroup is an effective, although sometimes contentious, way to develop the most technically sound approaches to such problems.

Fitting the various components together in a comprehensive management plan is challenging. It calls for both imagination and cooperation. Usually no one approach stands out as the obvious best choice. Instead, two or three permutations of several generally agreed on BMPs will evolve from the planning sessions.

Selection of the optimal approach—or more likely, the best candidates—should first involve careful assessment by the planning workgroup and then consultation with all elements of the watershed community, both organized interest groups and private landowners. The first phase should be conducted using the threefold framework of evaluation developed by the Department of Resource Development at Michigan State University (Figure 8-2). The premise behind this approach holds that the most effective and achievable management plan should address three elements of practicality: scientific validity, sociopolitical consideration, and economic consideration.

- No resource or environmental management plan should be considered unless it is scientifically valid. The technology proposed should be based in sound science and tested and validated. No attempt to manipulate the environment and peoples' land use prerogatives should be made unless it can be demonstrated in advance that the technique is reliable or at least that the risks are quantifiable and understandable.

- The proposed approach should be cost-effective and affordable by the community. Among technically sound plans to achieve desired goals, the most cost-effective (typically those with elements that have the greatest benefit-cost ratios) are the easiest to implement and most likely to satisfy the public interest.

- The management plan should have adequate social and political acceptability. A plan that seems rational and cost-effective may conflict with the collective values of the local public. Any action taken in addition to existing requirements should always be researched carefully for justification, efficacy, lead time required, and likely effects on various segments of the community.

The resource manager most likely to achieve success will consider and responsibly address each of these three elements. All candidate alternatives should be evaluated in this manner and revised as necessary.

Such evaluation not only generates the optimal plan (or plans where competing but different strengths are evident), but documents the rationale, essential for public review before the final selection is made.

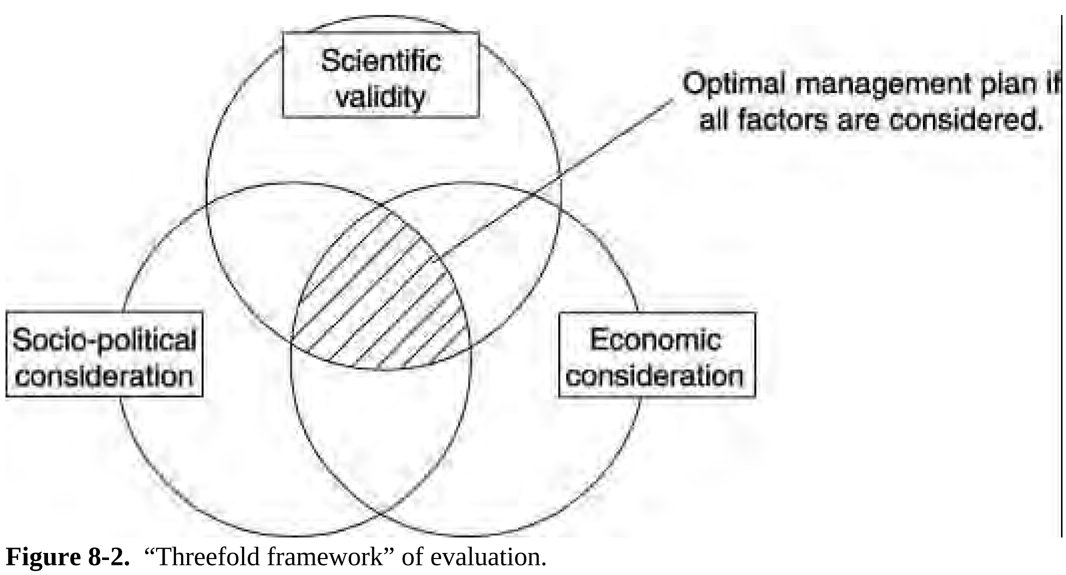

Figure 8-2. "Threefold framework" of evaluation.

Involving the public in the process throughout is highly beneficial, and invitations to meetings or advisories to all potentially interested parties should be provided regularly, if not from the outset then certainly before plan selection and approval are needed. A balance must be struck between making public announcements too early, which could arouse people before sufficient information has been generated, and making announcements too late, which may lead to suspicions that information is being withheld from the public.

Step 6: Detailed Management Plan Development

A detailed management plan should include all 10 steps of the process described here. The first five steps are necessary to achieve the design of the plan, but they also should be included so that anyone reading it will understand what has gone into the effort.

Natural resource management efforts can include one or more of these three elements: education, financing, and regulation; it is best to initiate them in the order presented here. Start with relatively low-cost information and education efforts to acquaint people with the problem and how you propose to address it, and to obtain their suggestions and perceptions. A good educational effort should be the incentive for volunteer agreements and cooperative action. A grant-in-aid or other assistance often is the key element to encourage individuals or jurisdictions to adopt appropriate water resource protection practices. Regulatory actions are necessary and appropriate when mandated by law, when cooperation and compliance are unlikely to occur otherwise, and when voluntary efforts have not succeeded.

Step 7: Implementation and Communication

Periodic progress reports during implementation of the management project are opportunities to communicate with administrators, other involved agencies, politicians interested in the project, the general public and landowners, and other interest groups. Progress reports should be brief and candid. They will be part of the public record so that all parties are properly informed, the post-project cry of inadequate notification is avoided or at least minimized, and techniques and methods used are documented.

Regional public meetings and hearings are excellent ways to communicate. The more controversial an issue, the more communication is necessary. To continue implementing a plan despite significant opposition without evaluating the consequences is a mistake, especially if a change to a step in the management plan with additional public consultation would still achieve the same objective.

Step 8: Evaluation Monitoring and Periodic Review

The management plan should always include "before, during, and after" water resource quality monitoring to demonstrate the responses of the system to management efforts. This is the reason for maintaining and expanding the initial survey stations. Monitoring data are important for evaluating progress and are included in the requisite progress reports described above. The change or lack thereof in the status of the estuarine or coastal waters is the ultimate determination of management success.

These built-in monitoring schedules should include seasonality and periodic data assessment intervals for management review to permit responses to changing circumstances, modifications of methods, schedules, and changes of emphasis as needed.

Step 9: Completion and Evaluation

Management projects are frequently planned, initiated, and concluded and even new initiatives undertaken to meet pressing schedules without sufficient evaluation of the initial project.

Reviews of progress reports, of the original objectives, and of the monitoring data will reveal whether the lake or reservoir trophic state was successfully protected or improved. Evaluation also provides documentation for determining whether the project's methods and techniques can be applied elsewhere, perhaps with modification. Finally, it will reveal any mistakes that should be avoided in future projects and perhaps will demonstrate that a sequel project is required to fully accomplish the original objectives.

Step 10: Continued Monitoring of the System

The monitoring initiated and expanded in the course of the project can now be reduced to periodic measuring of key variables at critical times and locations. The purpose now is to keep sufficiently informed of the status of the waters to ensure that the protection or remediation achieved is maintained. If periodic evaluation monitoring indicates a return of trophic decline, intervention should be possible at an early point so that costs of preserving that which was achieved are reduced. These evaluation and periodic monitoring steps essentially complete the process. If new issues arise, the manager returns to Step 1 with a new problem statement. General guidelines associated with evaluation monitoring are provided below.

Parameters To Sample

Each water quality parameter discussed in the Indicators chapter—TP, TN, chlorophyll *a*, Secchi depth, and dissolved oxygen as well as perhaps selenium and vegetation and indicator organisms—should be sampled during maintenance monitoring. Because the purpose of maintenance monitoring is to determine if conditions have changed or if criteria are exceeded, other physical or chemical variables need not be measured.

Sampling Frequency

Sampling efforts for maintenance monitoring can be adaptive and sequential, so that a certain minimum of information is collected at regular intervals, and if data indicate change or uncertainty, the sampling effort (in both time and space) can be increased to attempt to reduce the uncertainty. For example, a station in an undisturbed area could be sampled once every 5 years, from a single visit during an index period (say, midsummer). If results suggest a change in conditions beyond what is normally expected for these waters, then additional and more frequent sampling can be continued to determine if the departure from "normal" conditions is real and if it is ecologically significant. If TP, TN, chlorophyll *a*, and Secchi depth relationships have been established, it may be cost-effective to use Secchi depth as a preliminary indicator; if a trigger value is detected, more parameters can be measured.

Such variation also suggests different levels of maintenance monitoring, depending on existing knowledge of the waters and expectations. Maintenance monitoring may be done for several purposes:

- Routine monitoring of waters of known quality (i.e., sampled before) that are not expected to change greatly

- Initial sampling of a station of unknown quality

- Monitoring of a station or stations of known quality expected to change, say, as the result of watershed development or restoration efforts

Routine monitoring of stations of known quality is the least intensive and typically requires sampling once every several years, as in the example above. However, initial sampling of an estuary or coastal area of unknown quality requires the same sampling effort, and parameters, as the classification survey. Monitoring a known estuary or portion of an estuary or coastal area that is expected to change or suspected to have changed requires more intensive effort, typically an increase in sampling frequency to several times during the growing season, to obtain seasonal averages of indicator values.

The actual frequency of sampling should be determined by the number of samples required to detect an ecologically relevant change in the indicators, resources available for the monitoring program, and amount of time for a change to be detected. These considerations require power analysis using existing or preliminary data, and tradeoffs of desired significance level, desired power, desired effect size that is detectable, ecological significance, and most important, resources (labor and money) available for the monitoring program.

Sampling Location

For routine monitoring, it is recommended that the sampling locations be at least the same as for the classification survey to provide the database with a certain minimum continuity.

8.4 RESOURCES

Listed below are selected publications concerning coastal or estuarine and watershed management and protection.

- National Research Council. 2000. Clean Coastal Waters - Understanding and Reducing the Effects of Nutrient Pollution. National Academy Press. Washington, DC.

- Gibson, G. R., M.L. Bowman, J. Gerritsen, and B.D. Snyder. 2000. Estuarine and Coastal Marine Water Bioassessment and Biocriteria Technical Guidance. EPA 822-B-00-024. U. S. Environmental Protection Agency; Office of Water; Washington, DC.

- Sharpley, AN, ed. 2000. Agriculture and phosphorus management: the Chesapeake Bay. Boca Raton, FL: Lewis Publishers.

  This text is a compilation of conference proceedings describing nutrient dynamics in the watershed of Chesapeake Bay, with emphasis on agricultural loadings and practices. Although directed at an estuarine environment, much of the agriculturally based nutrient information has broad application.

- U.S. Environmental Protection Agency. 1993. Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters. EPA-840-B-92-002.

  The EPA Office of Water produced the 1993 guidance document to support the Coastal Zone Act Reauthorization Amendments of 1990. This document describes several management measures to control nonpoint sources of pollution, including nutrients.

- U.S. Environmental Protection Agency. 1995. Watershed Protection: A Project Focus. EPA 841-R-95-003.

  This document focuses on developing watershed-specific programs or projects. It provides a blueprint for designing and implementing watershed projects, including references and case studies for specific elements of the process. The document illustrates how the broader principles of watershed management, including all relevant Federal, State, Tribal, local, and private activities, can be brought to bear on water quality and ecological concerns.

- U.S. Environmental Protection Agency. 1995. Watershed Protection: A Statewide Approach. EPA 841-R-95-004.

  This document is primarily designed for State water quality managers. A common framework for a statewide watershed approach focuses on organizing and managing a State's major watersheds (called basins in this document). In this statewide approach, activities such as water quality monitoring, planning, and permitting are coordinated for multiple agencies on a set schedule within large watersheds or basins.

- U.S. Environmental Protection Agency. 1997. Monitoring Consortiums: A Cost-Effective Means to Enhancing Watershed Data Collection and Analysis. EPA 841-R-97-006.

  This document addresses coordination in watershed monitoring. As demonstrated in the document's four case studies, consortiums can stretch the monitoring dollar, improve cooperation among partners, and increase sharing of expertise as well as expenses of data collection and management.

- U.S. Environmental Protection Agency. 1997. Land Cover Digital Data Directory for the United States. EPA 841-B-97-005.

Land cover, which is the pattern of ecological resources and human activities dominating different areas of the Earth's surface, is one of the most important data sources used in watershed analysis and the management of water resources throughout the country. The 75 land cover data summaries in this directory include contact information to assist readers who may want to acquire copies of the digital data for their own use.

- U.S. Environmental Protection Agency. 1997. Designing an Information Management System for Watersheds. EPA 841-R-97-005.

  This document is an introduction to the information management responsibilities and challenges facing any watershed group. The document reviews the fundamentals of identifying information management needs, integrating different databases, evaluating hardware and software options, and developing implementation plans.

- U.S. Environmental Protection Agency. 1997. Information Management for the Watershed Approach in the Pacific Northwest. EPA 841-R-97-004.

  This document centers on a series of interviews with leaders and key participants in the statewide watershed approach activities in the State of Washington. The document reviews Washington's statewide watershed activities in case study fashion.

- U.S. Environmental Protection Agency. 1998. Inventory of Watershed Training Courses. EPA 841-D-98-001.

  This inventory provides one-page summaries of 180 watershed-related training courses offered by Federal and State agencies; it also lists resource professionals in the private sector.

- U.S. Environmental Protection Agency. 1997. Statewide Watershed Management Facilitation. EPA 841-R-97-011.

  This document addresses statewide watershed management and the process of facilitating the development or reorientation of statewide watershed programs. It includes State case histories.

- U.S. Environmental Protection Agency. 1996. Watershed Approach Framework. EPA 840-S-96-001.

  This publication revisits and updates EPA's vision for a watershed approach, first explained in a 1991 document entitled "Watershed Protection Approach Framework." It describes watershed approaches as coordinating frameworks for environmental management that focus public and private efforts to address the highest priority problems in defined geographic areas, involving both ground and surface water flow.

- U.S. Environmental Protection Agency. 1997. Top 10 Watershed Lessons Learned. EPA 840-F-97-001.

  Watershed work has been going on for many years now, and this 60-page document summarizes the top lessons that have been learned by watershed practitioners across the United States regarding what works and what does not.

- U.S. Environmental Protection Agency. 1999. Catalog of Federal Funding Sources for Watershed Protection (second ed.). EPA 841-B-99-003.

  Many sources of Federal funding are available to support different aspects of watershed protection and specific types of local-level watershed projects. This document presents information on 52 Federal funding sources (grants and loans) that may be used to fund a variety of watershed projects.

- U.S. Environmental Protection Agency. 1997. Watershed Training Opportunities. EPA 841-B-97-008.

  This is a 22-page booklet developed to highlight watershed training opportunities offered by EPA's Office of Water and the Watershed Academy. It covers training courses and educational materials on watersheds produced throughout the EPA Office of Water.

- U.S. Environmental Protection Agency. 1997. Stream Corridor Restoration: Principles, Processes and Practices. EPA 841-R-97-011.

  This document is a practical reference manual and logical framework to help environmental managers recognize stream restoration needs and design and implement restoration projects.

- U.S. Environmental Protection Agency. 1997. Protocol for Developing Nutrient TMDLs. EPA 841-B-99-007.

  This protocol is an organizational framework for the TMDL development process for nutrients. It leads to an understandable and justifiable TMDL.

CHAPTER 9

Use of Models in Nutrient Criteria Development

- Model Identification and Selection
- Model Classification
- Use of Models for Nutrient Investigation
- Management Applications

All models are wrong. Some models are useful.
(George E. P. Box)

9.1 INTRODUCTION

This chapter addresses the role of models in nutrient criteria development. It is closely linked to Chapter 5, which addresses database development, sampling designs, and monitoring. One system is said to model another when the observable variables in the first system vary in the same fashion as the observable variables in the second (NRC 2000). Chapter 7 of the NRC report goes on to state that models come in many forms. They may be empirically derived statistical relationships plotted on a graph, physical analogues (e.g., mesocosms) of the system of interest, analogues of different systems that have useful parallel relationships of observable factors (e.g., from physics, the flow of water through pipes to model the flow of electrons through an electrical circuit), or numerical models run on computers that are based on first principles or empirical relationships.

Environmental water quality models have several uses (e.g., reduce ecosystem complexity to a manageable level, improve the scientific basis for development of theory, provide a framework to make and test predictions, increase understanding of cause-and-effect relationships, and improve assessment of factor interaction). Reliable predictions stand out as a salient requirement because of the social and economic consequences if predictions are unreliable. Many times decisionmakers rely on models to guide their environmental management choices, especially when costly decisions are involved and the problem and solution involve complex relationships. This is exemplified by the decision of the Long Island Sound Hypoxia Management Conference (see Case Study for Long Island Sound). Generally, empirical and mathematical models are the most widely used models that statistically or mathematically relate nutrient loads or concentrations to important ecological response variables (e.g., dissolved oxygen deficiency, algal blooms and related decrease in water clarity, and loss of seagrasses). They both depend on the scientific robustness and accuracy of underlying conceptual models.

This chapter addresses both empirical and mathematical models. Considerably more space is devoted to mathematical models, because they are capable of addressing many more details of underlying processes when properly calibrated and validated. They also tend to be more useful forecasting (extrapolation) tools than simpler models, because they tend to include a greater representation of the physics, chemistry, and biology of the physical system being modeled (NRC 2000). A great danger in complex mathematical models is that error propagation is difficult to explicitly measure, and there is a tendency to use a more complex model than required, which drives costs up substantially and unnecessarily. Another consideration that is gaining acceptance is that mathematical models need to be appropriately scaled to spatial and temporal processes, or they may suffer problems similar to empirical models when one extrapolates the results of scaled experiments to full-sized natural systems. Also, empirical coefficients introduced into equations often hide the degree of uncertainty concerning the fundamental nature of the processes being represented.

Use of Empirical Models in Nutrient Criteria Development

Statistical models are empirical and are derived from observations. To be useful as predictive tools, relationships must have a basis in our understanding, typically represented by conceptual models. However, extrapolation from empirical data is known to be uncertain. Thus, these models are most reliable when used within the range of observations used to construct the model. When shown to meet program objectives and requirements, empirical models are a desirable place to begin model development and, if later determined to be required, they often provide insights into the structure needed for development of mathematical models. Empirical models typically are useful if only a subsystem of the larger ecosystem is of primary interest.

Frequently, the impression is given that the only credible water quality modeling approach is that of mathematical process-based dynamic computer modeling. This is not the case. For example, a Tampa Bay water quality modeling workshop in 1992 (Martin et al. 1996) produced the consensus recommendation that a multipronged (mechanistic and empirical) modeling approach be implemented to provide technical support for the water quality management process. The Tampa Bay National Estuary Program produced an empirical regression-based water quality model. The estimated N loads were related to observed chlorophyll concentrations using the regression model (Janicki and Wade 1996):

$$C_{t,s} = \alpha_{t,s} + \beta_s * L_{t,s}$$

where $C_{t,s}$ = average chlorophyll *a* concentration at month t and segment s,

$L_{t,s}$ = total N load at month t and segment s,

$\alpha_{t,s}$ and β_s are regression parameters.

A related model equated Secchi depth to average chlorophyll *a* concentrations (Greening et al. 1997). This analysis was followed by an empirical model that related N loadings to in-bay chlorophyll *a* concentrations.

There are many other examples of empirical models used to relate environmental forcing functions to ecological responses, especially nutrient load/concentration and response relationships. Much of the professional aquatic ecological literature reports on use of empirical models (e.g., Chapters 2 and 3). Empirical models have their limitations, but when judiciously applied, they offer a highly useful tool to water quality managers.

Use of Mathematical Models in Nutrient Criteria Development

Mathematical models can play an important role in assessing acceptable nutrient loads and concentrations in estuaries and near-coastal areas. For example, models are used to:

- Develop a relationship between external nutrient loads and resulting nutrient concentrations, which can then be used to define allowable loads

- Define the relationship between nutrient concentrations and other endpoints of concern, such as biomass or dissolved oxygen

- Provide an increased understanding of the factors affecting nutrient concentrations, such as the relative importance of point and nonpoint source loads

- Simulate relationships between light attenuation and expected depth of sea grass growth

The intent of this section is to describe the models available for assessing the relationship between nutrient loading and nutrient-related water quality criteria for estuaries and near-coastal waters. This chapter provides general guidance and some specific procedures for selecting and applying an appropriate model. It is divided into the following sections: (1) Model Identification and Selection, (2) Model Classification, (3) Use of Models for Nutrient Investigation, and (4) Management Applications.

Extensive EPA guidance (i.e., U.S. EPA 1985, 1990a-c; 1997; EPA document # 841-B-97-006) currently exists on these topics. This section serves primarily to condense the existing guidance with some modifications, to reflect changes in the science that have occurred subsequent to their publication. In addition, emphasis is placed on the simpler, more empirical techniques that are applied most easily. Readers are referred to the original guidance materials for more detailed discussions of the concepts described in this section.

9.2 MODEL IDENTIFICATION AND SELECTION

The first steps in the modeling process are model identification and selection. The goals are to identify the simplest model(s) that addresses all of the important phenomena affecting the water quality problems, and to select from those the most useful analytical formula or computer model. Selection of too simple a model can result in predictions of future water quality that are too uncertain to achieve the decisions or objectives of the study. On the other hand, selection of an overly complex model may also result in misdirected study resources, delays in the study, and increased cost. Predictive uncertainty may increase to unacceptable levels because of model parameters that cannot be adequately estimated with available data. Study costs will increase because of the additional data requirements and the expanded computer and staff time needed for model runs, analysis, and sensitivity studies.

Model Identification

Model identification entails four basic steps:

- Establish study objectives and constraints
- Determine water quality pollutant interactions
- Determine spatial extent and resolution
- Determine temporal extent and resolution

Each is discussed below.

Study Objectives and Constraints

The first step in identifying an appropriate model for a particular site is to clearly delineate the objectives of the modeling analysis. These objectives address questions such as:

- What are the nutrients of concern?
- What are the environmental endpoints of concern?
- What spatial and temporal scales are adequate for management concern?
- What management issues must the model address?
- What is the acceptable level of uncertainty in model predictions?

The nutrients of concern addressed in this document are nitrogen or phosphorus (depending on which is the limiting nutrient or will become limiting after controls are implemented). Environmental endpoints of concern are total nutrient concentration and other indicators of excessive nutrients such as chlorophyll/biomass and minimum dissolved oxygen. Local, State, and Federal regulations contribute to the definition of objectives by specifying time and space scales that the model must address: for example, the averaging period, or the season at which the criteria are applicable.

All expected uses of the model are to be stated clearly in advance. If the model will be used to predict future allowable nutrient loads, the specific conditions to be evaluated must be known. Then a final study objective is established that pertains to the required degree of reliability of model predictions, which may vary depending on whether the model application is designed for screening level estimation or for more detailed predictions.

The reliability objective is directly related to project constraints, as there is often a mismatch between desired model reliability and available resources. Resource constraints can cover four areas: data, time, level of effort, and expertise. Appropriate model selection must be balanced between competing demands. Management objectives typically favor a high degree of model reliability, but resource constraints generally prohibit the degree of reliability desired. Decisions often are required regarding whether to proceed with a higher-than-desired level of uncertainty, or to postpone modeling until additional resources can be obtained.

Water Quality/Pollutant Interactions

After the pollutants and water quality indicators are identified, the significant water quality processes must be determined. These processes directly or indirectly link the pollutants to be controlled with the primary water quality indicators. All other interacting water quality constituents thought to be significant should be included at this point. This consolidation can best be seen in a diagram or flow chart representing the mass transport and transformations of water quality constituents in a defined segment of water. Figure 9-1 illustrates variables and processes important to the eutrophication process. Not all of these need to be included in the actual model selected for use. Those excluded, however, should be considered externally and reflected in the coefficients.

At the end of this step all the available knowledge of a system should be assimilated in a way that permits major water quality processes and ecological relationships to be evaluated for inclusion in the numerical model description. This conceptual model is the starting point from which systematic reductions in complexity can be identified to provide an adequate representation of the system while meeting the objectives of the study.

The simplest level of model complexity considers only total nutrient concentrations and assumes that all of the processes shown in Figure 9-1 either have no effect on total nutrient concentrations (as is sometimes assumed for total nitrogen), or can be lumped into a single overall loss coefficient.

Models that simulate phytoplankton concentrations or dissolved oxygen typically include all of the processes shown in Figure 9-1, and sometimes many more, to describe such processes as sediment diagenesis and competition among multiple phytoplankton classes. Denitrification in the model is expressed in terms of the water column carbonaceous biochemical oxygen demand (CBOD).

Spatial Extent and Scale

Two spatial considerations must be addressed in the model identification process: spatial extent and scale. Spatial extent pertains to the specific boundaries of the area to be assessed. Spatial scale pertains to the number of dimensions to be considered and the degree of resolution to be provided in each dimension.

Several guidelines can help locate proper model boundaries. In general, the boundaries should be located beyond the influence of the discharge(s) being evaluated. Otherwise, proper specification of boundary concentrations for model projections is very difficult. Boundaries should be located where flow or stage and water quality are well monitored. Upstream boundaries should be located at a fall line, or at a gaging station in free-flowing, riverine reaches. Downstream boundaries are best located at the mouth of an estuary, or even nearby in the ocean. For large estuaries with relatively unaffected seaward reaches, the downstream boundary can be located within the estuary near a tidal gage and water quality monitoring station.

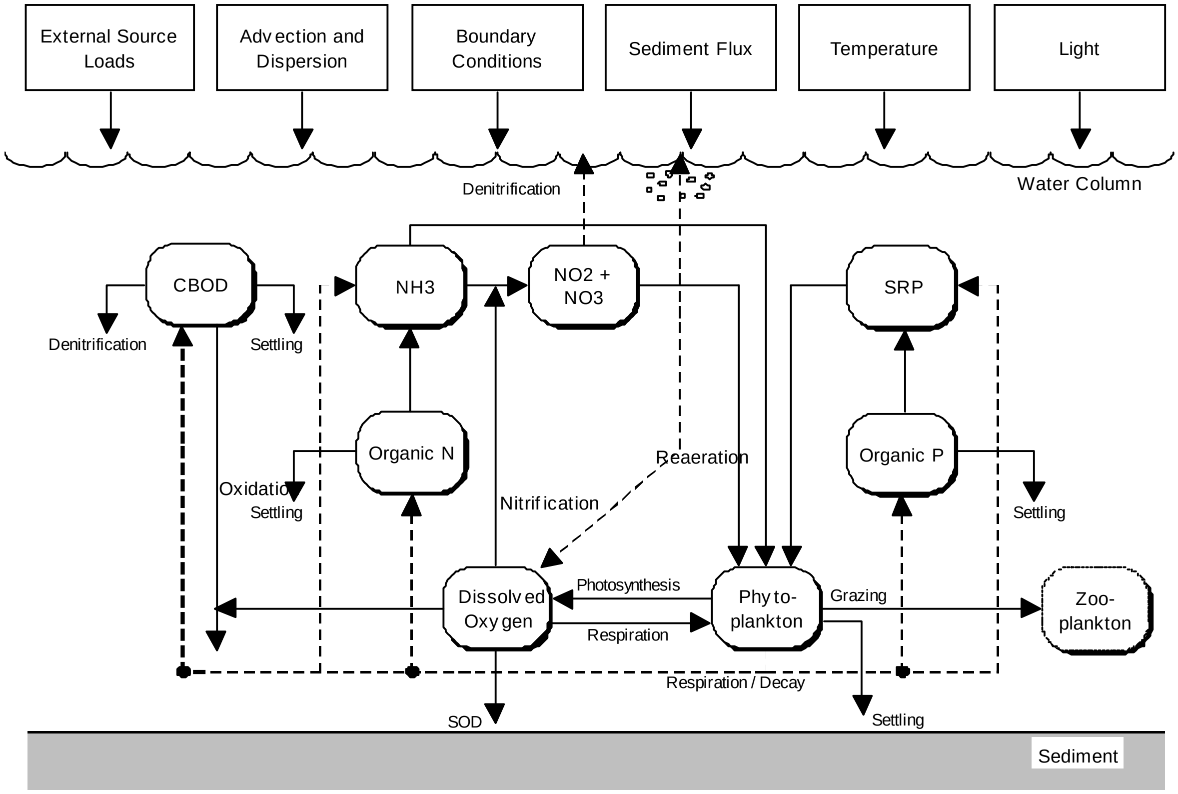

Figure 9-1. Eutrophication model framework : an example of hypoxia-based conceptual framework for water quality model. Source: Bierman et al. 1994.

Appropriate model spatial scale requires consideration of two factors: (1) the extent to which spatial gradients in water quality occur and (2) the extent to which these variations need to be considered from a management perspective. Real estuaries and near-shore waters all exhibit three-dimensional properties. There are gradients in hydrodynamic and water quality constituents over length, width, and depth. The effective dimensionality of an estuary includes only those gradients that affect the water quality analysis significantly.

One-dimensional models consider the change in pollutant concentration over a single dimension in space, typically oriented longitudinally down the length of an estuary. Two-dimensional models can consider concentration gradients in the lateral and longitudinal directions (termed x-y orientation), or concentration gradients that occur longitudinally and vertically (termed x-z orientation). Three-dimensional models describe changes in concentration that occur over all three spatial dimensions. These models provide the most detailed assessment of pollutant distribution with respect to a discharge; they also have the most extensive model input requirements and are the most difficult to apply.

Justifiable reductions in dimensionality result in savings in model development, simulation, and analysis costs. Usually the vertical and/or lateral dimension is neglected. Eliminating a dimension from the water quality analysis implies acceptable uniformity of water quality constituents in that spatial dimension. For example, use of one-dimensional models implies acceptably small deviations in concentration from the cross-sectional mean, both vertically and laterally. This judgment requires understanding both the transport behavior of estuaries and the specific objectives of the study.

Spatial variations are best determined by plotting observed water quality concentrations versus distance along the dimensions of concern. If such data are not available, other types of methods are available to estimate the importance of spatial variations. These are described in U.S. EPA (1990a) and discussed briefly here. The methods can be divided into three categories:

- Relatively simple desktop methods that compare the stratification potential caused by freshwater inputs to the mixing potential caused by tides and other currents

- Dye studies that observe the degree of mixing

- Geomorphological classification, which categorizes estuaries and the degree of mixing based on standard morphological categories (e.g., drowned river valleys)

Two situations exist where the observed spatial variations can be ignored. The first is when the primary location of water quality concern occurs in an area where these gradients are not important. A good example would be a nutrient modeling study to consider the impacts of a discharge on phytoplankton. If it takes 2 miles for a bankside discharge to undergo complete lateral mixing, but the location of maximum algal density is 5 miles downstream, lateral variability in water quality need not be described by the model. The second situation where a known gradient need not be modeled is where management objectives are not concerned with the gradient. Examples of this include water quality standards that are expressed on a spatially averaged basis.

The choice of spatial scale and layout of the model network requires considerable judgment. Knowledge of the regulatory problem must be combined with knowledge of the loading, transport, and transformation processes and an understanding of the model chosen to perform the simulations. Competing factors often must be balanced, such as precision and cost, or the better fit of one section of the network versus another.

Temporal Extent and Scale

The temporal resolution of water quality models falls into one of two broad categories, steady state or dynamic (i.e., time-variable). Steady-state models predict pollutant concentrations that are expected to result from a single set of loading and environmental conditions. Dynamic models predict changes in water quality over time in response to time-variable loads and environmental conditions.

Steady-state models are much easier to apply and require considerably fewer resources than dynamic models. This ease of application makes them the preferred modeling framework when loading to the system can be assumed to be constant and information on changes in concentration over time is not required. Potential uses of steady-state models include calculation of seasonal average total nutrient concentrations in response to seasonal average loads. Steady-state models also have been used to predict "critical condition" low-flow phytoplankton densities. Steady-state models are poorly suited for evaluating individual intermittent events (i.e., runoff) but can be used to evaluate the cumulative effect of multiple, intermittent events on a seasonal basis.

The timeframe to be represented for a particular steady-state simulation (e.g., monthly average, seasonal average) must be longer than the water residence time (flushing time) of the estuary. The water residence time is the time required to remove a parcel of water from an upstream location in an estuary. Factors that control flushing include tidal action, freshwater inflow, and wind stress. Typical flushing times range from days in small estuaries, or those dominated by tributary flow, to months in large estuaries during low tributary flow conditions. Several formulae have been used to estimate flushing times. The Fraction of Fresh Water Method, the Tidal Prism Method, and the Modified Tidal Prism Method are fully discussed in Mills et al. (1985) and briefly described in the following section.

Dynamic models should be used when information on changes in concentration over time is required. Dynamic models can be divided into two categories, quasi-dynamic and fully dynamic. Quasi-dynamic simulations predict variations on the order of days to months. The effects of tidal transport are time-averaged, and net or residual flows are used. Fully dynamic simulations predict hour-to-hour variations caused by tidal transport.

The duration of dynamic simulations can range from days to years, depending on the size and transport characteristics of the study area, the reaction kinetics and forcing functions of the water quality constituents, and the strategy for relating simulation results to the regulatory requirements. One basic guideline applies in all cases: the simulations should be long enough to eliminate the effect of initial conditions on important water quality constituents at critical locations. Flushing times provide the minimum duration for simulations of dissolved, nonreactive pollutants. The annual sunlight and temperature cycles almost always require that eutrophication simulations range from seasons to years.

Predicting the year-to-year eutrophication response of large estuaries is best accomplished by quasi-dynamic simulations. In general, if the regulatory need or kinetic response is on the order of hours, then fully dynamic simulations are required; if regulatory needs are long-term averages and the kinetic response is on the order of seasons to years, then quasi-dynamic or steady-state simulations are preferred.

Model Selection

The goal of model selection is to obtain a simulation model that effectively meets all study objectives. In the final analysis, how a model is used is more important to its success than exactly which model is used. Nevertheless, although selection of an appropriate model will not guarantee success, it will help. Selection of an inappropriate model will not guarantee failure, but will render a successful outcome more difficult.

Models may be classified in different and somewhat arbitrary ways. Some models may not quite fit in any category, or may fit well in several. In addition, models tend to evolve with use. The exact capabilities of the individual models described here may change. In particular, pollutant fate processes may be modified. Usually the computational framework and the basic transport scheme remain stable over time. For this reason, transport characteristics will provide the basis for the model classification scheme used here. Models selected for discussion here are general purpose, in the public domain, and available from or supported by public agencies.

9.3 MODEL CLASSIFICATION

Estuarine and near-coastal models consist of two components: hydrodynamics and water quality. Although the hydrodynamic component is independent of the water quality component, water quality depends on the transport processes controlled by hydrodynamics. As a result, estuarine models can be classified as Level I to Level IV, according to the temporal and spatial complexity of the hydrodynamic component of the model.

Level I includes desktop screening methodologies that calculate seasonal or annual mean total nutrient concentrations based on steady-state conditions and simplified flushing time estimates. Steady-state models use an unvarying flow condition that neglects the temporal variability of tidal heights and currents. These models are designed for relatively simple screening level analyses. They also can be used to highlight major water quality issues and important data gaps in the early model-identification stage of a more complex study.

Level II includes computerized steady-state or tidally averaged quasi-dynamic simulation models, which generally use a box or compartment-type network. Tidally averaged models simulate the net flow over a tidal cycle. These models cannot predict the variability and range of nutrient concentrations throughout each tidal cycle, but they are capable of simulating variations in tidally averaged concentrations over time. Level II models can predict slowly changing seasonal water quality with an effective time resolution of 2 weeks to 1 month.

Level III includes computerized one-dimensional (1-d) and quasi two-dimensional (2-d) dynamic simulation models. These real-time models simulate variations in tidal heights and velocities throughout each tidal cycle. One-dimensional models treat the estuary as well-mixed vertically and laterally. Quasi 2-d models employ a link-node approach that describes water quality in two dimensions (longitudinal and lateral) through a network of 1-d nodes and channels. Tidal movement is simulated with a separate hydrodynamic package in these models. The required data and modeling resources typically are unavailable to support models of Level III or above on a widespread basis.

Level IV consists of computerized 2-d and 3-d dynamic simulation models. Dispersive mixing and seaward boundary exchanges are treated more realistically than in the Level III 1-d models. These models are almost never used for routine nutrient assessment, because of excessive resource requirements.

Level I Models

Level I desktop methodologies may be employed using a hand-held calculator or computer spreadsheet and are based on steady-state conditions, first-order decay coefficients, simplified estimates of flushing time, and seasonal pollutant concentrations. The EPA screening methods provide a series of Level I analyses as described below.

EPA Screening Methods

WQAM is a set of steady-state desktop models that includes both one-dimensional and two-dimensional box model calculations (Mills et al. 1985). Specific techniques contained in WQAM are the Fraction of Freshwater Method, the Modified Tidal Prism Method, Advection-Dispersion Equations, and Pritchard's Two-Dimensional Box Model.

Fraction of Freshwater Method

The Fraction of Freshwater Method estimates pollutant concentrations in one-dimensional estuaries from information on freshwater and tidal flow by comparing salinity in the estuary with salinity in the local seawater. The fraction of freshwater at any location in the estuary is calculated by comparing the volume of freshwater at that location with the total volume of water:

$$fx = \frac{S_s - S_x}{S_s} \qquad (9\text{-}1)$$

where f_x = fraction of freshwater at location x, S_s = seawater salinity at the mouth of the estuary, and S_x = salinity at location x.

This ratio can be viewed as the degree of dilution of the freshwater inflow (as well as pollutants) by seawater. With this in mind, the total dilution of a pollutant input can be calculated by multiplying the seawater dilution by the freshwater dilution. This then provides a simple way to calculate concentrations of conservative pollutants. For any location x, at or downstream of the discharge, pollutant loads are diluted by tidal mixing and upstream flows. The amount of dilution can be calculated by:

$$C_x = f_x \frac{W}{Q} \qquad (9\text{-}2)$$

where C_x = constituent concentration at location x at or downstream of discharge, f_x = fraction of freshwater at location x, W = waste loading rate (mass/time), and Q = freshwater inflow (volume/time).

The right side of Equation 9-2 can be divided into two distinct terms. The term W/Q represents the classical equation for determining dilution in rivers caused by upstream flow. The second term, f_x, accounts for the further dilution of the river concentration by tidal influx of seawater.

Concentrations upstream of the discharge are estimated from the concentration at the point of mix and the relative salinity of the upstream location. The upstream concentrations are assumed to be diluted by freshwater to the same degree that salinity is diluted. The equation is:

$$C_x = f_d \frac{W}{Q} \frac{S_x}{S_d} \qquad (9\text{-}3)$$

where f_d = fraction of freshwater at discharge location, S_x = salinity at upstream location x, and S_d = salinity at discharge location.

The fraction of freshwater at the discharge location, f_d, is determined by applying Equation 9-1 at the discharge location. Equation 9-3 can be modified to assess the impact of nutrients entering from the seaward boundary by replacing the leading $f_d W/Q$ term with the boundary nutrient concentration.

Cumulative pollutant impacts from multiple sources are obtained through a two-step process. First, pollutant concentration caused by each source independent of all other discharges must be determined. This determination is accomplished by applying Equation 9-2 or 9-3, one discharge at a time, for any estuary location of interest. The second step is to determine the total concentration at that location. This determination is accomplished by adding all of the incremental concentrations caused by each discharge, as calculated in the first step. This process can be repeated for any location of interest.

The Fraction of Freshwater Method can be used to predict cumulative impacts in one-dimensional (i.e., narrow) systems with significant freshwater inflow. Upstream freshwater flow must be large with respect to total pollutant inflow for this method to be applicable. The method assumes conservative pollutant behavior. It is consequently best used to investigate total nitrogen concentrations, because overall loss rates of total nitrogen from the water column generally are small.

Modified Tidal Prism Method

The Modified Tidal Prism Method estimates dilution from the total amount of water entering an estuary. It is more powerful than the Fraction of Freshwater Method because it can consider not only tidal dilution but also nonconservative reaction losses. It is best applied to investigate total nutrient concentrations, but provides additional flexibility to describe pollutant losses that may occur through settling or denitrification.

The method divides an estuary into segments whose lengths and volumes are calculated using low-tide volumes and tidal inflow. The tidal prism (i.e., total volume of tidal inflow) is compared for each segment with the total segment volume to estimate flushing potential in that segment over a tidal cycle. The Modified Tidal Prism Method assumes complete mixing of the incoming tidal flow with the water resident in each segment.

The Modified Tidal Prism Method requires seven inputs:

- Freshwater inflow to the estuary
- Salinity of seawater at the downstream boundary
- Pollutant loading rate
- Salinity of each segment
- Low-tide volume for each segment
- Intertidal volume (tidal prism) for each segment
- First-order constituent loss rate for each segment

The first step of the method is to segment the estuary. This requires an initial time-consuming step of dividing the estuary into segments with lengths equal to the distance traveled by a particle over a tidal cycle. Cumulative subtidal and intertidal water volumes must be plotted for the estuary, and a graphical procedure is used to define model segmentation. Once the estuary has been segmented, a series of calculations can be performed to estimate constituent concentrations in each segment. Specific methods for dividing the estuary and performing the calculations are provided in Mills et al. (1985).

Advection-Dispersion Equations
Analytical equations have been developed to predict the concentration of nonconservative constituents in one-dimensional estuaries. These types of equations consider the processes of net seaward flows (advection) and tidally averaged mixing (dispersion), as well as simple decay. They can be used to predict total nutrient concentrations at various locations in an estuary in response to alternative nutrient loading rates. One-dimensional advection-dispersion equations can be expressed in several different forms (O'Connor 1965), with the most common form contained in the water quality assessment methodology. These equations require numerous simplifying assumptions, such as constant geometry and tidal mixing along the length of the estuary, but have proven to be a useful screening tool.

The advection-dispersion equations require five inputs: upstream freshwater flow rate (*R*), constituent loading rate (*W*), estuarine cross-sectional area (*A*), tidally averaged dispersion coefficient (*E*), and first-order decay rate coefficient (*k*). The first three inputs can be measured directly. The latter two inputs must be determined indirectly through the model calibration process described below. Two equations are provided, one which predicts concentrations at any distance (*x*) upstream of the discharge of concern and another for concentrations at any distance seaward of the discharge. Co is the concentration at the point of discharge. The equations are:

$$C = C_0 e^{j_2 x} \quad X > 0 \text{ (down estuary)} \qquad (9\text{-}4)$$

$$C = C_0 e^{j_1 x} \quad X < 0 \text{ (up estuary)}$$

where:

$$j_2 = \frac{R}{2AE}\left(1 - \sqrt{1 + \frac{4kEA^2}{R^2}}\right)$$

$$j_1 = \frac{R}{2AE}\left(1 + \sqrt{1 + \frac{4kEA^2}{R^2}}\right)$$

$$C_0 = \frac{W}{R}\frac{1}{\sqrt{1 + (4kE / U^2)}}$$

These equations can be used to evaluate multiple loading sources by independently applying Equation 9-4 for each loading source and summing the predicted concentrations across the estuary.

Pritchard's Two-Dimensional Box Model
Vertically stratified estuaries add a significant degree of complexity to the modeling analysis. Pritchard (1969) developed a relatively simple approach, which can predict nutrient concentration distributions along the length of an estuary in both an upper and lower layer. This approach is based on numerous simplifying assumptions, including:

- Steady-state conditions
- Conservative pollutant behavior
- Uniform constituent concentration within each layer or each segment

The following information is required: (1) freshwater flow rate into the head of the estuary, (2) pollutant mass loading rates, and (3) longitudinal salinity profiles along the length of the estuary in the upper and lower layers. The method solves a series of linear equations describing the salinity balance around each segment to determine net flows and dispersion between each segment. Specific methods for performing the calculations are provided in Mills et al. (1985).

Results from Pritchard's model can be used to directly calculate conservative constituent concentrations throughout the estuary or to serve as the hydrodynamic input to one of the Level II models described below.

Level II Models

Level II models include computerized steady-state and tidally averaged simulation models that generally use a box or compartment-type network. Steady-state models are difficult to calibrate in situations where hydrodynamics and pollutant releases vary rapidly. Consequently, these models are less appropriate when waste load, river inflow, or tidal range vary appreciably with a period close to the flushing time of the waterbody. Level II models are the simplest models available that are capable of describing the relationship between nutrient loads and some of the endpoints of concern of the eutrophication process (i.e., chlorophyll *a*, minimum dissolved oxygen).

The Level II models by EPA are QUAL2E and the Water Quality Analysis Simulation Program (WASP5), with its associated eutrophication program EUTRO5.

QUAL2E

QUAL2E is a steady-state, one-dimensional model designed for simulating conventional pollutants in streams and well-mixed lakes (U.S. EPA 1995) and is not recommended for estuaries. Rather, QUAL TX, which allows tidal boundary conditions, may be more appropriate, but documentation on this model is very sparse.

WASP6.0

The Water Quality Analysis Simulation Program (WASP6.0) is a general, multidimensional model that utilizes compartment modeling techniques (Ambrose et al. 1993). The equations solved by WASP6.0 are based on the principle of conservation of mass. Operated in the quasi-dynamic mode, WASP6.0 requires the user to supply initial segment volumes, network flow fields, and inflow time functions. The user also must calibrate dispersion coefficients between compartments. WASP6.0 has the capability of simulating nutrient-related water quality issues at a wide range of complexity.

EUTRO5

EUTRO5 is the submodel in the WASP6.0 system that is designed to simulate conventional pollutants. EUTRO5 combines a kinetic structure adapted from the Potomac Eutrophication Model with the WASP transport structure. EUTRO5 predicts DO, carbonaceous BOD, phytoplankton carbon and chlorophyll *a*, ammonia, nitrate, organic nitrogen, organic phosphorus, and orthophosphate in the water column and, if specified, the underlying bed. In addition to segment volumes, flows, and dispersive exchanges, the user must supply deposition and resuspension velocities for organic solids, inorganic solids, and phytoplankton. Rate constants and half-saturation coefficients for the various biochemical transformation reactions must be specified by the user. Finally, the time- and/or space-variable environmental forcing functions, such as light intensity, light extinction, wind speed, cloud cover, temperature, and benthic fluxes, must be input.

Level III Models

Level III includes computerized 1-d and 2-d models that simulate variations in tidal height and velocity throughout each tidal cycle. Level III models enable characterization of phenomena varying rapidly within each tidal cycle, such as pollutant spills, stormwater runoff, and batch discharges. Level III models also are deemed appropriate for systems where the tidal boundary impact, as a function of the hydrodynamics and water quality, is important to the modeled system within a tidal period.

Tidally varying (intratidal) models have found most use in the analysis of short-term events, in which the model simulates a period of time anywhere from one tidal cycle to a month. Some seasonal simulations also have been conducted.

In using Level III models, one must decide whether a 1-d longitudinal system is sufficient, or whether a 2-d model is required to capture the longitudinal and lateral variations in the estuary. For estuaries whose channels are longer than their width and reasonably well mixed across their width, a 1-d model may be chosen. If large differences exist in water quality from one side of an estuary to the other, or if vertical stratification is important, then a 2-d model is appropriate.

All Level III models considered here can simulate nutrient-eutrophication interactions. These models also include settling rates and benthic flux rates for several different constituents, such as phosphorus, nitrogen, and sediment oxygen demand. The Level III model distributed by EPA is the WASP6.

Level IV Models

Level IV includes a variety of computerized 2-d and 3-d dynamic simulation models. Dispersive mixing and seaward boundary exchanges are treated more realistically than in the Level III 1-d models. Although not routinely used in nutrient analyses, they are now finding use by experts in special studies. Level IV models are required when variations in concentrations in all three dimensions are of concern. The time-variable nature of a Level IV model ensures the need for a time-variable watershed model in order to provide for the nonpoint source inputs. Fully 3-d models that can predict longitudinal, lateral, and vertical transport are the most complex and expensive to set up and run.

At present, no Level IV model is supported by EPA. Three current Level IV models, CE-QUAL-W2, Integrated Compartment Model (ICM), and EFDC, are described below.

CE-QUAL-W2

CE-QUAL-W2 is a dynamic 2-d (x-z) model developed for stratified waterbodies (Environmental and Hydraulics Laboratories 1986). This is a U.S. Army Corps of Engineers modification of the Laterally Averaged Reservoir Model (Edinger and Buchak 1983; Buchak and Edinger 1984a,b). CE-QUAL-W2 consists of directly coupled hydrodynamic and water quality transport models. Hydrodynamic computations are influenced by variable water density caused by temperature, salinity, and dissolved and suspended solids. Developed for reservoirs and narrow, stratified estuaries, CE-QUAL-W2 can handle a branched and/or looped system with flow and/or head boundary conditions. With two dimensions depicted, point and nonpoint loadings can be distributed spatially.

CE-QUAL-W2 simulates as many as 20 other quality variables. Primary physical processes included are surface heat transfer, shortwave and longwave radiation and penetration, convective mixing, wind- and flow-induced mixing, entrainment of ambient water by pumped-storage inflows, inflow density current placement, selective withdrawal, and density stratification as influenced by temperature and dissolved and suspended solids. Major chemical and biological processes in CE-QUAL-W2 include the effects on DO of atmospheric exchange, photosynthesis, respiration, organic matter decomposition, nitrification, and chemical oxidation of reduced substances; uptake, excretion, and regeneration of phosphorus and nitrogen and nitrification-denitrification under aerobic and anaerobic conditions; carbon cycling and alkalinity-pH-CO_2 interactions; trophic relationships for total phytoplankton; accumulation and decomposition of detritus and organic sediment; and coliform bacteria mortality.

CH3D-ICM

CH3D is a 3-d, finite-difference hydrodynamic model, developed by the U.S. Army Corps of Engineers Waterways Experiment Station (WES) in Vicksburg, MS. Results from CH3D have been linked to the ICM to model water quality in the Chesapeake Bay. The ICM was developed as the integrated-compartment eutrophication model component of the Chesapeake Bay model package. The model contains detailed eutrophication kinetics, modeling the carbon, nitrogen, phosphorus, silica, and dissolved oxygen cycles.

CH3D-ICM is a linkage of CH3D, a hydrodynamic model, and ICM, a water quality model. CH3D is a hydrodynamic model developed for the Chesapeake Bay Program (Johnson et al. 1991). The model can be used to predict system response to water levels, flow velocities, salinities, temperatures, and the three-dimensional velocity field. CH3D makes hydrodynamic computations on a curvilinear or boundary-fitted platform grid. Deep navigation channels and irregular shorelines can be modeled because of the boundary-fitted coordinates feature. Vertical turbulence is predicted by the model and is crucial to a successful simulation of stratification, destratification, and anoxia. A second-order model based on the assumption of local equilibrium of turbulence is employed.

ICM is a finite-difference water quality model that may be applied to most waterbodies in one, two, or three dimensions (Cerco and Cole 1995). The model predicts time-varying concentrations of water quality constituents and includes advective and dispersive transport. The model also considers sediment diagenesis benthic exchange. ICM incorporates detailed algorithms for water quality kinetics. Interactions among state variables are described in 80 partial-differential equations that employ more than 140 parameters. An improved finite-difference method is used to solve the mass conservation equation for each cell in the computational grid and for each state variable.

EFDC

EFDC is a linked three-dimensional, finite-difference hydrodynamic and water quality model developed at the Virginia Institute of Marine Sciences (Hamrick 1996). EFDC contains extensive water quality capabilities, including a eutrophication framework based on the ICM model. EFDC is a general-purpose hydrodynamic and transport model that simulates tidal, density, and wind-driven flow; salinity; temperature; and sediment transport. Two built-in, full-coupled water quality/eutrophication submodels are included in the code.

EFDC solves the vertically hydrostatic, free-surface, variable-density, turbulent-averaged equations of motion and transport; transport equations for turbulence intensity and length scale, salinity, and temperature in a stretched, vertical coordinate system; and horizontal coordinate systems that may be Cartesian or curvilinear-orthogonal. Equations describing the transport of suspended sediment, toxic contaminants, and water quality state variables also are solved.

The model uses a finite-difference scheme with three time levels and an internal-external mode splitting procedure to achieve separation of the internal shear, or baroclinic, mode from the external free-surface gravity wave, or barotropic, mode. An implicit external-mode solution is used with simultaneous computation of a two-dimensional surface elevation field by a multicolor successive overrelaxation procedure. The external solution is completed by calculation of the depth-integrated barotropic velocities using the new surface elevation field. Various options can be used for advective transport, including the "centered in time and space" scheme and the "forward in time and upwind in space" scheme.

Summary of Model Capabilities

The important features of the models selected for discussion in this manual are summarized in Tables 9-1 and 9-2. The information provided in these tables is primarily qualitative and sufficient to determine whether a model may be suitable for a particular application. For complete information, consult the appropriate user's manuals, the supporting agency, and other experienced users.

9.4 USE OF MODELS FOR NUTRIENT INVESTIGATION

This section describes procedures for using models to perform nutrient assessment in estuaries and near-coastal waters. It describes the model calibration and validation process, where model parameters that best describe the waterbody of interest are selected. In addition, guidance is provided on using models for nutrient management and assessment.

The first subsection describes a general procedure for calibrating nutrient models, and briefly describes the validation procedure used to estimate the uncertainty of such models. The subsection also describes some statistical methods for testing the calibrated models. These methods are useful to aid in the various calibration phases and also in the validation phase to measure how well model predictions and measurements of water quality agree.

The second subsection provides guidance on the management application of a calibrated model. Methods to project effects of changes in waste loads and to determine causes of existing conditions are discussed. Finally, a case study application is provided.

Model Calibration and Validation

Model calibration is the process of determining model parameters most appropriate for a given site-specific application. Calibration of a model involves a comparison of the measured and simulated receiving water quality conditions. The nature of the model calibration process depends upon the complexity of the model selected. Simpler models contain relatively few parameters that need to be calibrated, whereas more complex models contain many.

Calibration alone is not adequate to determine the predictive capability of a model for a particular estuary. To map out the range of conditions over which the model can be used, one or more additional independent sets of data are required to determine whether the model is predictively valid. This model validation exercise defines the limits of usefulness of the calibrated model. Without validation testing, the model merely describes the conditions defined by the calibration data set. The degree of uncertainty of any projection or extrapolation of the model remains unknown.

Table 9-1. Basic model features

Methods/Model	Time Scales	Spatial Dimensions	Hydro-dynamics	Data Expertise Requirements	Distributing Agency	Scale of Effort
Fraction of Freshwater	SS	1D	0	Minimal	EPA	Days
Modified Tidal Prism	SS	1D	0	Minimal	EPA	Days
Advection-Dispersion Equations	SS	1D	0	Minimal	EPA	Days
Pritchard's 2-D Box Model	SS	2D (xz)	0	Minimal	EPA	Days
QUAL2E	SS	1D	I	Moderate	EPA	Few months
WASP5	Q/D	1D, 2D (xy), or 3D	I, S	Moderate to substantial	EPA	Few months
CE-QUAL-W2	D	2D (xz)	S	Substantial	Army Corps	Several months
CH3D-ICM	D	3D	S	Extreme	EPA	Months to years
EFDC	D	3D	S	Extreme	EPA	Months to years

D - dynamic; Q - quasi-dynamic (tidal-averaged); SS - steady state; x - 1-dimensional, xy - 2-dimensional, longitudinal-lateral; xz - 2-dimensional, longitudinal-vertical; xyz - 3-dimensional; B - compartment or box 3d; xx - link node branching 2d; 0 - No hydraulics specified, inferred from salinity data; I - hydrodynamics input; S - hydrodynamics simulated.

In general, models are calibrated in phases, beginning with the selection of the model parameters and coefficients that are independent of parameters to be calibrated later. For purposes of this discussion, the process is divided into the categories of hydrodynamic calibration and water quality calibration. The discussion covers the parameters that need to be calibrated for each level of model as well as the specific model outputs to be used for the calibration comparison. Calibration of the more complex models requires detailed guidance; the reader is referred to other documents (e.g., U.S. EPA 1990b; Thomann and Mueller 1987) for a discussion that is more comprehensive than is feasible here.

Hydrodynamic Calibration

The first phase of calibration concentrates on the hydrodynamic and mass transport models. Two Level I models, the Fraction of Freshwater Method and the Tidal Prism Method, have no hydrodynamic parameters that require calibration. In these simplest cases, all hydrodynamic and mass transport processes are implicitly considered via specification of observed salinity values. Although there are no parameters to calibrate for these models, there is merit in testing the model's predictive validity by comparing predicted concentrations with field observations of a conservative (i.e., nondecaying) substance, if such data are available.

For the remaining Level I and Level II models, only one hydrodynamic parameter requires calibration: the tidal dispersion coefficient. It is possible to calibrate the hydrodynamic and mass transport portions of these models by determining values for this coefficient that best describe observed salinity or conservative tracer measurements.

Table 9-2. Key features of selected models

Model	Key Features	Advantages	Disadvantages/ Limitations
WQAM	Simplified equations to simulate dilution, advection, dispersion, first-order decay, empirical relationships between nutrient loading, and total nutrient concentration	Few data requirements; can be employed easily with a hand calculator or computer spreadsheet	Limited to screening and midlevel applications
QUAL2E	Steady-state model provides adequate simulation of water quality processes, including DO-BOD and algal growth cycles	User-friendly Windows interface; widely used and accepted; able to simulate all of the conventional pollutants of concern	Limited to simulation of time periods during which stream flow and input loads are essentially constant
WASP5	Based on flexible compartment modeling approach; can be applied in 1, 2, or 3 dimensions	Has been widely applied to estuarine situations; considers comprehensive DO and algal processes; can be used in 3-d simulations by linking with hydrodynamic models	Coupling with multi-dimensional hydrodynamic models requires extensive site-specific linkage efforts
CE-QUAL-W2	Uses an implicit approach to solve equations of continuity and momentum; simulates variations in water quality in the longitudinal and lateral directions	Simulates the onset and breakdown of vertical stratification; most appropriate where vertical variations are an important water quality consideration	Application requires extensive modeling experience
CH3D-ICM	Finite-difference model can be applied to most water bodies in 1 to 3 dimensions; predicts time-varying concentrations of constituents; includes advective and dispersive transport	State-of-the-science eutrophication kinetics	Computationally intensive; requires extensive data for calibration and verification; requires a high level of technical expertise to apply effectively
EFDC	Linked 3-d, finite-difference hydrodynamic and water quality model contains extensive water quality capabilities; water quality concentrations can be predicted in a variety of formats suitable for analysis and plotting	3-d description of water quality parameters of concern; entire range of hydrodynamic, sediment, eutrophication, and toxic chemical constituents can be considered	Computationally intensive; requires extensive data for calibration and verification; requires a high level of technical expertise to apply effectively

Level III models typically contain two calibration parameters, the channel roughness coefficient and the dispersion coefficient. Occasionally these models are calibrated with current velocity and water surface elevation data, but more often are indirectly calibrated from salinity or conservative tracer measurements that also must be used to calibrate the mass transport model. Indirect calibration can result in an imprecise description of both the circulation and mass transport algorithms, but this is not a severe drawback unless the critical water quality components of the waste load allocation model are sensitive to small changes in circulation and mass transport.

Level IV hydrodynamic models contain several calibration parameters, including bottom and surface friction coefficients; vertical, lateral, and horizontal eddy viscosity coefficients; and wind speed coefficients. Calibration efforts for these types of models are beyond the scope of this document.

Kinetic Process Calibration

The second phase of calibration involves selection of the set of kinetic coefficients describing nutrient cycles and other aspects of the eutrophication process. Again, the effort required is directly related to the complexity of the model selected.

Two Level I models—the Fraction of Freshwater Method and Pritchard's model—have no kinetic parameters that require calibration. The models assume that constituent concentrations undergo no kinetic processes that affect their concentration, and typically are appropriate only for estimating total nitrogen concentrations. The remaining Level I models can describe nonconservative constituents, and lump all kinetic processes into a single overall decay coefficient. Model calibration in these cases consists of a comparison of predicted versus observed total nutrient concentrations.

The calibration of higher level nutrient and phytoplankton models requires significant expertise because of the complexity of the interactions between a number of the components of the cycles involved.

Coefficients that require calibration in these models pertain to: transformation rates among various forms of a given nutrient; maximum phytoplankton growth rates; phytoplankton respiration rates; phytoplankton growth sensitivity to light and nutrients; and phytoplankton and detrital settling velocities.

Model Validation

Validation testing is designed to confirm that the calibrated model is useful at least over the limited range of conditions defined by the calibration and validation data sets. The procedure is not designed to validate a model as generally being useful in every estuary, or even as useful over an extensive range of conditions found in a single estuary. Validation, as employed here, is limited strictly to indicating that the calibrated model is capable of producing valid results over a limited range of conditions. Those conditions are defined by the sets of data used to calibrate and validate the model. As a result, it is important that the calibration and validation data cover the range of conditions over which predictions are desired.

Validation testing is performed with an independent data set collected during a second field study. The field study may occur before or after the collection of calibration data. For the best results, however, the validation data should be collected after the model has been calibrated. This schedule of calibration and validation ensures that the calibration parameters are fully independent of the validation data. Often it is difficult to assemble the necessary resources to conduct the desired number of surveys. Therefore, it is important that surveys be scheduled in an innovative manner and the choice of calibration and validation data sets remain flexible to make the test of the calibrated model as severe as possible.

Too often, limited studies attempt calibration but not validation. This approach, in effect, limits the study to describing the conditions during the calibration data collection period and increases the uncertainty associated with the waste load allocation. In fact, model prediction uncertainty cannot be reliably assessed in these cases.

Model Testing

During and after the calibration and validation of a model, at least two types of testing are important. First, throughout the calibration procedure, a sensitivity test helps determine which parameters and coefficients are the most important. Second, a number of statistical tests help define the extent of agreement between model simulations and measured conditions.

The sensitivity analysis is simply an investigation of how much influence changes in model coefficients have on simulated results. Typically, important coefficients, parameters, boundary conditions, and initial conditions are varied by a positive or negative constant percentage to see what effect the change has on critical predictions. The coefficients and parameters are changed one at a time and the effects typically are ranked to show which parameters have the most influence and which have the least.

The second type of testing involves assessment of the “goodness of fit” for model simulations, compared with measurement of important water quality parameters. In addition to making a visual assessment, a number of statistical tests have proven useful. These include root mean square error, relative error, and regression analysis. Other more detailed statistical analyses are described in U.S. EPA (1990b).
The root mean square (rms) error is a criterion that is widely used to evaluate the agreement between model predictions. The rms error can be defined as:

$$rms = \left[\sum \left(C_m - C_s \right)^2 N \right]^{0.5} \qquad (9\text{-}5)$$

where C_m = measured concentration, C_s = simulated concentration, and N = number of measurements.

The rms error can be used to compute simultaneous discrepancies at a number of points, or it can be used to compute discrepancies between measurements and predictions at a single point over time. Global rms errors can be computed for a series of measurements at multiple points over time.

When discrepancies between model simulations and measurements are not uniform over parts of the estuary or over time, the relative error may be a more appropriate statistic for testing calibration or validation. The relative error is defined as:

$$e = \left|\overline{C}_m - \overline{C}_s\right| \Big/ \overline{C}_m \qquad (9\text{-}6)$$

where the overbars denote the average measured or simulated value. Averages can be performed over multiple sites or over time. The relative error behaves poorly for small values of measurements if discrepancies are not proportional to the magnitude of the measurement (i.e., small values of C_m magnify discrepancies) and if $C_m > C_s$ (as the maximum relative error is usually taken to be 100%). Therefore, the relative error is best for computing composite statistics when discrepancies are not constant, as may occur when calibration over an extensive range is attempted.

Regression analysis is very useful in identifying various types of bias in predictions of dynamic-state variables. The regression equation is written as:

$$C_m = a + bC_s + \varepsilon \qquad (9\text{-}7)$$

where a = intercept value, b = slope of the regression line, and ϵ = the error in measurement mean, C_m. The standard linear regression statistics computed from the above equation provide insight into the goodness of fit for a calibration. The square of the correlation coefficient, r^2, measures the percent of the variance accounted for between measured and predicted values. The slope estimate, b, and intercept, a, can indicate any consistent biases in the model calibration. A model calibration that perfectly described all available data would have a correlation coefficient of 1.0, a slope of 1:1, and a zero intercept.

9.5 MANAGEMENT APPLICATIONS

Once the model is calibrated and validated, it can be used to simulate future conditions to determine effects of changes in waste loads or to investigate causes of existing problems. This section describes three types of management application: (1) load-response analysis, (2) determination of acceptable nutrient loads, and (3) investigation of causes of nutrient problems.

Load-Response Analysis

A load-response analysis consists of performing multiple model simulations using different loading rates and examining the water quality predicted for each simulation. The most common use of a model to investigate nutrients in estuaries is to determine the water quality throughout an estuary in response to changes in nutrient loads. Models are designed to predict water quality based on loadings and environmental conditions (Figure 9-2).

This type of analysis also requires specification of the environmental conditions (e.g., freshwater inflows, tidal conditions) to be considered. The results of the load-response analysis are directly related to the environmental conditions specified for the model simulation. For example, use of summer-average environmental conditions in the model will show the response in summer-average water quality to changes in loads. For the simplest Level I models, environmental conditions are specified implicitly through the use of salinity observations. Predictions from these models will correspond to the environmental conditions that were in effect when the salinity was measured. The more complex models require explicit definition of environmental condition and can be used to provide predictions for a wide range of environmental conditions.

Acceptable Nutrient Loads

The most common use for water quality models is to define allowable loads necessary to achieve water quality objectives. As seen in Figure 9-2, models predict water quality for a specified set of loads and environmental conditions. Determination of acceptable loads typically requires an iterative procedure, as shown in Figure 9-3. The first iteration consists of performing a model simulation using existing loads and comparing predicted water quality with objectives. Assuming that the existing loads result in unacceptable water quality, additional model simulations are performed using incremental reductions in nutrient loads until water quality objectives are achieved.

The approach shown in Figure 9-3 can be used to define necessary reductions in total loads as well as reductions in individual contributors to the total load.

The results of the above approach are highly dependent on the environmental conditions selected, as allowable loading rates can vary substantially across different environmental conditions. Two approaches are available for selecting critical conditions for use in defining allowable loads. These approaches are termed the critical conditions approach and the continuous simulation approach. In the critical conditions approach, a single set of environmental conditions is selected for analysis. These conditions typically represent critical or worst-case conditions, that is, those environmental conditions that will result in the poorest water quality for a given set of loads. The rationale for the critical conditions approach assumes that if loads are defined to meet water quality objectives during "critical" conditions, the same loads will result in attaining water quality objectives during most other conditions as well.

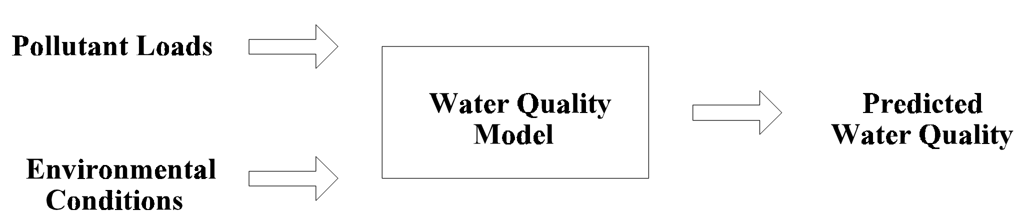

Figure 9-2. Use of models in load-response analysis.

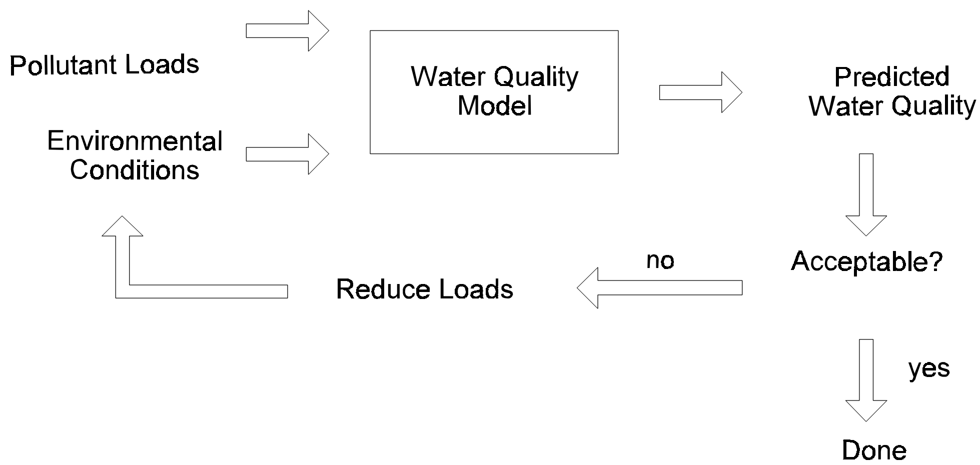

Figure 9-3. Use of models in determining allowable loads.

The continuous simulation approach performs simulations for as long a duration as is feasible, using historically observed variations in environmental conditions. The predicted water quality resulting from the continuous simulation is analyzed to determine the frequency with which water quality problems are observed to occur.

The overall intent of the modeling analysis is to define loads that will restrict the occurrence of water quality problems to an acceptable frequency. Each of the above two methods has particular strengths and weaknesses for performing this task. The continuous simulation approach provides a direct means to consider frequency of occurrence (i.e., number of years per problem) but has extreme resource requirements. The ability to perform continuous simulation of sufficient duration typically is constrained by the availability of data describing historical environmental conditions, or by computational requirements for the higher level models. The critical conditions approach has much more manageable resource requirements; however, there are no clear methods to establish appropriate critical conditions. In estuaries, freshwater, tides, wind, and other factors all can affect water quality. Selection of appropriate values for each environmental parameter requires considerable judgment. Furthermore, the specific level of protection associated with any single set of environmental conditions cannot be evaluated without performing a continuous simulation.

Case Study Example

Water quality models also can be used to gain an increased understanding of the relative importance of various loading sources to an estuary or near-coastal water. It is possible to investigate the contribution of individual loading sources to the water quality problem by performing a series of simulations examining each loading source separately. Because most water quality models assume a linear relationship between pollutant load and resulting water quality impact, it is possible to determine overall impacts to the estuary by summing the impacts from each source.

Investigation of Causes of Nutrient Problems

Shipps Creek (Figure 9-4) is a long, narrow, tidal tributary receiving nutrient inputs from upstream runoff and a single wastewater treatment plant (WWTP). This case study example demonstrates the use of models to perform three tasks:

- Estimate the contribution of various loading sources to the overall summer-average total nitrogen concentration

- Estimate the effect of a 50% reduction in loads from the WWTP on total nitrogen concentrations throughout the estuary

- Estimate the reduction in loading necessary to achieve an average total nitrogen concentration of 0.100 mg/L in the lower half of the estuary

Short-term answers were required, and screening-level accuracy was judged acceptable because of the short timeframe and limited data available.

The Fraction of Freshwater Method was selected because the estuary was considered one-dimensional, long-term average results were acceptable, and the water quality target was specified in terms of total nutrient levels.

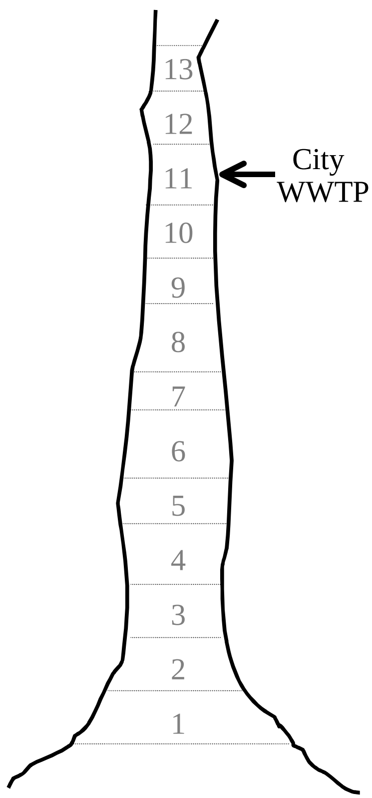

Figure 9-4. Shipps Creek site map and salinity monitoring location.

Information available to support the study included summer-average salinity measurements at 13 locations along the length of the estuary, summer-average freshwater inflows and total nitrogen loads to the estuary, nitrogen loads from the WWTP, and nitrogen and salinity concentrations outside of the estuary. This information was compiled as shown in Table 9-3 to allow implementation of the Fraction of Freshwater Method. The top two rows of Table 9-3 show the measured loading rates and seaward boundary conditions. The two leftmost columns define segments centered around each of the salinity measurements. The third column applies Equation 9-1 to calculate the fraction of freshwater in each segment.

The fourth through sixth columns in Table 9-3 apply Equations 9-2 or 9-3 as appropriate to determine the incremental contribution to the total nitrogen (TN) concentrations from each of three possible contributing sources: upstream (TN_{UP}), wastewater treatment plant (TN_{WWTP}), and the downstream seaward boundary (TN_{SEA}). Equation 9-2 is applied only to those segments downstream of the loading source, whereas Equation 9-3 is applied to those segments upstream of the loading source. Equation 9-2 is applied to all segments for examining impacts from upstream sources, whereas Equation 9-3 is applied to all segments for examining impacts from the seaward boundary. For determining WWTP impacts, Equation 9-2 is applied to segments 1-11 and Equation 9-2 is applied to segments 12-13. The final column in Table 9-3 sums the incremental contributions from each of the sources to provide a prediction of overall TN concentrations throughout the estuary.

Table 9-3. Calculation spreadsheet for Shipps Creek estuary

Freshwater Inflow		**Seawater Salinity**	**Upstream Load**	**WWTP Load**	**Seawater TN**	
Q = 100,000 cmd		**Ss = 30 ppt**	**W = 5,000 g/day**	**W = 500,000 g/day**	**0.005 ppm**	
Segment #	Salinity, S_i (ppt)	Fraction of Freshwater, f_I	TN_{UP} (mg/L)	TN_{WWTP} (mg/L)	TN_{SEA} (mg/L)	Overall TN (mg/L)
1	29	0.03	0.002	0.017	0.005	0.023
2	27	0.10	0.005	0.050	0.005	0.060
3	25	0.17	0.008	0.083	0.004	0.096
4	23	0.23	0.012	0.117	0.004	0.132
5	21	0.30	0.015	0.150	0.004	0.169
6	19	0.37	0.018	0.183	0.003	0.205
7	18	0.40	0.020	0.200	0.003	0.223
8	16	0.47	0.023	0.233	0.003	0.259
9	14	0.53	0.027	0.267	0.002	0.296
10	12	0.60	0.030	0.300	0.002	0.332
11	10	0.67	0.033	0.333	0.002	0.368
12	5	0.83	0.042	0.110	0.001	0.152
13	1	0.97	0.048	0.025	0.000	0.074

The results of this modeling analysis are shown graphically in Figure 9-5, showing the overall TN distribution as well as its component. Although the Fraction of Freshwater Method does not require calibration, it would be worthwhile at this point to confirm that the model predictions of TN throughout the estuary were consistent with observed data collected over the same time period. Figure 9-5 shows that the WWTP is the dominant source of nitrogen throughout most of the estuary. Upstream sources are the dominant component only at the extreme head of the estuary. Nitrogen contributions from the seaward boundary are small throughout the system. The results in Table 9-2 and Figure 9-4 satisfy the first objective of this study, which was to determine the contribution of various loading sources to the overall summer-average total nitrogen concentration. The second objective of the study was to determine the water quality resulting from a 50% reduction in WWTP TN loads. This was accomplished by reapplying Equations 9-2 and 9-3 using one-half of the original WWTP loads. Results of this analysis are shown in Figure 9-6, indicating a decrease in peak TN concentrations from 0.368 to 0.202 mg/L and a decrease in lower estuary (defined as segments 1-6) average concentrations from 0.111 to 0.064 mg/L. This nearly 50% reduction in concentrations was expected, because the original analysis had demonstrated that the WWTP was the dominant loading source to the estuary.

The final objective of the study was to determine the loading reductions necessary to achieve a lower estuary average concentration of 0.08 mg/L. No single answer exists to this question, because three separate sources of nitrogen to the estuary contribute to the total concentration. Analysis of the data in Table 9-3 shows that the incremental contribution of the upstream, WWTP, and seaward sources to lower estuary average concentrations were 0.010, 0.100, and 0.004 mg/L, respectively. Because the seaward

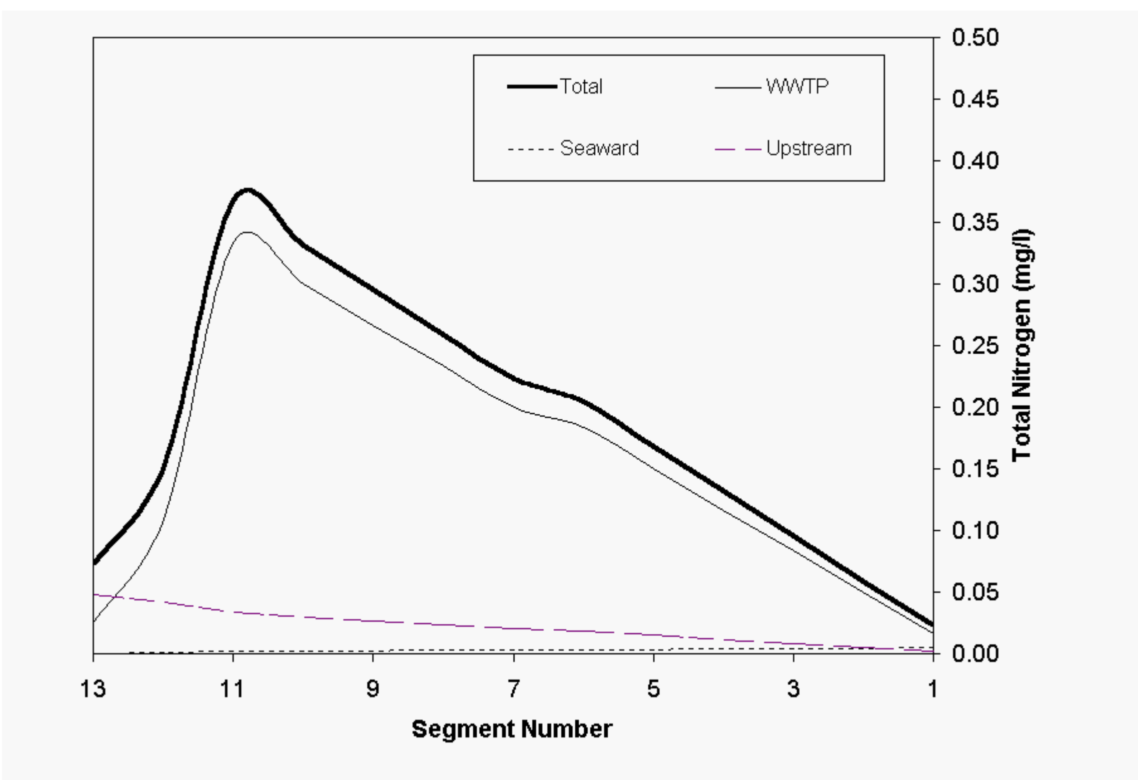

Figure 9-5. Model results for existing conditions.

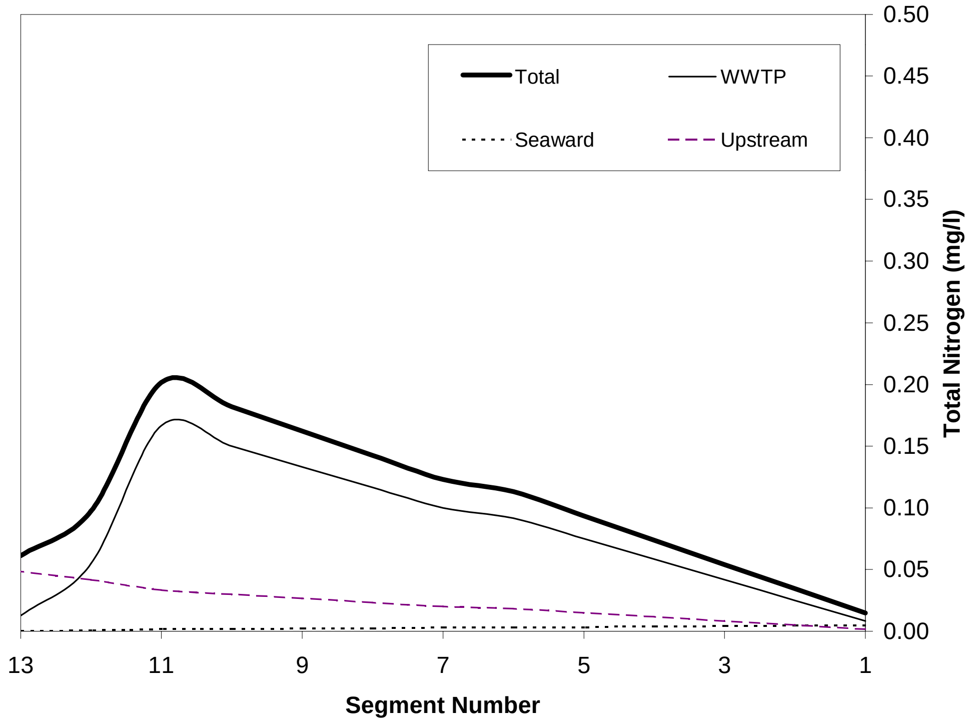

Figure 9-6. Model results for 50% reduction in WWTP load.

boundary nitrogen concentration cannot be controlled, management reductions must be restricted to either the upstream sources or the WWTP. The WWTP load must be reduced by at least 20% to meet the target TN concentration of 0.080 mg/L; otherwise, its contribution alone will exceed the target. Beyond the initial 20% reduction in the WWTP source, further reductions must come either from the WWTP or upstream sources. The specific allocation of these load reductions among sources is an economic and social decision that the model is not designed to address. The model is expressly designed, however, to test alternative proposals of load reductions to determine if they will meet the water quality objective. For example, a 25% reduction in both upstream and WWTP sources resulted in an average concentration of 0.087 mg/L (i.e., above the target), but a 30% reduction in WWTP loads coupled with a 40% reduction in upstream loads was shown to just meet the target.

All model simulations presented here should be viewed with extreme caution, because they are based on an uncalibrated, screening-level model. The level of uncertainty for these predictions cannot be quantified and is expected to be quite large. The model results do, however, provide the best possible estimate describing the relationship between nutrient loads and resulting concentrations, given the available resources.

Overview of Chesapeake Bay Airshed, Watershed, and Estuary Models

The cross-media models used in the Chesapeake Bay analysis consist of three models: an airshed model, a watershed model, and a model of the Chesapeake estuary. These models are linked so that the output of one simulation provides input data for another. The simulation period is the 10-year period of January

1, 1985, to December 31, 1994. Versions of these models have been used by the Chesapeake Bay Program for more than a decade and have been refined and upgraded several times.

Airshed Model

The Chesapeake Bay Program airshed model provides estimates of atmospheric deposition loads of nitrogen. A product of the EPA National Exposure Research Laboratory in Research Triangle Park, North Carolina, RADM (pronounced "radum") is an acronym for Regional Acid Deposition Model. RADM is a three-dimensional model that tracks nutrient emissions across the eastern United States. Two RADM grids meet various resolution needs. A large grid scale covers the entire RADM domain and contains 20,000 square cells of 6,400 square kilometers each. A fine grid scale covers the region of the Chesapeake Bay watershed and has 60,000 cells, each covering 400 square kilometers. The model domain in the vertical is 15 cells deep, reaching from ground level to the top of the free troposphere. The depth of the cells increases with altitude. One of the findings of the RADM model is that the Chesapeake Bay airshed, defined as the area accounting for 75% of the deposition in the watershed, is approximately 5.5 times the size of the watershed.

RADM is used to drive scenarios associated with reductions in atmospheric deposition of nitrogen. A base condition deposition represents an estimate of the current condition of atmospheric deposition in the watershed and is developed from a regression of National Atmospheric Deposition Program (NADP) data. RADM scenarios of atmospheric deposition reductions are incorporated into Watershed Model scenarios by adjusting the base NADP condition on a segment-by-segment basis with a percent change prescribed by RADM scenario results. Results from RADM specify loads of wet and dry deposition to the Chesapeake watershed for the State Implementation Program (SIP) and Limit of Technology scenarios. Deposition loads are input directly to the land surfaces of the watershed model or to the tidal water surface of the Chesapeake Bay estuary model package as daily loads of wet deposition (from rain washout of atmospheric nitrate and ammonia) and 12-year average loads of dry deposition.

Three atmospheric deposition loads were used for the Chesapeake analysis: (1) the base condition of atmospheric deposition, (2) the estimated atmospheric deposition of nitrogen equivalent to the 1998 SIP controls of atmospheric deposition, and (3) the estimated atmospheric deposition under full limit of technology control (see Table 9-4).

Table 9-4. Chesapeake watershed nitrogen deposition under varying management schemes for emissions of nitrogen atmospheric deposition precursors

Scenario	TN Deposition (millions of kg/year)
Base Condition	204
State Implementation Plan (SIP)	178
Limit of Technology	128

Sources: Chesapeake Bay Program Phase IV Watershed Model and RADM.

Watershed Model

The watershed model simulates nutrient and sediment loads delivered to the Chesapeake Bay from all areas of the watershed. Land uses of cropland, pasture, urban areas, and forests are simulated on an hourly time-step, tracing the fate and transport of input nutrient loads from atmospheric deposition, fertilizers, animal manures, and point sources. The simulation is an overall mass balance of nitrogen and phosphorus nutrients in the basin, so that the ultimate fate of input nutrients is simulated, either as incorporation into crop or forest plant material, incorporation into soil, or river runoff. Nitrogen fates included volatilization into the atmosphere and denitrification. Transport in rivers is simulated to the tidal waters of the Chesapeake Bay. Sediment is simulated as eroded material washed off land surfaces and transported to the tidal bay. Scenarios are run for 10 years on a 1-hour time-step, and results are aggregated into 10-year average loads for comparison among scenarios.

To simulate the delivery of nutrients and sediment to the bay, the watershed was divided into 86 major model segments, each with an average area of 194,000 hectares. Segmentation, based on three tiers of criteria, partitioned the watershed into regions of similar characteristics. The first criterion was segmentation of similar geographic and topographic areas, which were further delineated in terms of soil type, soil moisture holding capacity, infiltration rates, and uniformity of slope. The second criterion involved finer segmentation based on spatial patterns of rainfall. Each segment had a bank-full travel time of about 24–72 hours. The third criterion used to further delineate segments was based on features of the river reach. River reaches containing a reservoir were separated into a reservoir simulation and a river simulation of the free-flowing river. For example, the James basin had 11 model segments, 2 represented reservoirs on the James and Appomattox, and the segmentation generally became finer with closer proximity to tidal waters.

Model segments were located to take advantage of observed data locations, so that a model segment outlet was located close to monitoring stations. Water quality and discharge data were collected from Federal and State agencies, universities, and other organizations. More than 150 subsegments were used at the interface between the watershed and estuary models to accurately deliver flow, nutrient, and sediment loads to appropriate areas of the estuary. Increased simulation accuracy motivated the division of basins into multiple segments and into simulation time-steps of an hour, but all scenario results were reported at the level of the basin and for 10-year average loads.

The watershed model has been in continuous operation at the Chesapeake Bay Program since 1982 and has had many upgrades and refinements since that time. The watershed model used for this application was Phase 4.1 based on the HSPF Version 11 code (Hydrologic Simulation Program - Fortan - HSPF). Version 11 is a widely used public domain model supported by EPA, USGS, and the U.S. Army Corps of Engineers.

Estuary Model

The Chesapeake Bay Estuary Model Package (CBEMP) is actually several models simulating different aspects of water quality in the bay and tributaries. A water quality model simulates 22 parameters, or state variables, as listed in Table 9-5.

Table 9-5. Water quality state variables used in CBEMP

Temperature	Dissolved organic nitrogen
Salinity	Labile particulate organic nitrogen
Inorganic suspended sediments	Refractory particulate organic nitrogen
Diatoms	Total phosphorus
Cyanobacteria (blue-green algae)	Dissolved organic phosphorus
Other phytoplankton	Labile particulate organic phosphorus
Dissolved organic carbon	Refractory particulate organic phosphorus
Labile particulate organic carbon	Dissolved oxygen
Refractory particulate organic carbon	Chemical oxygen demand
Ammonium	Dissolved silica
Nitrate + nitrite	Particulate biogenic silica

Zooplankton were separated into two size classes: microzooplankton (>44 microns) and mesozooplankton (>202 microns).

Linked to the water quality model is a hydrodynamic model, simulating the hydrodynamics, or water movement, throughout the tidal estuary. The hydrodynamic model produced three-dimensional predictions of velocity, diffusion, surface elevation, salinity, and temperature on an intratidal time scale. The model grid of the hydrodynamic and water quality models consists of more than 10,000 cells.

The modeling process involves simulation of living resource parameters, such as dissolved oxygen, chlorophyll concentrations, and submerged aquatic vegetation (SAV). Computed parameters are compared with living resource standards, and an estimation is made of the degree to which computed conditions benefit the resources of interest (e.g., fish, oysters). In addition, the CBEMP includes the direct interactive simulation of SAV and water quality. Three phytoplankton groups were simulated.

Over seasonal time scales, the bay sediments are a significant source of dissolved nutrients to the overlying water column. The role of sediments in the systemwide nutrient budget is especially important in summer when seasonal low flows diminish riverine nutrient input. In addition, water temperatures enhance biological processes in the sediments, creating greater sediment oxygen demand. Bay sediments retain a long-term nutrient load “memory” of several years; that is, sediment nutrient fluxes to the water column are determined by organic nutrient inputs from several previous years. Therefore, the water quality model was coupled directly to a predictive benthic-sediment model. These two models interact at each time-step, with the water quality model delivering settled organic material to the sediment bed, and the benthic-sediment model calculating the flux of oxygen and nutrients to the water column.

Linked to the CBEMP are the watershed and airshed models, which provide daily input data. Generally, 10-year scenarios are run on 15-minute time-steps with output generated each 10 days. The estuary model has been in operation since 1987, with two major model refinements released since the initial 1987 steady-state model.

Further information on the entire suite of Chesapeake Bay Program models, their documentation, and application can be found at: http://www.chesapeakebay.net/model.htm.

CASE STUDY
SAN FRANCISCO BAY PROGRAM:
MANAGING COASTAL RESOURCES OF THE U.S.

The following case study is extracted from U.S. Geological Survey Fact Sheet FS-053-95 (available online at: http://water.usgs.gov/wid/html/sfb.html)

Coastal ecosystems, such as bays and estuaries, are among our most disturbed natural environments. These ecosystems also are among our most valuable habitats—estuaries supported U.S. fisheries valued at $19 billion in 1990. Although many human activities cause change in the coastal zone, they occur against a background of natural change. Effective coastal-zone management requires that we identify and understand these separate causes of ecosystem change. With this goal in mind, the United States Geological Survey (USGS) began in 1968 a broad program of scientific study in San Francisco Bay (Figure 1). The program is based on a conviction that sustained, multifaceted investigation of one estuary will produce general lessons to guide the management of natural resources associated with all our coasts.

The USGS San Francisco Bay Program has produced more than 250 reports, including three books and a review of the human modifications of the bay. These publications are a source of guidance to resource managers as they work to understand how human activities (such as water diversion, commercial trade, and waste inputs) cause change in the coastal zone. The program has been organized around themes. One of the most important themes is the integrated study of nutrients, toxic substances, and living resources at lower levels of the food chain—the phytoplankton and bottom dwelling invertebrates. Close collaboration between chemists and ecologists has helped to explain how plant and animal species of coastal ecosystems are organized into food chains, how nutrients and toxic contaminants are incorporated into these food chains, and how the lessons learned from detailed scientific understanding can be applied to develop effective monitoring programs and rational environmental standards.

Nutrient Enrichment

Human settlement around coastal water bodies has led to increased inputs of nutrients such as nitrogen and phosphorus. Many estuaries are now among the most intensively fertilized environments on Earth. Each day, San Francisco Bay receives more than 800 million gallons of municipal wastewater containing 60 tons of nitrogen. In response to these concerns, the USGS developed a biological monitoring procedure that has been used continuously since 1977 near a waste-treatment facility. Monitoring continued as wastewater-treatment technologies improved. This is the longest continuous record of contaminant concentrations in a natural environment of the United States. The transfer of monitoring procedures developed by the USGS to local agencies and businesses serves as a model of cooperation between research and regulatory agencies.

Management Questions

Water-quality managers need to know how nutrient inputs cause changes in water quality, the natural capacity of coastal waters to assimilate added nutrients, the level of waste treatment required to protect living resources from the harmful effects of nutrient enrichment, and if programs of nutrient reduction are having beneficial effects.

USGS Contributions

Since 1968, the continuous study of San Francisco Bay by the USGS has given that agency a unique opportunity to follow ecosystem responses to improved wastewater-treatment methods as mandated by State and Federal legislation. One result of the implementation of these improved methods has been a

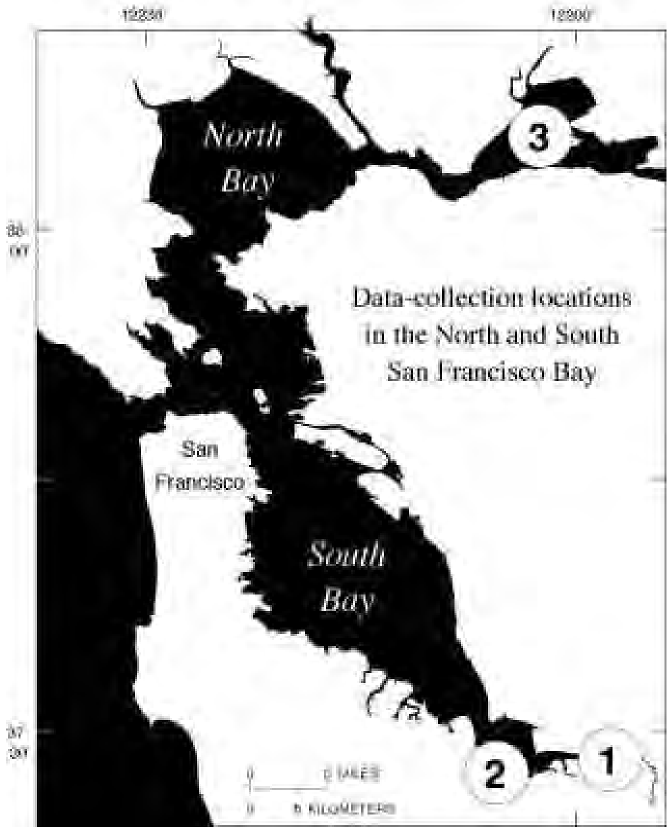

Figure 1. San Francisco Bay has been a focus of intensive investigation by the USGS since 1968.

large reduction in the input of ammonia-nitrogen from some municipal wastewater-treatment facilities (Figure 2).

USGS studies show that in spite of its nutrient enrichment, San Francisco Bay has not been affected by harmful algal blooms. This seeming paradox is explained partly by the abundant bottom-dwelling invertebrates (small clams, mussels, crustaceans) that filter the water and remove new algae as fast as they are produced. Feeding by these animals is a form of natural waste treatment that helps control the growth of algae in a nutrient-rich environment.

Concepts and measurement techniques from this USGS program are now incorporated into a locally funded and managed Regional Monitoring Program.

Lessons Learned

- The chemical quality of coastal waters can respond almost immediately to waste-treatment improvements.
- Responses of biological communities to these chemical changes can take years or even decades.
- Coastal water bodies have differing sensitivities to waste loading. The most cost-effective national strategy for regulating nutrient inputs will consider these differences among ecosystems.

For Further Information:

Visit the USGS website on San Francisco Bay at: http://sfbay.wr.usgs.gov/

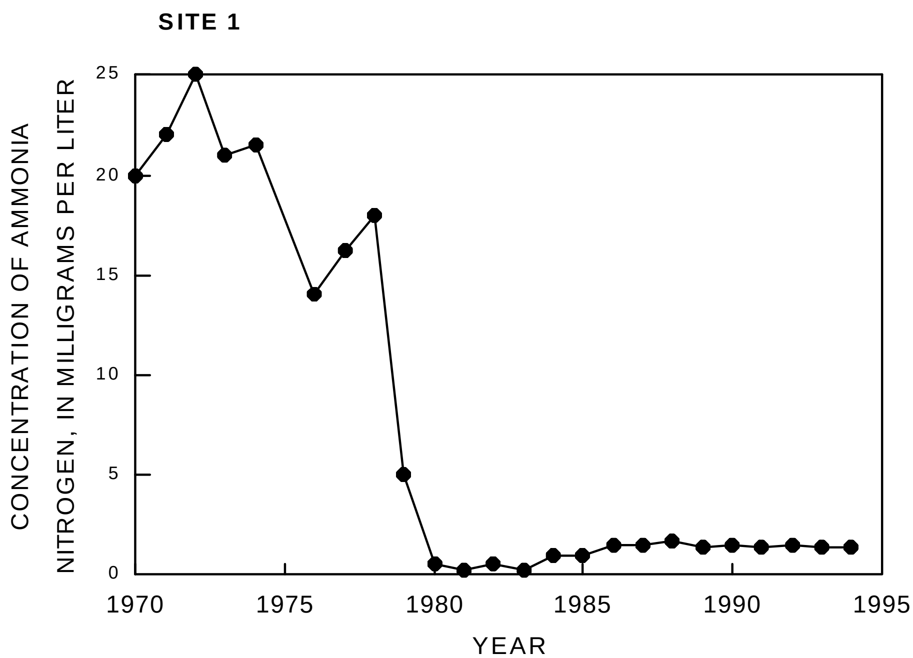

Figure 2. Implementation of advanced wastewater treatment in 1979 immediately reduced the input of ammonia- nitrogen to South San Francisco Bay. In prior decades, the South Bay had repeated episodes of oxygen depletion and animal die-offs. USGS measurements have shown a complete cessation of these episodes since 1980. Spawning salmon have recently been observed in South Bay streams for the first time since the early 1900's. See figure 1 for location of site.

CASE STUDY
LONG ISLAND SOUND - HYPOXIA
U.S. Environmental Protection Agency
http://www.epa.gov/region01/eco/lis/hypox.html

The Problem
During the summers of 1987-93, from half to two-thirds of the Sound's bottom waters experienced dissolved oxygen levels below 5 milligrams of oxygen per liter of water (mg/L). Levels of dissolved oxygen of 5 mg/L and higher are generally accepted as being protective of the Sound's estuarine life. In 1989, a particularly bad summer, more than 500 square miles (40 percent) of the Sound's bottom waters had dissolved oxygen levels less than 3 mg/L. During most of these years, dissolved oxygen in a portion of the Sound (up to 50 square miles) fell below 1 mg/L and in 1987 anoxia, the absence of any oxygen, was recorded in a portion of the Western Narrows.

These low levels of dissolved oxygen cause significant, adverse ecological effects in the bottom water habitats of the Sound. To date, research shows that the most severe effects (such as mortality) occur when dissolved oxygen levels fall below 1.5 mg/L at any time and below 3.5 mg/L in the short-term (i.e., 4 days), but that there are probably mild effects of hypoxia when dissolved oxygen levels fall below 5 mg/L. The levels regularly observed in the Sound during late summer:

- Reduce the abundance and diversity of adult finfish;

- Reduce the growth rate of newly-settled lobsters and perhaps juvenile winter flounder;

- Can kill species that cannot move or move slowly, such as lobsters caught in pots and starfish, and early life stages of species such as bay anchovy, menhaden, cunner, tautog, and sea robin;

- May reduce the resistance to disease of lobsters and other species; and

- Diminish the habitat value of Long Island Sound.

The Cause of the Problem
Excessive discharges of nitrogen, a nutrient, are the primary cause of hypoxia. Nitrogen fuels the growth of planktonic algae. The algae die, settle to the bottom of the Sound and decay, using up oxygen in the process.

Natural stratification of the Sound's waters occurs during the summer when warmer, fresher water "floats" on the top of cooler, saltier water that is more dense. This natural stratification forms a density difference between the two layers called a pycnocline. This prevents mixing of surface and bottom waters.

Oxygen from the atmosphere and photosynthesis keep the surface layer well oxygenated, but the oxygen cannot pass through the pycnocline into the bottom layer very easily. Decaying algae and other organic material in the sediment and animal respiration in the bottom layer use up oxygen faster than it is replenished. Hypoxia develops and usually persists as long as the stratification lasts (usually one to two months in late summer).

But hypoxia in Long Island Sound is too complex to fully understand using direct observations alone. Natural variations in weather and other physical factors affect the extent and severity of hypoxia. The Management Conference has constructed mathematical models in order to understand the relationship among natural variations, human-caused pollutant loadings to the Sound, and dissolved oxygen levels in the Sound. Two models, a water quality model that approximates the biological and chemical processes of the Sound and a hydrodynamic model that describes physical processes, have been developed.

An intensive field program in Long Island Sound to collect data for the computer models was undertaken from April 1988 to September 1989. These data were used to calibrate and verify the models to ensure that they reproduce the important features of the Sound. The water quality model, called LIS 2.0, provided needed insight into the causes of hypoxia and was the basis for actions to begin to reduce nitrogen discharges to the Sound. However, because it simulates the movement of the Sound's waters in only two dimensions (east-west and surface to bottom) and in a simplified manner, the LIS 2.0 model did not provide the best technical foundation for identifying the total level of reduction in nitrogen loads that should be attained or the most cost-effective means to achieve targeted reductions. The hydrodynamic model, developed by the National Oceanic and Atmospheric Administration and completed in July, 1993, uses tide and current measurements to simulate the water's circulation in three dimensions (east-west, north-south, surface to bottom). It was coupled to the water quality model, to create LIS 3.0. The LIS 3.0 model provides an advanced tool to relate sources of nitrogen from specific geographic areas to the hypoxia problem in the western Sound. Because the impact of the nitrogen load from different management zones can be deter-mined using LIS 3.0, the LISS can assign priorities for management to ensure that the most the cost-effective options are pursued.

The modeling, combined with field monitoring and laboratory studies, provided a level of detail to support some clear conclusions about hypoxia in the Sound, its causes, and its solutions. In addition, the models allowed the LISS to simulate water quality conditions as they were in the past, as they are today, and as they could be in the future under alternative nitrogen control scenarios.

- The most oxygen that can be dissolved in Long Island Sound at summer water temperatures is about 7.5 milligrams per liter (mg/L) of water. This is known as the saturation level.

- Oxygen concentrations greater than 5.0 mg/L provide healthy conditions for aquatic life. Concentrations between 5.0 mg/L and 3.5 mg/L are generally healthy, except for the most sensitive species. When concentrations fall below 3.5 mg/L, conditions become unhealthy. The most severe effects occur if concentrations fall below 2.0 mg/L, even for short periods of time.

- The growth of algal blooms in Long Island Sound is dependent upon the availability of nutrients. These blooms end when the pool of nitrogen available for continued growth is depleted.

- In pre-colonial days, natural, healthy biological activity brought oxygen levels below saturation due to the natural loadings of organic material and nitrogen, but oxygen levels probably fell below 5 mg/L only in limited areas and for short periods of time.

- Under today's higher nutrient and organic material loading conditions, minimum oxygen levels average approximately 1.5 mg/L. These levels are associated with severe hypoxia.

- By substantially reducing nitrogen loadings to the Sound, the minimum oxygen levels in the bottom waters during late summer can be increased to an average of about 3.5 mg/L, thereby significantly reducing the probability and frequency of severe hypoxia and reducing the area affected by hypoxia.

- Increases in nitrogen delivered to the Sound could significantly worsen the hypoxia problem, causing larger areas to have lower oxygen levels for longer periods of time. The probability of events like the summer of 1987, when anoxia (no oxygen) became a reality in the Sound, offshore of Hempstead Harbor, would also increase.

Understanding the components of the load of nitrogen entering the Sound is fundamental to understanding the plan:

- In 1990, defined as a baseline year by the Management Conference, the total nitrogen load was 90,800 tons per year.

- By 1992, the total nitrogen load had increased to 93,600 tons per year; this increase was anticipated and was a consequence of terminating ocean disposal of sewage sludge from New York City and the need to treat some of the sludge at facilities within the basin, reintroducing nitrogen to the wastestream.

- Of the 93,600 tons per year, approximately 39,900 tons are from natural sources and not subject to reductions by management activity.

- The remaining 53,700 tons of nitrogen per year are associated with human activities and have the potential to be reduced through management actions.

- 10,700 tons of nitrogen per year enter the Sound through its boundaries -- the East River in the west and The Race in the east; efforts to reduce this substantial western load will come under the auspices of the New York-New Jersey Harbor Estuary Program.

- 2,200 tons of nitrogen per year enter the Sound from direct atmospheric deposition; the Management Conference estimates that this load will be reduced to 1,540 tons of nitrogen per year through implementation of the 1990 Clean Air Act amendments.

- The remaining 40,800 tons of nitrogen per year are a result of human activity coming from point and nonpoint source discharges in the Sound's drainage basin and are the subject of the plan. Point source discharges, primarily sewage treatment plants, result in 32,400 tons of nitrogen each year and nonpoint source discharges, such as agricultural and stormwater runoff, result in 8,400 tons of nitrogen each year.

The Plan to Solve the Problem

The goal of the hypoxia management plan is to eliminate adverse impacts of hypoxia resulting from human activities. Achievement of this goal will require very large investments of capital, a long-term commitment, and the assistance of the New York-New Jersey Harbor Estuary Program. Therefore, the Management Conference has established interim targets for dissolved oxygen and has outlined a phased approach to achieving them, using what is known now to support early phases and committing to take additional steps as increased understanding of the environment will dictate in the future.

Interim Dissolved Oxygen Targets

Using scientific information on the relationship between oxygen levels and ecological effects, the Management Conference has established interim target levels for oxygen that, if achieved, would minimize the adverse impacts of hypoxia. In summary, the interim dissolved oxygen targets for the bottom waters of the Sound are to:

- Maintain existing dissolved oxygen levels in waters that currently meet State standards;

- Increase dissolved oxygen levels to meet standards in those areas below the State standards but above 3.5 mg/L; and,

- Increase short-term average dissolved oxygen levels to 3.5 mg/L in those areas currently below 3.5 mg/L, ensuring that dissolved oxygen never goes below 1.5 mg/L at any time.

- There are also interim targets for the surface waters of the Sound.

Phased Approach

The Management Conference is implementing a phased approach to reducing nitrogen loadings to the Sound from point and nonpoint source discharges within the Sound's drainage basin.

- Phase I, as announced in December of 1990, froze nitrogen loadings to the Sound in critical areas at 1990 levels to prevent hypoxia from worsening.

- Phase II, as detailed in the plan, includes significant, low-cost nitrogen reductions that begin the process of reducing the severity and extent of hypoxia in the Sound.

- Phase III will present nitrogen reduction targets to meet the interim targets for dissolved oxygen, which will prevent most known lethal and sublethal effects of hypoxia on the Sound's estuarine life. Phase III also will lay out the approach for meeting the nitrogen load reduction targets.

Phase I - The Nitrogen Loading Freeze

Phase I was announced in December 1990. It called for a freeze on point and nonpoint nitrogen loadings to the Sound in critical areas at 1990 levels. It committed the States and local governments to specific actions to stop a 300-year trend of ever increasing amounts of nitrogen entering the Sound.

Since 1990, activities have been underway in New York and Connecticut to manage nitrogen from sources within the New York and Connecticut portions of the drainage basin, starting with adoption of the Phase I “freeze” on loadings.

- Connecticut reacted quickly to obtain $15 million in State funds to ensure that the nitrogen freeze was implemented. Consent orders are in place to cap the nitrogen loads at the 15 affected facilities.

- In New York City, the New York State Department of Environmental Conservation (NYSDEC) and the city have reached full agreement on sewage treatment permit limits, freezing total nitrogen loadings at 1990 levels.

- In Westchester County, the NYSDEC has issued final permits to the four existing sewage treatment plants, freezing their aggregate load at the 1990 level. This was done with the full agreement of the county.

- On Long Island, the NYSDEC proposed individual permits that freeze the loads from individual discharges at 1990 levels; in response, the dischargers proposed establishment of an aggregate limit. State and local authorities agreed on aggregate load limits for targeted facilities.

Phase I agreements to control nonpoint sources centered around three categories:

- Use of existing nonpoint source and stormwater management programs to focus on nitrogen control with the objective of freezing the loads.

- Assessing tributary loads to Long Island Sound to begin planning for their control.

- Assigning priorities for management to coastal subbasins where nitrogen loads were estimated to be the highest.

Phase II - Low Cost Nitrogen Reductions

For Phase II, the LISS made a commitment in 1994 to reduce nitrogen discharges to the Sound from peak loadings by approximately 7,550 tons per year. This phase consists of incorporating a variety of low-cost nitrogen removal technologies at selected sewage treatment plants. The States have moved aggressively to implement nitrogen control activities, using innovative strategies and seeking the cooperation of local governments.

In Connecticut, the goal was to achieve a reduction of 850 tons per year in nitrogen loads. The State of Connecticut has awarded more than $15 million through its State Clean Water Fund to 11 southwestern sewage treatment plants to test and demonstrate the efficiency of upgrades for nitrogen treatment. In addition, the first plant in the State designed to denitrify has been constructed in Seymour. As of December 1997, the load of nitrogen from plants in the Phase II agreement has been reduced by almost 900 tons per year, exceeding the Phase II goal.

The State of New York revised the permits issued to sewage treatment plants, with the consent of local authorities, to establish nitrogen limits at 1990 levels. The permits include an aggregate load for facilities within Management Zones 7-11 (New York City, Westchester County, and Long Island). The New York goal was to reduce nitrogen loadings by 6,700 tons per year from peak loadings from actions to be completed by 2006. The goal of these actions was to compensate for the increased load due to sludge treatment and reduce loadings back below 1990 levels. As of 1997, one sewage treatment plant in Westchester County and four in New York City have implemented nitrogen removal technologies. New York City is required to implement additional nitrogen removal technologies at the upper East River sewage treatment plants. As of December 1997, the load of nitrogen from sewage treatment plants in New York had decreased by 3,000 tons per year from peak loadings. In addition, New York City has entered into a consent order to provide nitrogen removal at the reconstructed Newtown Creek facility, scheduled for completion in 2007.

In addition, both States have:

- Developed materials and conducted training for treatment plant personnel on nitrogen removal technologies and procedures.

- Required sewage treatment plants to identify in their plans how they will remove nitrogen, if required to do so.

- Required nutrient monitoring at sewage treatment plants to improve understanding of nitrogen sources and treatment plant capability.

- Increased the share of nonpoint source pollution control funds targeted to projects that reduce nitrogen loads to the Sound.

- Formulated Coastal Nonpoint Pollution Control Programs to address coastal nonpoint sources of nitrogen.
- Undertaken demonstration projects that address a variety of nonpoint source control issues and technologies (e.g., urban runoff treatment by artificial pond/wetland systems, parking lot runoff treatment, septic system technologies to treat and remove nitrogen, controlling runoff from agricultural land and from marinas).

As of December 31, 1997, nitrogen loadings to the Sound from point and non-point sources within the New York and Connecticut portions of the watershed have been reduced as a result of these activities by 3,900 tons per year from peak loadings.

Phase III - Nitrogen Reduction Targets to Eliminate Severe Hypoxia

While steps taken in Phases I and II will help to reduce the extent of hypoxia, additional nitrogen reduction is needed to restore the health of Long Island Sound. Phase III sets the course by setting specific nitrogen reduction targets for each of the 11 management zones around the Sound. An array of environmental and economic considerations were taken into account throughout the process.

Oxygen Benchmarks

The water quality standard for oxygen in Long Island Sound is 6 mg/L in Connecticut and 5 mg/L in New York. Modeling indicates that even if maximum nitrogen reduction technologies were implemented, the water quality standards for oxygen would not be achieved throughout the summer in all areas of the Sound. To help establish priorities for action, the LISS has identified oxygen conditions that will minimize adverse impacts on living resources of the Sound.

Two major research efforts, a laboratory study by the EPA’s Office of Research and Development and a field study by the Connecticut Department of Environmental Protection (CTDEP) have provided much of the information on how low oxygen conditions affect living resources in the Sound. Both studies corroborated that severe effects occurred whenever levels of oxygen fell below 2.0 mg/L. The field surveys noted large reductions in the number and types of aquatic life present. The lab experiments recorded reductions in growth and increases in mortality. In both studies, effects became significant when oxygen levels fell below 3.5 mg/L, though some effects occurred at levels between 3.5-5.0 mg/L.

As a result, the LISS has determined that unhealthy conditions occur whenever oxygen levels fall below 2.0 mg/L at any-time or remain below 3.5 mg/L over a 24-hour period. Most adverse impacts can be prevented if oxygen levels exceed these conditions, and they have been used as benchmarks to assess the relative benefits of alternative management strategies for improving the health of Long Island Sound.

Cost-effectiveness

LISS managers looked at a range of nitrogen reduction options for the three major sources of nitrogen in the watershed, sewage treatment plants, industrial facilities, and nonpoint source runoff to determine the most cost-effective option.

- *Sewage Treatment Plants*: As nitrogen removal requirements become more stringent, the cost of controls tends to increase. To identify a cost-effective level of treatment, LISS managers arrayed the possible nitrogen reduction options for all 70 sewage treatment plants in the 11 management zones and calculated the average oxygen improvement in the Sound per dollar spent. Improvements at sewage treatment plants that had better than average cost-effectiveness at improving oxygen conditions in the Sound were identified. These actions, in total, could achieve a 62 percent reduction in loads, or 122,044 pounds/day.

- *Industrial Facilities*: A limited number of industrial facilities directly contribute nitrogen to the Sound; all are located in Connecticut and contribute an estimated 6,717 pounds per day of nitrogen to the Sound. Because information on the cost of reducing nitrogen from industrial sources was not readily available, these facilities were not included in the cost analyses used for sewage treatment plants. Instead, the cost-effective level of treatment identified for sewage treatment plants, 62 percent, was applied to the industrial sources, resulting in a 4,165 pounds per day reduction for industrial facilities. This represents an aggressive but cost-effective level of nitrogen control for these sources.

- *Nonpoint Sources*: Decisions on controls of nonpoint source runoff must be made in the broader context of watershed management, since control measures will also help reduce suspended solids, toxic contaminants, pathogens, and floatable debris. The LISS recommends that aggressive controls of nonpoint source pollution be implemented for both existing and new development, through both habitat protection and restoration activities, and structural and nonstructural best management practices. This effort could result in a 10 percent reduction in the non-point source load from sources within the New York and Connecticut portions of the watershed, or 2,604 pounds per day.

Adding the potential nitrogen reductions from cost-effective controls on sewage treatment plants, industrial sources, and nonpoint runoff sources results in a total reduction of 128,813 pounds per day (23,500 tons per year). The next step is to allocate responsibility for achieving these reductions among the 11 management zones fairly.

Allocating Responsibility

The cost curve analysis provided an option for allocating nitrogen reductions among the sewage treatment plants. Sewage treatment plant upgrades with greater than average cost-effectiveness would be implemented while upgrades with below average cost-effectiveness would not be implemented. However, the LISS decided that relying on the cost curve analysis alone would not be a fair or even feasible approach and would not provide the best solution to allocating nitrogen reduction.

There are several reasons for this conclusion. Most importantly, the cost estimates were general and not uniform in their development. More accurate cost estimates must await detailed facilities planning based upon a clear definition of the nitrogen discharge limits that will have to be met. In addition, local concerns and considerations such as the need to purchase land for expansion and to distinguish between costs for nitrogen removal versus ongoing maintenance, expansions for growth, and secondary upgrade needs (which were not included in the cost estimates) were not addressed evenly in the cost analysis.

Cost considerations aside, it is necessary for all sewage treatment plants to share the burden of nitrogen removal. All sewage treatment plants contribute nitrogen to Long Island Sound, albeit with different effect. All jurisdictions will benefit from improved water quality. Therefore, it is reasonable to expect all contributors to the problem to con-tribute to the solution.

For those reasons, LISS has assigned each management zone equal responsibility to reduce its share of the nitrogen load. To achieve a similar level of oxygen improvement from reductions allocated to each zone by the same percentage, the load reduction target was adjusted slightly to 23,800 tons per year from the original 23,500 tons per year. The total human-derived load coming from sewage treatment plants, industrial point sources, and nonpoint sources, including atmospheric depositions within the water-shed, is 40,650 tons per year. Therefore, the Soundwide nitrogen target is a 58.5 percent reduction in the human-derived load from point and nonpoint sources in the watershed.

Phase III Actions

Phase III actions will minimize adverse impacts of hypoxia caused by human activities in a cost-effective manner, while ensuring that new information is gathered to refine and improve management over the long term. Using the framework described above, the LISS set a 58.5 percent reduction target for the enriched load of nitrogen from sources within the New York and Connecticut portions of the watershed.

Strategies

Attaining the nitrogen reduction targets will require aggressive control of point sources, such as sewage treatment plants and industrial sources, and nonpoint sources, such as on-site sewage systems and runoff from roads, parking lots, and construction sites. To achieve the reduction targets, the States, working with local governments, will select the mix of point and nonpoint source controls to be implemented in each management zone. Recognizing that each watershed is different, the plan provides the States and municipalities considerable flexibility in determining how nitrogen reduction actions are carried out within each zone.

By August 2000, the States will take the following actions:

- Develop watershed plans for each management zone that will set the course for achieving the targets as scheduled.
- Consistent with those plans, incorporate limits on the amount of nitrogen that can be discharged from sewage treatment plants and industrial sources into discharge permits.
- Conduct comprehensive nonpoint source management and habitat restoration activities.

Because the total nitrogen load entering the Sound from human sources is dominated by point source discharges, the plan emphasizes technologies that can be applied to sewage treatment facilities and industrial discharges.

In order to achieve significant reductions in the nonpoint source nitrogen load, home owners, farmers, businesses, municipalities, and the States will need to reduce current inputs of nitrogen to the watershed and restore and preserve the nitrogen removal capabilities of existing natural systems. These reductions can be achieved using a number of approaches—resource-based land use decisions at the local level, watershed-wide use of appropriate structural and nonstructural best management practices (e.g., stormwater detention ponds, artificial wetlands, streetsweeping, cleaning catch basins), habitat protection and restoration, and pollution prevention management practices. All approaches will require a concerted education and outreach effort.

Timing

The planning, financing, and construction of upgrades to sewage treatment plants necessary to achieve the 58.5 percent reduction target will require sustained effort and commitment over a long period of time. Therefore, the LISS recommends phasing-in the necessary reductions over 15 years:

- 40 percent in 5 years,
- 75 percent in 10 years, and
- 100 percent in 15 years.

Cost

The *Comprehensive Conservation and Management Plan* identified that the cost of achieving maximum nitrogen removal from all point sources would range from $6 to $8 billion ($5.1 to $6.4 billion in New York State and from $900 million to $1.7 billion in Connecticut). Because of the successful demonstration of full scale nitrogen removal technologies at sewage treatment plants undertaken as part of Phase II, the estimated costs of capital improvements at sewage treatment plants have decreased. The estimated cost of achieving maximum nitrogen removal levels at the 70 treatment plants in New York and Connecticut is now about $2.5 billion

Because of the cost-effective approach described above, the LISS nitrogen reduction strategy would not require all treatment plants to meet limit-of-technology reductions. As a result, the incremental capital cost of achieving the Phase III point source controls was estimated to be $300 million for New York State and $350 million for Connecticut. These cost estimates have been questioned and will be revised as more detailed facility planning and design is performed. However, they show clearly that the potential cost of achieving our goals can be much less than originally estimated.

Nonpoint source controls will be implemented as part of broader watershed and habitat protection efforts. The cost of controlling nonpoint sources is more difficult to estimate than the cost of point source controls. Rather than one type of technology applied to a similar source, a variety of strategies can be applied to control a variety of nonpoint sources of nitrogen. As a result, the costs of achieving nonpoint nitrogen reductions will be addressed in the zone-by-zone plans developed by the States.

Financing

As recommended in the *Comprehensive Conservation and Management Pla*n, the main source of funding for these wastewater treatment facility improvements will be the State Revolving Fund programs. The EPA, through the federal Clean Water Act, provides financing to support State Revolving Fund loan programs.

Connecticut uses the capitalization grant from EPA to leverage with State bond funds to provide grants and low interest loans, at 2 percent interest over 20 years, to finance improvements at municipal facilities. Connecticut provides about $50 million per year in State bonding to supplement the $15 million per year provided under the Clean Water Act. At this capitalization rate, Connecticut should be able to meet municipal financing needs to implement Phase III nitrogen reductions. During fiscal year 1997, CTDEP awarded $250 million from their Clean Water Fund to finance projects of benefit to Long Island Sound, including major sewage treatment plant upgrades in Norwalk and Waterbury.

New York State established its State Revolving Fund in the custody of the Environmental Facilities Corporation. This public corporation benefits local governments in New York State by offering below-market interest rate loans to municipalities to finance wastewater improvements. Currently, the interest rate is set at up to one-half of the market rate to be repaid in 20 years. Lower rates of interest, including zero interest loans, are available for communities that can demonstrate an inability to pay the standard subsidized rate. Another major source of funding in New York State is the $1.75 billion Clean Water/Clean Air Bond Act approved by voters in November 1996. The Bond Act targeted $200 million for Long Island Sound that will be available for sewage treatment upgrades, habitat restoration, nonpoint source control, and pollution prevention.

The possible funding sources for non-point source controls reflect the diversity of both the sources and the control options. Grant funding through federal and State water quality management, natural resources management, and coastal zone management programs is available for nonpoint source activities. The State Revolving Fund loan program is also available to fund stormwater management and habitat restoration projects but has not been used to a great extent for these types of activities due to the magnitude of existing point source funding needs in Connecticut and New York.

Effluent Trading

To provide further flexibility and incentives for maximizing the timeliness and cost-effectiveness of nitrogen reduction actions, the LISS is investigating the feasibility of allowing effluent trading. Trading, if employed as part of the nitrogen reduction effort, may be an innovative way to use market forces to more efficiently meet water quality goals. The LISS is developing a trading proposal and will convene a public forum for federal, State, and local water quality officials, together with public and private interests, to evaluate its potential.

Enforcement

The provisions of the federal Clean Water Act provide a vehicle for ensuring that nitrogen reduction targets are legally enforceable. Section 303(d) of the Act requires the identification of a Total Maximum Daily Load for pollutants that will result in the attainment of water quality standards. Once a Total Maximum Daily Load has been established, the act calls for reductions to be allocated to sources so that the load target is met. New York and Connecticut and EPA will use their authorities to provide an enforceable foundation for achieving the nitrogen reduction targets. By August, 1998 the States will propose a Total Maximum Daily Load designed to meet State oxygen standards. The current Long Island Sound standards were developed with limited data on how low oxygen levels affect aquatic life in Long Island Sound. EPA is currently developing regional marine oxygen criteria that will provide a more scientifically valid basis for the development of oxygen standards. Based on this information, the States may, in the future, modify their oxygen standards. While LISS managers predict significant improvement in water quality as the nitrogen reduction targets are implemented, the attainment of current water quality standards at all times and in all areas is not expected. For this reason, the LISS will continue to assess what other kinds of actions will be needed to bring the Sound into full compliance with water quality standards.

These actions may include control of nitrogen and carbon sources outside of the Long Island Sound basin (e.g., tributary import from point and non-point sources north of Connecticut, atmospheric deposition, boundary import from point and nonpoint sources affecting New York Harbor and The Race). Alternatives to nitrogen reduction, such as aeration, will need to be considered as a possible means to achieve water quality standards in remaining areas.

Evaluating Progress

The LISS will track, monitor, and report on progress in meeting the nitrogen reduction targets annually. In addition, a formal review of the goals and objectives of the program will be performed every 5 years, coinciding with the progress checkpoints for nitrogen reduction. The review will consider:

- Progress and cost of implementation, including a reevaluation of the knee-of-the-curve analysis used to establish the Phase III nitrogen reduction targets,
- Improvements in technology, including the results of quality controlled pilot projects,
- The regional dissolved oxygen criteria to be published for comment,
- Water quality standards,
- Refined information on the ecosystem response to nitrogen reductions,

- The results of peer reviewed modeling, and

- Research on the impacts of hypoxia to living resources and their habitats.

Each of these factors will be considered in a balanced manner in the reevaluation process. As a result of the review, the LISS may recommend improvements that could result in changes in how the overall program will be implemented.

For More Information:

Mark Tedesco
EPA Long Island Sound Office
888 Washington Blvd.
Stamford, CT 06904-2152
Phone: 203/ 977-1541
Fax: 203/ 977-1546

CASE STUDY
NP BUDGET FOR NARRAGANSETT BAY

S. V. Smith
University of Hawaii
808-956-8693
http://data.ecology.su.se/MNODE/North%20America/NRB.HTM

Narragansett Bay, Rhode Island (41˚ 35’ N, 71˚ 20’W), is a relatively well mixed, near-oceanic salinity estuary on the northeast (Atlantic) coast of the U. S. It occupies an area of 264 km^2 (Table 1) and has a mean depth of 9.7 m. Note that both the area and volume differ from the comments in Nixon et al. (1995), but seem consistent with Kremer and Nixons (1978) explicit tabulation. Freshwater flow into the system averages about 8.2 x 10^6 m^3/d, from a watershed of 3500 km^2. Primary production in the system is dominated by phytoplankton (29 mol C m^{-2} yr^{-1}) with a C:N:P ratio of about 112:13:1. The budget described below is based on data collected primarily in the late 1970’s and through much of the 1980’s. Details of this kind of analysis can be found at the LOICZ - Biogeochemical Modelling web site at: http://data.ecology.su.se/MNODE/index.htm.

Sector area and volume data are from Kremer and Nixon (1978). Sector nutrient concentrations are annual averages (based on surface and deep water data) also from Kremer and Nixon. Sector nutrient masses are calculated as volume x concentration. The sectors at the bay mouth (#5, 8) are used for "oceanic values."

Nutrient exchange fluxes (Table 2) are calculated using an average 26-day exchange time, as calculated by Pilson (1985) with a water and salt budget (analogous to procedure in Gordon et al., 1996). The bay

Table 1. Sector areas, volumes, and nutrient concentrations. Data are used to calculate volume-averaged concentrations for the outer portion of the bay ("ocean") and the bay proper

SECT. #	VOL.10^6 m^3	AREA 10^6 m^2	DIP µM	NH4 µM	NO3 µM	Sum DIN µM	DIP 10^6 mol	NH_4 10^6 mol	NO_3 10^6 mol	S DIN 10^6 mol
1	130	20.1	1.8	12	11	23	0.23	1.56	1.43	2.99
2	300	44.6	1.5	7	6	13	0.45	2.10	1.80	3.90
3	115	28.5	1.6	3	6	9	0.18	0.35	0.70	1.05
4	463	61.9	1.4	4	4	8	0.65	1.85	1.85	3.70
5	204	20.0	1.0	1	3	4	0.20	0.27	0.60	0.87
6	222	26.0	1.2	2	5	7	0.27	0.44	1.11	1.55
7	573	38.5	1.0	2	4	6	0.57	1.15	2.29	3.44
8	554	24.2	0.7	1	2	3	0.39	0.39	1.11	1.50
SUM	2561	264								
		bay	1.3	4	5	9				
		ocean (secs. # 5,8)	0.8	1	2	3				

Table 2. Hydrographic exchange fluxes of nutrients

SOURCE OF FLUX	DIP 10^6mol/yr	NH_4 10^6mol/yr	NO_3 10^6mol/yr	DIN 10^6mol/yr
residual flow	-3	-7	-11	-18
net exchange flow	-18	-108	-108	-216
total hydrography	-21	-115	-119	-234

volume divided by the residence time gives a mixing exchange volume of 98.5 x 10^6 m^3/d, while the residual outflow equals the freshwater inflow (i.e., 8.2 x 10^6 m^3/d). It would, in principle, be possible to time-step through the data (at monthly increments, for example). However, to do that would require having the flow data to go with the nutrient data. Further, inspection of the graph by Nixon et al. (1995) of flow data and comparison of Pilson's (1985) flow—residence time regression equation suggests that the residence time over this range of flow is well approximated by a constant value for the exchange time. Various authors describe the bay as well mixed, and this is supported by the water composition data in Kremer and Nixon (1978). We therefore use a 1-box model to perform these calculations, rather than a vertically stratified model to describe hydrographic fluxes.

In Table 3, all boundary fluxes except hydrography were taken directly from by Nixon et al. (1985). Hydrographic flux was calculated as above. Δ Y's (the nonconservative fluxes) are calculated by difference (as described in Gordon et al., 1996). No data are available for DOP, DON, or for either inorganic or organic C, so the budget is based on inorganic N and P only. As discussed by Gordon et al. and consistent with comments in Nixon et al., it seems safe to assume that DOP and DON nonconservative fluxes do not contribute strongly to the overall nonconservative fluxes in this system.

Rates for the D Y's per unit area are calculated using the bay area of 264 km^2 (Table 4). Note that this area estimate is about 25% lower than the value used by Kremer and Nixon (1978). We have used the smaller area and volume on the basis that these are the data used to calculate the volume-averaged concentrations. Net (nfix-denit) is calculated on the assumption that the N:P ratio of D DIP is 13:1, then D DIN is balanced. Net (p-r) is calculated from the DIP flux, using a C:P ratio of 112:1.

Table 3. Total boundary fluxes of nutrients and inferred internal reactions—the system budget

Process	DIP 10^6mol/yr	NH_4 10^6mol/yr	NO_3 10^6mol/yr	DIN 10^6mol/yr
atmosphere	0	6	19	25
rivers	13	113	177	290
urban runoff	2	13	4	17
sewage	9	136	6	142
hydrography	-21	-115	-119	-234
D Y	-3	-153	-87	-240
(nfix-denit)				-201

Table 4. Nonconservative fluxes of materials and stoichiometrically inferred biogeochemical pathways

	DIP mmol m^{-2} yr^{-1}	NH_4 mmol m^{-2} yr^{-1}	NO_3 mmol m^{-2} yr^{-1}	DIN mmol m^{-2} yr^{-1}	C mol m^{-2} yr^{-1}
D Y	-11	-580	-329	-909	
D DIN_{exp}				-143	
(nfix-denit)				-766	
(p-r)					1.2

Nixon et al. (1995) have data with which the present budgetary estimates may be compared: They estimate DIP and DIN fluxes from the ocean to the bay by a hydrographic budget analogous to values used here for both influx and efflux, but they do not use this same hydrography to estimate nutrient fluxes to the ocean. Their inward DIP and DIN fluxes, obtained by time-stepping through the oceanic nutrient concentration data (bottom water only), are 27 and 115 x 10^6 mol/yr. The calculations here (using annual average data) are 29 and 108 x 10^6 mol/yr. The agreement is within 10%. It should be close, because both Nixon et al. and the calculations here are performing essentially the same calculation. Three points for minor disagreement would be that the values here just used a constant exchange rate (instead of time-varying); values used here were picked data off a graph; and surface and bottom values were averaged (on the graph, these are effectively identical in the outermost bay sectors).

Instead of using hydrography to estimate outward DIP and DIN flux, those authors estimate DIP and DIN fluxes from the bay to ocean by difference with other terms in their budget, to close the budget. They get 41-51 x 10^6 and 240-470 x 10^6 mol/yr. Again pulling the hydrographic terms apart, the calculations here yield 50 x 10^6 and 342 x 10^6 mol/yr (in both cases, within their range). It is worth noting that if the water exchange volume is incorrect, it would affect both influx and efflux of nutrients, hence have a relatively small effect on the difference between influx and efflux. The point here, of course, is that the difference between influx and efflux is probably more reliable than either of the individual fluxes.
Nixon et al. use a variety of considerations for two different sets of incubation data to assign baywide denitrification a range of 85-170 x 10^6 mol/yr (compared to 201 x 10^6 from the hydrographic budget; using their high values, agreement is within 20%).

Those authors estimate respiration to consume 8100 to 9200 x 10^6 mol/yr of organic C. Using their estimate for primary production (p) of 29 mol C m^{-2} yr^{-1} and the DIP-derived estimate for production - respiration (*p-r*) of 1.2, *r* is estimated to be 27.8 mol C m^{-2} yr^{-1}. Scaling by the bay area, this gives respiration to be 7340 x 10^6 mol/yr (within 20% of their lower estimate). If we were to use the are value given in Nixon et al. (328 km^2, instead of the value of 264 km^2 from Kremer and Nixon (1978), the respiration would be 9118 x 10^6 mol/yr (within their range).

Efforts to control the release of nutrients into Narragansett Bay have recently addressed nitrogen contributions from Publically Owned Treatment Works (POTWs) throughout the watershed. One nutrient reduction option currently being pursued is to maximize nitrogen removal from the final effluent by modifying operating conditions with existing equipment at the facility. Retrofitting existing facilities will also be considered where appropriate. A second venue involves drafting water quality based permit limits over the next few years to limit nitrogen in the final effluent of POTWs. Finally, a total maximum daily load (TMDL) for nitrogen is currently under development for the Providence River upstream of Narragansett Bay through the NPDES permitting process. A model is being developed that once calibrated, will set nitrogen load limits for POTWs that discharge to the river.

For Further Information:

S. V. Smith
Department of Oceanography,
University of Hawaii
1000 Pope Road Honolulu, Hawaii 96822 USA
email: svsmith@soest.hawaii.edu
phone: 808-956-8693
fax: 808-956-7112

CASE STUDY
NUTRIENT MANAGEMENT AND SEAGRASS RESTORATION IN TAMPA BAY, FLORIDA

Holly Greening, Tampa Bay Estuary Program

Abstract: Participants in the Tampa Bay Estuary Program have agreed to adopt nitrogen loading targets for Tampa Bay based on the water quality and related light requirements of the seagrass species *Thalassia testudinum*. Based on modeling results, it appears that light levels can be maintained at necessary levels by "holding the line" at existing nitrogen loadings. However, this goal may be difficult to achieve given the 20% increase in the watershed's human population and associated 7% increase in nitrogen loading that are projected to occur over the next 10-20 years.

To address the long-term management of nitrogen sources, a Nitrogen Management Consortium of local electric utilities, industries and agricultural interests, as well as local governments and regulatory agency representatives, has developed a Consortium Action Plan to address the target load reduction needed to "hold the line" at 1992-1994 levels. To date, implemented and planned projects collated in the Consortium Action Plan meet and exceed the agreed-upon nitrogen loading reduction goal.

The Tampa Bay estuary is located on the eastern shore of the Gulf of Mexico in Florida, USA. At more than 1000 km^2, it is Florida's largest open water estuary. More than 2 million people live in the 5700 km^2 watershed, with a 20% increase in population projected by 2010. Land use in the watershed is mixed, with about 40% of the watershed undeveloped, 35% agricultural, 16% residential, and the remaining commercial and mining.

Major habitats in the Tampa Bay estuary include mangroves, salt marshes and submerged aquatic vegetation. Each of these habitats has experienced significant areal reductions since the 1950s, due to physical disturbance (dredge and fill operations) and water quality degradation, particularly impacting the seagrasses due to loss of light availability. Five species of seagrass are commonly found in Tampa Bay, with *Thalassia testudinum* (turtlegrass) and *Syringodium filiforme* (manatee grass) dominating in the higher salinity areas and *Halodule wrightii* (shoalgrass) and *Ruppia maritima* (widgeon grass) most commonly found in lower salinities.

The importance of seagrass as a critical habitat and nursery area for fish and invertebrates, and as a food resource for manatees, sea turtles and other estuarine organisms has been recognized by the Tampa Bay resource management community for several decades. In 1990, Tampa Bay was accepted into the U.S. Environmental Protection Agency's (EPA) National Estuary Program. The Tampa Bay National Estuary Program (TBNEP), a partnership that includes three regulatory agencies and six local governments, has built on the resource-based approach initiated by earlier bay management efforts. Further, it has developed water quality models to quantify linkages between nitrogen loadings and bay water quality, and models that link water quality to seagrass goals.

Recent recommendations from the National Academy of Science National Research Council (NRC) include those which regional watershed programs might consider in developing nutrient management strategies. The NRC recommendations are based on the process designed by the Tampa Bay Estuary Program partners to develop and implement a seagrass protection and restoration management program for Tampa Bay. Critical elements of the Tampa Bay process are to:

1. Set specific, quantitative seagrass coverage goals for each bay segment.
2. Determine seagrass water quality requirements and appropriate nitrogen loading targets.
3. Define and implement nitrogen management strategies needed to achieve load management targets.

STEP 1. SET QUANTITATIVE RESOURCE MANAGEMENT GOALS

Establishment of clearly defined and measurable goals is crucial for a successful resource management effort. In 1992, TBNEP adopted an initial goal to increase current Tampa Bay seagrass cover to 95% of that present in 1950.

Based on digitized aerial photographic images, it was estimated that approximately 16,500 ha of seagrass existed in Tampa Bay in 1950. At that time, seagrasses grew to depths of 1.5 m to 2 m in most areas of the bay. By 1992, approximately 10,400 ha of seagrass remained in Tampa Bay, a loss of more than 35% since the 1950 benchmark period. Some (about 160 ha) of the observed loss occurred as the result of direct habitat destruction associated with the construction of navigation channels and other dredging and filling projects within existing seagrass meadows, and is assumed to be nonrestorable through water quality management actions.

In 1996, the TBNEP adopted a bay-wide minimum seagrass goal of 15,400 ha. This goal represented 95% of the estimated 1950 seagrass cover (minus the nonrestorable areas), and includes the protection of the existing 10,400 ha plus the restoration of an additional 5,000 ha.

STEP 2. DETERMINE SEAGRASS WATER QUALITY REQUIREMENTS AND APPROPRIATE NITROGEN LOADING RATES

Once seagrass restoration and protection goals were established by the participants, the next steps established the environmental requirements necessary to meet agreed-upon goals and subsequent management actions necessary to meet those requirements.

<u>A. Determine environmental requirements needed to meet the seagrass restoration goal</u>
Recent research indicates that the deep edges of *Thalassia testudinum* meadows, the primary seagrass species for which nitrogen loading targets are being set, correspond to the depth at which 20.5% of subsurface irradiance (the light that penetrates the water surface) reaches the bay bottom on an annual average basis. The long-term seagrass coverage goal can thus be restated as a water clarity and light penetration target. Therefore, in order to restore seagrass to near 1950 levels in a given bay segment, water clarity in that segment should be restored to the point that allows 20.5% of subsurface irradiance to reach the same depths that were reached in 1950.

<u>B. Determine water clarity necessary to allow adequate light to penetrate to the 1950 seagrass deep edges</u>
Water clarity and light penetration in Tampa Bay are affected by a number of factors, such as phytoplankton biomass, non-phytoplankton turbidity, and water color. Water color may be an important cause of light attenuation in some bay segments; however, including color in the regression model did not produce a significant improvement in the predictive ability of the regression model. Results of the modeling effort indicate that, on a baywide basis, variation in chlorophyll *a* concentration is the major factor affecting variation in average annual water clarity.

<u>C. Determine chlorophyll a concentration targets necessary to maintain water clarity needed to meet the seagrass light requirement</u>
An empirical regression model was used to estimate chlorophyll <u>a</u> concentrations necessary to maintain water clarity needed for seagrass growth for each major bay segment. The adopted segment-specific

annual average chlorophyll a targets (ranging from 4.6 μg/l to 13.2 μg/l) are easily measured and tracked through time, and are used as intermediate measures for assessing success in maintaining water quality requirements necessary to meet the long-term seagrass goal.

D. Determine nutrient loadings necessary to achieve and maintain the chlorophyll a targets
Water quality conditions in 1992-1994 appear to allow an annual average of more than 20.5% of subsurface irradiance to reach target depths (i.e., the depths to which seagrasses grew in 1950) in three of the four largest bay segments. Thus, a management strategy based on "holding the line" at 1992-1994 nitrogen loading rates should be adequate to achieve the seagrass restoration goals in these segments. This "hold the line" approach, combined with careful monitoring of water quality and seagrass extent, was adopted by the TBNEP partnership in 1996 as its initial nitrogen load management strategy.

As an additional complicating factor, a successful adherence to the "hold the line" nitrogen loading strategy may be hindered by the projected population growth in the watershed. A 20% increase in population, and a 7% increase in annual nitrogen load, are anticipated by the year 2010. Therefore, if the projected loading increase (a total of 17 U.S. tons per year) is not prevented or precluded by watershed management actions, the "hold the line" load management strategy will not be achieved.

STEP 3. DEFINE AND IMPLEMENT NITROGEN MANAGEMENT STRATEGIES NEEDED TO ACHIEVE LOAD MANAGEMENT GOALS

Local government and agency partners in the TBNEP signed an Intergovernmental Agreement (IA) in 1998 pledging to carry out specific actions needed to "hold the line" on nitrogen loadings. The IA includes the responsibility of each partner for meeting the nitrogen management goals, and a timetable for achieving them. How those goals are reached will be left up to the individual communities as defined by them in their Action Plans. The Tampa Bay National Estuary Program was also renamed the Tampa Bay Estuary Program as part of the progression from the planning phase to implementation of the adopted Comprehensive Conservation and Management Plan.

To maintain nitrogen loadings at 1992-1994 levels, local government Action Plans address that portion of the nitrogen target which relates to non-agricultural stormwater runoff and municipal point sources within their jurisdictions, a total of 6 U.S. tons of nitrogen per year through the year 2010 (Table 1).

To address the remaining 11 U.S. tons of nitrogen of the 17 total per year each year through the year 2010 needed to "hold the line" (attributed to atmospheric deposition, industrial and agricultural sources and springs), a Nitrogen Management Consortium of local electric utilities, industries and agricultural interests, as well as the local governments and regulatory agency representatives in the TBEP, was established (Table 2). The Nitrogen Management Action Plan developed by public and private partners in the Consortium combines for each bay segment all local government, agency and industry projects that will contribute to meeting the five year nitrogen management goal. To ensure that each partner was using similar nitrogen load reduction assumptions for similar projects, guidelines for calculating nitrogen load reduction credits were developed with the partners, and were used by each of the partners in the development of their action plans.

The types of nutrient reduction projects included in the Consortium's Nitrogen Management Action Plan range from traditional nutrient reduction projects such as stormwater upgrades, industrial retrofits and agricultural best management practices to actions not primarily associated with nutrient reduction, such as land acquisition and habitat restoration projects. A total of 105 projects submitted by local governments, agencies and industries are included in the Plan; 95% of these projects address nonpoint

Table 1. Tampa Bay Nitrogen Management Goals

SOURCE CATEGORY	CUMULATIVE 1995-1999 GOALS FOR NITROGEN REDUCTION/MANAGEMENT							TOTAL (reduction in annual load) (tons)
	Pinellas County	City of Clearwater	City of St. Petersburg	Hillsborough County	City of Tampa	Manatee County	TB Consortium*	
Old Tampa Bay	0.30	0.20	0.05	0.40	0.10	<0.01	1.05	2.10
Hillsborough Bay	<0.01	<0.01	<0.01	4.75	8.45	<0.01	28.25	41.50
Middle Tampa Bay	<0.01	<0.01	0.90	2.50	<0.01	0.50	7.15	11.05
Lower Tampa Bay	<0.01	<0.01	<0.01	<0.01	<0.01	8.35	17.00	25.35
Boca Ciega Bay	0.85	<0.01	1.05	<0.01	<0.01	<0.01	2.00	3.90
TOTAL	1.15	0.20	2.00	7.65	8.55	8.85	55.45	83.85
%	1.4	0.2	2.4	9.1	10.2	10.6	66.1	100.0

* Tampa Bay Nitrogen Management Consortium

Table 2. Public and Private Partners of the Tampa Bay Nitrogen Management Consortium, July 2001

Public Partners:	Private Partners:
City of Tampa	Florida Phosphate Council
City of Clearwater	Florida Power & Light
City of St. Petersburg	Tampa Electric Company
Manatee County	Florida Strawberry Growers Association
Hillsborough County	IMC-Phosphate
Pinellas County	Cargill Fertilizer, Inc.
Manatee County Agricultural Extension Service	CF Industries, Inc.
Environmental Protection Commission of Hillsborough County	Pakhoad Dry Bulk Terminals
Tampa Bay Regional Planning Council	Eastern Associated Terminals Company
Florida Department of Environmental Protection	CSX Transportation
Florida Fish and Wildlife Commission/Florida Marine Research Institute	
Southwest Florida Water Management District	
U.S. Army Corps of Engineers	
U.S. Environmental Protection Agency	
Tampa Port Authority	
Tampa Bay Estuary Program	
Florida Department of Agriculture and Consumer Services	

sources and account for 71% of the expected total nitrogen reduction. Half (50%) of the total load reduction will be achieved through public sector projects, and 50% by industry.

Table 3 summarizes expected reductions from those projects which were completed by the end of 1999. A total of 134 tons per year reduction in nitrogen loading to Tampa Bay is expected from the completed projects, which exceeds the 1995-1999 reduction goal of 84 tons per year by 60%. An updated estimate of nitrogen loadings to the bay from all sources was initiated by TBEP in summer 2001, after which the effectiveness of the proposed projects in maintaining loads to the bay will be evaluated.

Examples of specific projects and expected nitrogen loading reductions include the following:

> **Stormwater facilities and upgrades:** Stormwater improvements or new facilities include both public and private examples. Stormwater retrofits using alum injection to urban lakes reduced total nitrogen (TN) loading by an estimated 6.4 tons per year. Stormwater improvements eliminated an estimated 2 tons of TN loading per year. Industrial stormwater improvements at phosphate

Table 3. Tampa Bay Nitrogen Management Consortium Summary of Goals and Expected Reductions (cumulative tons TN reduced or precluded/year by the year 2000

Bay Segment	1995-1999 Nitrogen Reduction Goal	Expected Reduction: Completed or Ongoing Projects[1]	Expected Reduction: Atmospheric Deposition[2]
Old Tampa Bay	2.10	5.1	3.6 - 6.2
Hillsborough Bay	41.5	65.9	13.9 - 24.0
Middle Tampa Bay	11.1	21.1	4.6 - 7.9
Lower Tampa Bay	25.4	36.2	5.7 - 10.0
Boca Ciega Bay	3.9	5.6	1.2 - 2.1
Total	84.0	133.9	29.0- 50.2

[1] Projects have been completed or are under construction. These summaries do not include reductions expected from atmospheric deposition reductions.

[2] Range of atmospheric deposition reductions expected, based on two methods.

fertilizer factories and transport terminals reduced almost 20 tons TN loading per year by the year 2000.

Land acquisition and protection: Land acquisition and maintenance of natural or low intensity land uses precludes higher density uses and higher rates of TN loading. Land acquisition precluded more than 15 tons TN loading per year by the end of 1999.

Approved overlay districts requiring additional nutrient control in management areas precluded an estimated 10 tons per year TN loading.

Wastewater reuse: Wastewater reuse programs resulted in a 6.4 ton per year reduction on TN loading. Conversion of septic systems to sewer reduced TN loading by 1.7 tons per year.

Emissions reduction: Estimated emissions reduction from coal-fired electric generating plants between 1995-1997 resulted in reductions of NO_x emissions of 11,700 - 20,000 tons. To estimate the reduction of nitrogen deposition which reaches the bay (either by direct deposition to the bay's surface, or by deposition and transport through the watershed), a 400:1 ratio (NO_x emissions units to nitrogen units entering the bay) is assumed. Expected reductions from atmospheric deposition thus ranged from 29 to50 tons per year by 1999. To date, emissions reductions have not been included in the estimated total TN reduction to the bay, pending agreement on estimation methods.

Habitat restoration: Although typically conducted for reasons other than nutrient reduction, habitat restoration to natural land uses reduces the amount of TN loading per acre in runoff. Habitat restoration projects have been completed or are underway in all segments of Tampa Bay's watershed. Estimated TN load reduction from completed habitat restoration projects totaled an estimated 7 tons per year.

Agricultural BMPs: Water use restrictions have promoted the use of microjet or drip irrigation on row crops (including winter vegetables and strawberries) and in citrus groves. Micro-irrigation has resulted in potential water savings of approximately 40% or more over conventional systems and an estimated 25% decrease in fertilizer applied. Nitrogen reduction estimates from these actions total 6.4 TN tons per year.

Education/public involvement: For those projects for which nitrogen load reductions have not been calculated or measured, but some reductions are expected, the Consortium Action Plan assumes a 10% reduction estimate until more definitive information is available. These programs have reduced TN loading by an estimated 2 tons per year.

Industrial upgrades: A phosphate fertilizer mining and manufacturing plant has terminated the use of ammonia in flot-plants (an element of the fertilizer manufacturing process), resulting in a reduction of 21 tons per year of nitrogen loading. Other fertilizer manufacturing companies have upgraded their product conveyor systems, resulting in a TN reduction of more than an estimated 10 tons per year due to control of fertilizer product loss. The termination of discharge by an orange juice manufacturing plant into a tributary of Tampa Bay has resulted in a reduction of more than 11 tons per year TN loading.

The approach advocated by the TBEP stresses cooperative solutions and flexible strategies to meet nitrogen management goals. This approach does not prescribe the specific types of projects that must be included in the Action Plan; Consortium partners have been encouraged to pursue the most cost-effective options to achieve the agreed-upon goals for nitrogen management. The TBEP will review and revise nitrogen management goals every five years, or more often if significant new information becomes available.

SUMMARY

The Tampa Bay management community has agreed that protection and restoration of Tampa Bay living resources is of primary importance. Through the TBEP process (initiated in 1991), partners have adopted nitrogen loading targets for Tampa Bay based on the water quality requirements of *Thalassia testudinum* and other native seagrass species. A long-term goal has been adopted to achieve 15,400 ha of seagrass in Tampa Bay, or 95% of that observed in 1950. To reach the long-term seagrass restoration goal, a 7% increase in nitrogen loading associated with a projected 20% increase in the watershed's human population over the next 20 years must be offset. Government and agency partners in the Tampa Bay Estuary Program and private industries and interests participating in the Nitrogen Management Consortium have identified and implemented specific nitrogen load reduction projects to ensure that water quality conditions necessary to meet long-term living resource restoration goals for Tampa Bay are achieved.

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CASE STUDY
RESTORING CHESAPEAKE BAY WATER QUALITY

Contact: Richard Batiuk, 410/267-5731; batiuk.richard@epa.gov

Original Nutrient Reduction Goal

In 1987, the Chesapeake Bay Program partners set a 40 percent reduction goal for nitrogen and phosphorus to improve low oxygen conditions in the deep trench of the mainstem bay. The goal was later defined to apply only to "controllable" sources, and only from the States—Maryland, Virginia, Pennsylvania—and the District of Columbia are also listed as impaired tidal waters.

All listed impaired waters are scheduled to have a Total Maximum Daily Load or TMDL developed. A TMDL defines the pollutant load that a waterbody can assimilate without causing violations of water quality standards and allocates the loading to contributing point sources and nonpoint source categories. Once a TMDL is established by a State and approved by EPA through regulatory action, it is implemented through regulatory and nonregulatory programs. A regulatory TMDL covering the entire 64,000 square mile bay watershed will be put in place by 2011 if bay water quality is not restored.

Keeping A Cooperative Approach to Bay Restoration

To avoid potential negative impacts that a regulatory TMDL process might have on the successful, cooperative efforts being used by the States' tributary strategy programs, the Chesapeake 2000 Agreement lays out a series of commitments directed toward seeking a cooperative solution to restoring bay water quality by 2010.

The bay watershed partners will define the water quality conditions necessary to support bay living resources—fish, crabs, oyster, and bay grasses by 2001. These required conditions will be defined through a series of Chesapeake Bay water quality criteria for dissolved oxygen, water clarity, and chlorophyll *a* currently under development.

Important distinct bay and tributary tidal water habitats are being identified and characterized as designated uses, where the above bay criteria will be applied to fully protect the aquatic living resources.

The States with bay tidal waters—Maryland, Virginia, Delaware, and the District of Columbia—have all committed to adopting these bay criteria and tidal water designated uses into their individual State water quality standards by 2003.

Critical to supporting the States' adoption of the bay criteria and refined tidal waters designated uses will be a baywide Use Attainability Analysis (UAA).

Loading caps on nutrients and sediments needed to meet the bay water quality criteria will be allocated to major tributary basins and individual States within those basins by December 2001.

Tributary strategies, detailed implementation plans to reach the allocated loading caps will be developed in cooperation with local watershed stakeholders.

A reevaluation planned for 2005 will provide an opportunity for any necessary mid-course corrections on the road to restoring bay water quality by 2010.

Bay Criteria: Defining Restored Bay Water Quality

The Chesapeake 2000 Agreement committed the signatories to the following: "by 2001, define the water quality conditions necessary to protect aquatic living resources." These water quality conditions are being defined through the development of Chesapeake Bay specific water quality criteria for dissolved oxygen, water clarity, and chlorophyll *a*. Collectively, these three water quality parameters provide the best and most direct measures of the impacts of too much nutrient and sediment pollution on the bay's aquatic living resources—fish, crabs, oysters, and underwater bay grasses.

Bay Criteria

Dissolved Oxygen

Fish and other aquatic life require levels of dissolved oxygen to survive. Seasonal algae blooms deplete dissolved oxygen, potentially rendering deep waters of the bay uninhabitable to certain species, such as the endangered Atlantic Sturgeon during certain times of the year. Bay dissolved oxygen levels should be those required by the aquatic communities inhabiting different parts of the bay during different times of the year, fully reflective of natural conditions.

Chlorophyll a

Measurements of chlorophyll indicate levels of phytoplankton or algal biomass in the water column. Bay chlorophyll levels should be moderate: not so high as to cause harmful algal blooms that lead to poor quality food, shading of light in shallow water habitats, and low dissolved oxygen conditions when the algae die off and sink to the bottom.

Water Clarity

Underwater grasses collectively are an essential component of the bay's living resources habitat. Decreased water clarity inhibits the growth of underwater bay grasses. Water clarity is adversely affected by increased sediment loads and algal biomass spurred by excess nutrient inputs to the bay. Bay water quality conditions should generally provide high water clarity—sunlight penetration—to support restoration of underwater grasses throughout the bay's extensive shallow water habitats.

Chesapeake Bay Dissolved Oxygen Criteria

Chesapeake Bay Dissolved Oxygen Dynamics

The Chesapeake Bay has a built-in, natural tendency toward reduced dissolved oxygen conditions, particularly within its deeper waters because of the physical morphology and estuarine circulation. Its highly productive, shallow waters, coupled with its tendency to retain, recycle, and regenerate the nutrients delivered from the atmosphere and surrounding watershed set the stage for a nutrient-rich environment. The mainstem Chesapeake Bay and its major tidal rivers with deep channels coming off shallower, broad shoal waters, and the significant influx of freshwater flows result in stratification of the water column, essentially locking off deeper bottom waters from mixing with higher oxygenated surface waters. Combined together, the retention/efficient recycling of nutrients and water column stratification lead to severe reductions in dissolved oxygen concentrations during the warmer months of the year, generally May to September.

Nearshore, shallow waters in the Chesapeake Bay also periodically experience episodes of low to no dissolved oxygen conditions, in part, resulting from intrusions of bottom water forced onto the shallow flanks by sustained winds (Carter et al.1978, Tyler 1984, Seilger et al.1985, Malone et al.1986). Diel cycles of low dissolved oxygen conditions often occur in nonstratified shallow waters where nighttime water column respiration temporarily depletes dissolved oxygen levels (D'Avanzo and Kremer 1994).

The timing and spatial and volumetric extent of hypoxic and anoxic waters vary from year to year, largely driven by local weather patterns, timing and magnitude of freshwater river flow and concurrent delivery of nutrients and sediments into tidal waters, and the corresponding springtime phytoplankton bloom (Officer et al.1984, Seliger et al.1985). In Chesapeake Bay mainstem, the onset of low to no dissolved oxygen conditions can be as early as April and persist through September, until fall turnover of the water column. The deeper waters of major tidal tributaries can exhibit hypoxic and anoxic conditions, with the nature, extent, and magnitude of low dissolved oxygen and the causative factors varying from river to river.

The scientific underpinnings of these Chesapeake Bay specific criteria have been in the works for decades. Seasonal low dissolved oxygen conditions in the Chesapeake Bay were first documented in the 1930s (Newcombe and Horne 1938). Basic understanding of dissolved oxygen dynamics, critical to derivation of criteria reflective of ecosystem process, began with the research cruises of the Chesapeake Bay Institute from the 1950s through the late 1970s. A 5-year multidisciplinary research program starting in the late 1980s, funded by the Maryland and Virginia Sea Grant Program, yielded significant advances in understanding of all facets of oxygen dynamics, effects, and ecosystem implications (Smith et al. 1991). These investigations laid the groundwork for more management-focused applications of the science.

Chesapeake Bay Dissolved Oxygen Restoration Goal

Published in 1992, the Chesapeake Bay dissolved oxygen restoration goal was developed in response to the Chesapeake Executive Council's commitment "to develop and adopt guidelines for the protection of water quality and habitat conditions necessary to support the living resources found in the Chesapeake Bay system and to use these guidelines." The dissolved oxygen restoration goal consisted of a narrative statement supported by specific target dissolved oxygen concentrations applied over specified averaging periods and locations. Dissolved oxygen effects information was compiled for 14 identified target species[1] of fish, molluscs, and crustaceans as well as for other supporting benthic and planktonic species within the bay food web. The target concentrations and their specified temporal averaging and spatial application were determined from analysis of dissolved oxygen levels that would provide the levels of protection described within the narrative restoration goal. Best professional judgment was used in areas where there were gaps in the information base on dissolved oxygen effects available a decade ago.

The original dissolved oxygen restoration goal and its supporting framework made three breakthroughs at that time of significance to supporting derivation and management application of the Chesapeake Bay specific dissolved oxygen criteria within this document. The dissolved oxygen target concentrations varied with vertical depth through the water column as well as horizontally across the expanse of the bay and its tidal tributaries, directly reflecting variations in required levels of protection for different living resource habitats. The averaging periods for each target concentration were tailored to specific habitats, recognizing that short-term exposures to concentrations below the target concentrations were allowable and still protective of living resources. The dissolved oxygen goal document contained a methodology through which water quality monitoring data and model-simulated outputs collected over varying frequencies could be directly assessed in terms of the percentage of time that areas of bottom habitat or volumes of water column habitat were predicted to meet or exceed the applicable target dissolved oxygen concentrations.

Regionalizing the EPA Virginian Province Saltwater Dissolved Oxygen Criteria

With the publication of the EPA *Ambient Water Quality Criteria for Dissolved Oxygen (Saltwater): Cape Cod to Cape Hatteras* came a decade's worth of systematically developed dissolved oxygen effect data along with synthesis and close evaluation of several decades of effects data published in the scientific

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[1] These target species were from a larger list of commercially, recreationally , and ecologically important species reported in *Habitat Requirements for Chesapeake Bay Living Resources-Second Edition* (Funderburk *et al.* 1991).

peer reviewed literature (Thursby et al.2000). The approach to derive these dissolved oxygen criteria combined features of the traditional water quality criteria with a new biological framework. A mathematical model was used to integrate time (replacing the concept of an averaging period) and establish protection limits for different life stages (i.e., larvae verus juveniles and adults). Where practical, data were selected and analyzed in manners consistent with the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (hereafter referred to as the EPA Guidelines) (Stephan et al.1985).

The EPA Virginian Province dissolved oxygen saltwater criteria document addressed three areas of protection: (1) juvenile and adult survival, (2) growth effects, and (3) larval recruitment effects. In doing so, the criteria document segregated effects on juveniles and adults from those on larvae. The survival data on the sensitivity of the juveniles and adults are handled in a traditional EPA guidelines manner. To address cumulative effects of low dissolved oxygen on larval recruitment to the juvenile life stage (i.e., larval survival as a function of time) a new biological approach was taken. These criteria were derived using a mathematical model that evaluates the effect of dissolved oxygen conditions on larvae by tracking the intensity and duration of low dissolved oxygen effects across the larval recruitment season. Protection of larvae of all species is provided by using low dissolved oxygen effects data on larval stages of nine sensitive estuarine/coastal organisms.

The Virginian Province saltwater dissolved oxygen criteria document and its underlying effects database and methodologies were structured to support regional specific derivation of dissolved oxygen criteria tailored to the species, habitats, and nature of dissolved oxygen exposure regimes of different estuarine, coastal, and marine waters. The segregation by life stages allows the criteria to be factored into the refined tidal water designated uses, which themselves, in part, reflect use of different habitats by different life stages. This segregation by life stage is a significant difference from traditional aquatic life criteria.

However, the Virginian Province saltwater criteria were not explicitly set up to address natural vertical variations in dissolved oxygen concentration. If Chesapeake Bay specific criteria were derived through a strict application of the EPA saltwater criteria methodology, there would not be the flexibility needed to tailor each set of criteria to the refined tidal water designated uses. The resultant bay criteria would be driven solely by larval effects data irrespective of depth and season.

The Chesapeake Bay specific criteria were derived through the regional application of the Virginian Province effects database and application of traditional toxicological and new biological-based criteria derivation methodologies. Chesapeake Bay specific science was factored directly into each step of the criteria derivation process. The extensive Virginian Province dissolved oxygen effects database was first focused down on only Chesapeake Bay species and then supplemented with additional Chesapeake Bay species effects data from the scientific literature. The Virginian Province larval recruitment model was modified to better reflect Chesapeake Bay conditions, with its application broadened to include additional Chesapeake Bay species. Finally, specific steps were taken to factor the requirement to provide protection of species listed as threatened/endangered in Chesapeake Bay into the bay-specific criteria.

Current State water quality standards generally require 5 mg/L of dissolved oxygen throughout all of the bay's waters—from the deep trench near the bay's mouth to the shallows at the head of the bay. Even though the 5 mg/L standard is baywide, bay region scientists believe *natural* conditions dictate that in some sections of the bay, such as the deep channel, bay waters *cannot* achieve the current 5 mg/L standard during the warmer months of the year. Additionally, scientists believe other areas, such as prime migratory fish spawning areas, require *higher* levels of dissolved oxygen to sustain life during the late winter to early summer timeframe. The amount of oxygen needed in the bay tidal waters depends on specific needs of the aquatic living resources and where they live and during which time of the year they live there.

The Chesapeake Bay dissolved oxygen criteria vary significantly across the five proposed tidal water designated uses to fully reflect the wide array of species living in these different bay habitats (Figure 1). These working draft dissolved oxygen criteria were developed by the Chesapeake Bay Dissolved Oxygen Criteria Team, a bay region team composed of scientists, State and federal managers, and technical stakeholders (Table 1). A draft document describing the Chesapeake Bay Dissolved Oxygen Criteria in greater detail is available for review and comment at www.chesapeakebay.net. There is a year-long process and schedule, including three public reviews, leading to publication of these Chesapeake Bay specific water quality criteria by EPA by June 2002.

Chesapeake Bay Chlorophyll *a* Criteria
Chlorophyll *a* is used to measure the abundance and variety of microscopic plants or algae that form the base of the food chain in the bay. Excessive nutrients can stimulate nuisance algae blooms, resulting in reduced water clarity, reduced amount of good quality food and depleted oxygen levels in deeper water. By its very nature, chlorophyll *a* is both an integrated biological measure of production of the primary food source of the entire bay food web as well as a critical indicator of water quality through its direct role in reducing light penetration and fueling bacterial processes leading to low dissolved oxygen levels. As stated upfront by Harding and Perry (1997), “chlorophyll *a* is a useful expression of phytoplankton biomass and is arguably the single most responsive indicator of N [nitrogen] and P [phosphorus] enrichment in this system [Chesapeake Bay].” Determining the levels of chlorophyll *a* which are fully protective of the refined designated uses of the vast tidal waters that compose the Chesapeake Bay and tributaries must factor in all the different roles chlorophyll *a* plays in defining a restored, more balanced Chesapeake Bay ecosystem.

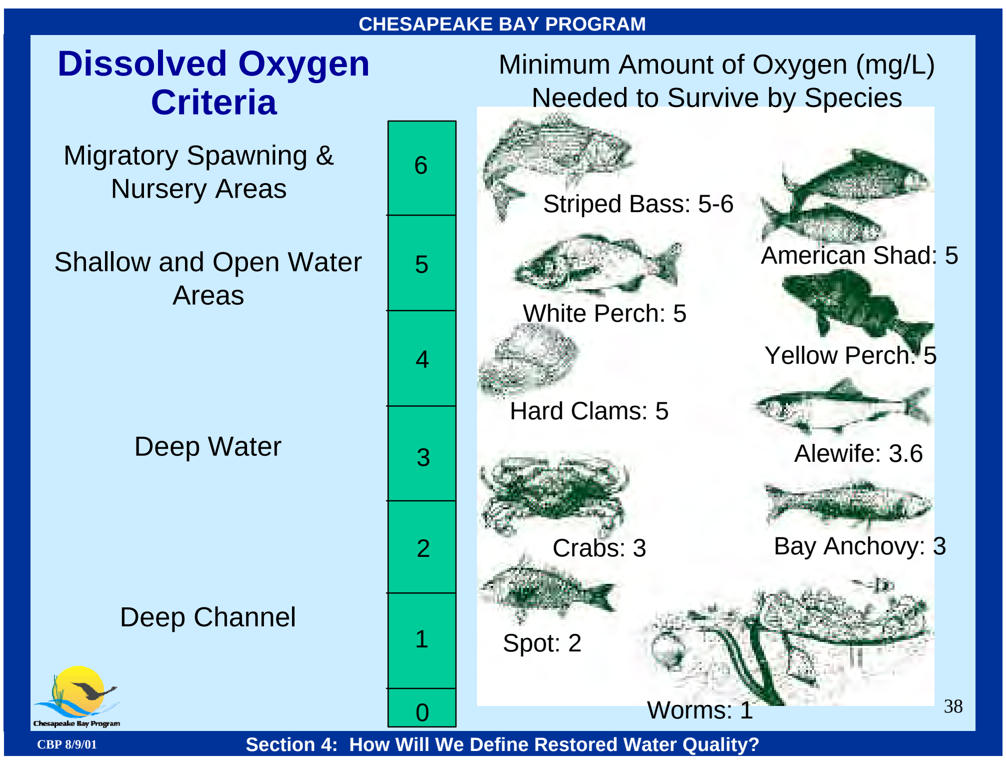


Figure 1. Dissolved Oxygen Criteria, Chesapeake Bay.

Table 1. Working Draft Chesapeake Bay Dissolved Oxygen Criteria (July 3, 2001)

Designated Use	Criteria Concentration/Duration	Temporal Application
Migratory spawning and nursery	7 day mean of 6 mg/L[a]	February 15th - June 10th
	Instantaneous minimum of 5 mg/L	
	30 day mean of 5 mg/L	June 11th - February 14th
	7 day mean of 4 mg/L	
	Instantaneous minimum of 3.5 mg/L	
Shallow/open water	30 day mean of 5 mg/L	All year round
	7 day mean of 4 mg/L	
	Instantaneous minimum of 3.5 mg/L	
Deeper water	30 day mean of 3 mg/L	April through September
	Instantaneous minimum of 1.7 mg/L	
	30 day mean of 5 mg/L	October through March
	7 day mean of 4 mg/L	
	Instantaneous minimum of 3.5 mg/L	
Deep channel	Instantaneous minimum of 1 mg/L	April through September
	30 day mean of 5 mg/L	October through March
	7 day mean of 4 mg/L	
	Instantaneous minimum of 3.5 mg/L	

[a] Applied to tidal fresh waters with long term averaged salinities less than 0.5 parts per thousand.

The derivation of the Chesapeake Bay chlorophyll *a* criteria were based on the convergence of several independent lines of evidence—historical observed concentrations, literature values related to trophic status, direct contributions to light attenuation, and contribution to dissolved oxygen conditions—collaborating chlorophyll *a* concentrations derived as a result of characterizing set of phytoplankton reference communities.

Phytoplankton Reference Community/Food Quality Connection

Estimates of phytoplankton taxon biomasses were derived from the Maryland and Virginia Chesapeake Bay Monitoring Program phytoplankton count data (1984-1999) and along with other phytoplankton indicators—chlorophyll *a*, pheophytin, and primary productivity—were used to investigate differences in biomass, taxonomic composition, and food value for the range of water quality conditions currently experienced in the Chesapeake Bay. The biological data were sorted into categories based on season- and salinity-specific concentrations/levels of three parameters in the associated water quality data: dissolved inorganic nitrogen, ortho-phosphate and Secchi depth. Relatively small secchi depths and excess dissolved inorganic nitrogen and excess ortho-phosphate characterized the Poor water quality categories. Relatively high light levels and algal growth-limiting concentrations of dissolved inorganic nitrogen and ortho-phosphate characterized the good water quality categories. Mixed water quality conditions (i.e., one or two water quality parameters qualified as Better but the other(s) did not) and extreme subsets of the Poor and Better categories (i.e. Worst and Best) were also investigated. Qualitative and quantitative measures of the phytoplankton community composition and biomass distributions were then evaluated relative to these water quality classifications and implications for food quality and quantity for filter feeding fish and shellfish.

Historically Observed Concentrations

Several recent in-depth reviews and evaluations of historically observed (1950s to early 1980s) and current (1984-1998) chlorophyll *a* concentrations provide a strong basis for collaborating the Chesapeake Bay specific chlorophyll *a* criteria (Harding 1994, Harding and Perry 1997, Olson and Lacouture, in review). Using information from five decades of water quality data provides insights into both chlorophyll *a* concentrations that are attainable under a range of otherwise natural conditions (meteorological, river flow, tidal flushing) as well as concentrations reflective of a healthier bay ecosystem.

Literature Values Related to Trophic Status

Throughout the scientific literature, there are several defining papers which through synthesis of a wide array of data from many different aquatic systems center down on ranges of conditions reflective of different trophic states of water bodies (e.g., Wetzel 1985, Ryding and Rast 1989, Smith et al.1992). Chlorophyll *a* is a principal parameter quantified within these literature reviews. The strength of this collaborative line of evidence is that information is drawn from diversity of systems across the spectrum of healthy to clearly eutrophied water bodies. This approach provides insights into common characteristics associated with trophic status that can not be drawn through the study of a single, although large, water body like Chesapeake Bay.

Direct Contributions to Light Attenuation

Over the past four decades, the Chesapeake Bay ecosystem has had an extensive, widely distributed underwater grass community undergo severe declines followed by a decade and a half slow but steady recovery. The bay management and scientific communities have invested significant resources in the investigation of this grand natural experiment, learning much about the causes of the decline and potential solutions for continued, yet accelerated restoration. Two comprehensive technical synthesizes of this wealth of scientific knowledge and insights have been published which provide direct quantitative insights into the role of chlorophyll *a* in the recovery of underwater bay grasses (Batiuk et al.1992, 2000). This collaborative line of evidence draws on the chlorophyll *a* connection to reductions in light penetration through the water column.

Contribution to Dissolved Oxygen Conditions

It is well known and documented that algae uneaten by higher trophic levels—zooplankton, oysters and fish of all kinds—becomes the fuel, through its breakdown by bacteria, for reducing dissolved oxygen levels. Through an analysis of Chesapeake Bay water quality model simulated outputs from scenarios which simulated dissolved oxygen conditions which met the dissolved oxygen criteria, the model simulated chlorophyll *a* concentrations of desired dissolved oxygen conditions were quantified.

Appropriate chlorophyll *a* levels vary, depending on the salinity of the water. The proposed criteria for chlorophyll *a* are split out from tidal freshwater all the way to very salty—polyhaline—waters. Season of the year is also important, with spring and summer being the most important times of year that high chlorophyll *a* levels can impact living resources in the bay.

These working draft chlorophyll *a* criteria were developed by the Chesapeake Bay Chlorophyll and Nutrient Criteria Team, a bay region team composed of scientists, State and federal managers, and technical stakeholders (Table 2). A draft document describing the Chesapeake Bay Chlorophyll *a* Criteria in greater detail is available for review and comment at www.chesapeakebay.net. There is a

Table 2. Working Draft Chesapeake Bay Chlorophyll *a* Criteria (July 3, 2001)

Salinity Regime	Chesapeake Bay Chlorophyll Criteria (ug/L)			
	Spring (March-May)		Summer (July-September)	
	Median	Maximum	Median	Maximum
Tidal Fresh	8	12	9	16
Oligohaline	10	23	6	23
Mesohaline	6	27	7	16
Polyhaline	3	7	4	9

year-long process and schedule, including three public reviews, leading to publication of these Chesapeake Bay specific water quality criteria by EPA by June 2002.

Connection to Underwater Bay Grasses

The loss of submerged aquatic vegetation, or SAV, from shallow waters of Chesapeake Bay, which was first noted in the early 1960s, is a widespread, well-documented problem. Although other factors, such as climatic events and herbicide toxicity, may have contributed to the decline of SAV in the bay, the primary causes are eutrophication and associated reductions in light availability. The loss of SAV beds are of particular concern because these plants create rich animal habitats that support the growth of diverse fish and invertebrate populations. Similar declines in SAV have been occurring worldwide with increasing frequency during the last several decades. Many of these declines have been attributed to excessive nutrient enrichment and decreases in light availability.

Chesapeake Bay Water Clarity Criteria

One of the major features contributing to the high productivity of Chesapeake Bay has been the historical abundance of SAV. There are over 20 freshwater and marine species of rooted, submerged flowering plants in bay tidal waters. These underwater grasses provide food for waterfowl and are critical habitat for shellfish and finfish. SAV also affect nutrient cycling, sediment stability, and water turbidity.

The health and survival of these plant communities in Chesapeake Bay and its tidal tributaries depend on suitable environmental conditions that define the quality of SAV habitat. Key to the restoration of these critical habitats and food sources is the return of levels of light penetration in shallow waters necessary to support the survival, growth, and repropagation of diverse, healthy underwater bay grass communities.

Bay Water Clarity Derivation Approach

Through the combined efforts of the bay's scientific and resource management communities, two internationally recognized technical syntheses of information supporting quantitative habitat requirements for Chesapeake Bay SAV have been published in the past decade (Batiuk et al.1992, Batiuk et al. 2000). Key findings, the underlying light requirements, and management-oriented diagnostic tools and restoration targets have been reported in the peer reviewed scientific literature (Dennison et al. 1993, Kemp et al. in review; Gallegos 2001, Koch 2001, Bergstrom in preparation, Carter and Rybicki in preparation, Karrh in preparation, Kemp et al. in preparation). These two technical syntheses of worldwide literature, bay-specific research and field studies, and recent model simulation and data evaluation provide the scientific foundation for the Chesapeake Bay water clarity criteria described here. Readers are encouraged to consult these two syntheses and the resultant scientific literature papers for more in-depth technical details and documentation.

The Chesapeake Bay specific water clarity criteria derivation follows four successive stages: first, determination of water-column-based light requirements for SAV survival and growth, then quantification of the factors contributing to water column light attenuation. The contributions from epiphytes to light attenuation at the leaf surface are then factored into methods for estimating total light attenuation. Finally, a set of minimal light requirements are determined as the actual criteria values.

The draft bay criteria propose that water clarity criteria should apply to areas of the bay that are up to 2 meters deep (approximately 6 feet). Areas where SAV never occurred historically, or where natural factors prevent its growth (e.g., strong currents, rocky bottoms) would be excluded. The water clarity criteria reflect the different light requirements for underwater plant communities that inhabit low salinity versus higher salinity shallow water habitats throughout the bay (Figure 2).

These working draft water clarity criteria were developed by the Chesapeake Bay Water Clarity Criteria Team, a bay region team composed of scientists, State and federal managers, and technical stakeholders (Table 3). A draft document describing the Chesapeake Bay Water Clarity Criteria in greater detail is available for review and comment at www.chesapeakebay.net. There is a year-long process and schedule, including three public reviews, leading to publication of these Chesapeake Bay specific water quality criteria by EPA by June 2002.

These Chesapeake Bay criteria will be applied to a series of designated uses which, in turn, reflect key habitats throughout the bay and its tidal tributaries.

Excessive Nutrient and Sediment Loads

The causes of these water quality impairments—excessive loadings of nitrogen, phosphorus, and sediment—will be addressed through commitments to determine reductions in loadings needed to achieve the bay criteria. These required loading reductions will be established as caps on loadings

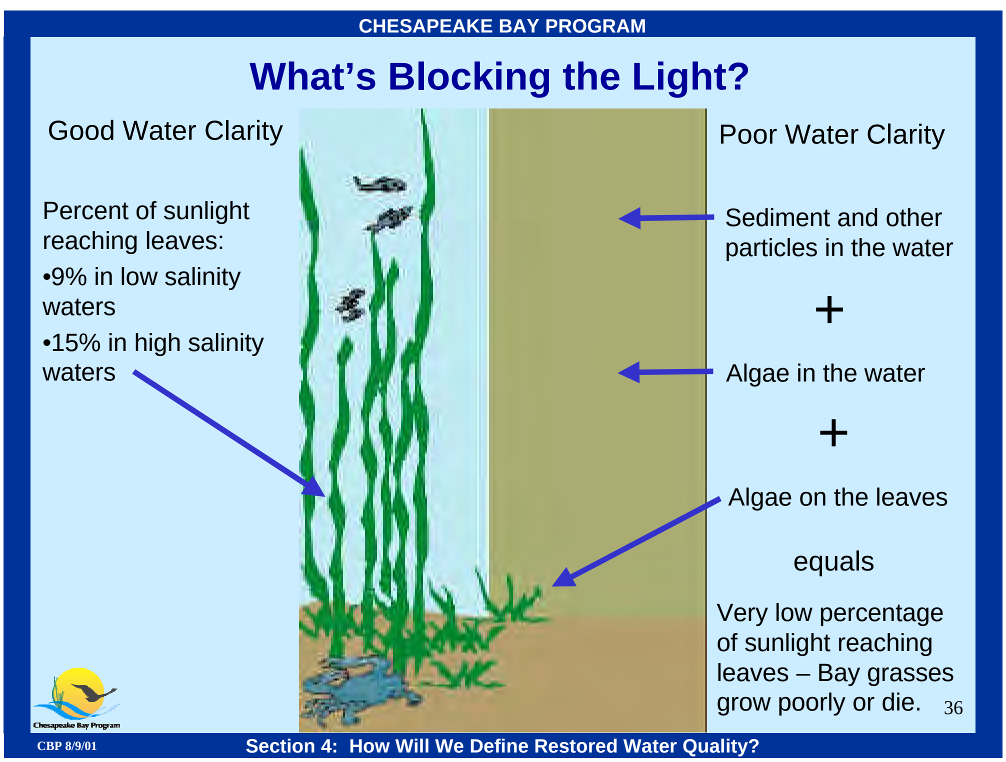


Figure 2. Bay Water Clarity.

Table 3. Working Draft Chesapeake Bay Water Clarity Criteria (July 3, 2001)

Habitat Category	Criteria Concentration (percent ambient light)	Temporal Application
Tidal fresh shallow water	9 %	April - October
Oligohaline shallow water	9 %	April - October
Mesohaline shallow water	15 %	April - October
Polyhaline shallow water	15%	March-May, Sept-Nov.

allocated to each tributary basin within the Chesapeake Bay watershed. This approach is consistent with EPA's regional establishment of ambient concentration-based nutrient criteria, but places more emphasis on the water quality parameters with a direct impact on aquatic living resources. Through this approach nutrients and sediments are addressed directly through caps on loading determined through application of the linked bay airshed-watershed-tidal water quality models and analysis of Chesapeake Bay Monitoring Program data in place of the development of ambient nutrient and sediment criteria.

Relationship with National Efforts to Develop Nutrient Criteria

At the same time, a parallel effort is currently underway by EPA to develop ecoregion specific numerical nutrient criteria across the country to meet the objectives of the Clean Water Action Plan. A nutrient criteria team has been established by EPA-Region III to implement the National Nutrient Strategy issued by EPA last year for the mid-Atlantic region. The EPA Region III team is focusing its nutrient criteria development efforts on the free flowing stream, rivers, lakes, and wetlands within the mid-Atlantic States, not the Chesapeake Bay tidal waters. Whereas the bay criteria are focused on dissolved oxygen, water clarity and chlorophyll *a*, the EPA Region III team is developing ambient concentration criteria for total nitrogen, total phosphorus, chlorophyll, and turbidity.

The advanced scientific understanding of water quality impacts on aquatic bay living resources combined with the state of the art linked bay airshed-watershed -water quality models enabled the bay watershed partners to develop criteria for water quality measures directly influencing aquatic resources. The cause of reduced water quality conditions—too much nitrogen, phosphorus, and sediment—will be addressed through the establishment of loading caps. The bay models enable the partners to effectively translate the desired dissolved oxygen, water clarity, and chlorophyll *a* conditions back into reduced loadings of nutrients and sediments from the surrounding watershed and airshed. Bay science has shown that it is the delivered loads of nutrients and sediment, not just the ambient concentrations, that have had an impact on oxygen, light, and algae levels in the bay tidal waters.

Bay Tidal Water Designated Uses

Because conditions throughout the Chesapeake Bay tidal water habitats differ based on depth, salinity and season, a uniform baywide water quality standard does not take into account the varying needs of different plants and animals. As a result, current State water quality standards, which differ between the four jurisdictions with tidal waters, need to be revised and expanded to account for the natural variability in conditions found throughout the bay. Each of the three bay criteria will differ from one region of the bay and its tidal tributaries to another, as determined by the plants and animals residing in that area. Once the bay criteria and the tidal water designated uses are adopted as State water quality standards, these tailored set of standards will apply to similar habitats across all jurisdictions.

An area's designated use refers to a waterbody's function—such as fishable or swimable—and takes into account the use of the water body for public water supply, the protection of fish, shellfish, and wildlife, as well as its recreational, agricultural, industrial and navigational purposes. The existing Maryland, Virginia, Delaware, and District of Columbia designated uses for the bay's tidal waters do not fully reflect the wide variety of different habitats found throughout the bay and its tidal tributaries. Where two jurisdictions boundaries join, each State has different designated uses for the same waterbodies.

Five refined Chesapeake Bay tidal water designated uses have been established to more fully reflect the different aquatic living resource communities inhabiting a variety habitats and, therefore, the different intended aquatic life uses of those tidal habitats (Figure 3).

The *Migratory Spawning & Nursery* designated use is the propagation and growth of balanced indigenous populations of ecologically, recreationally, and commercially important anadromous, semi-anadromous, and tidal fresh resident fish species inhabiting spawning and nursery grounds.

The *Shallow Water* designated use is the propagation and growth of balanced, indigenous populations of ecologically, recreationally, and commercially important fish, shellfish and underwater grasses inhabiting shallow waters habitats.

The *Open Water* designated use is the propagation and growth of balanced, indigenous populations of ecologically, recreationally, and commercially important fish, and shellfish species that inhabit open water habitats.

The *Deep Water* designated use is the propagation and growth of balanced, indigenous populations of ecologically, recreationally, and commercially important fish and shellfish species inhabiting deep water habitats.

The *Deep Channel* designated use is to provide a refuge for balanced, indigenous populations of ecologically, recreationally, and commercially important fish species that depend on deep channel

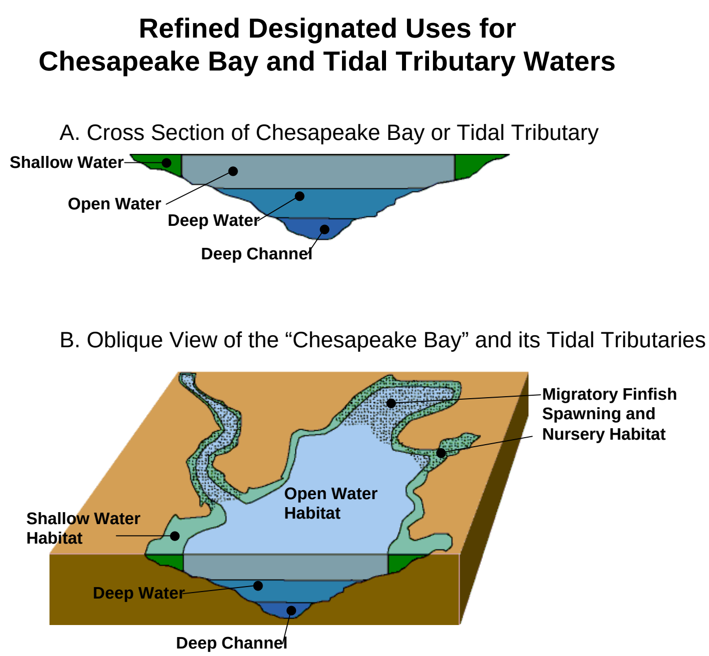


Figure 3. Refined Designated Uses for Chesapeake Bay Tidal Tributary Waters.

habitats for overwintering during colder months of the year and the propagation and growth of benthic infaunal and epifaunal worms and clams that provide food for bottom feeding fish and crabs.

These tidal water designated uses were developed by the Chesapeake Bay Water Quality Standards Coordinators Team, a bay region team composed of water quality standards coordinators from all six States, the District of Columbia, EPA Region 2, 3, and headquarters offices. Table 4 shows how refined tidal water designated uses relate to the bay criteria.

The watershed partners are evaluating the refined tidal water designated uses and the applicable bay criteria through a baywide use attainability analysis. The final tidal water designated uses will be adopted by Maryland, Virginia, Delaware, and the District of Columbia, along with the applicable bay water quality criteria into their State water quality standards by 2003. These refined designated uses will add more specifics to the existing State designated uses and apply consistently across jurisdictions for similar habitats.

Baywide Use Attainability Analysis

The Chesapeake 2000 Agreement commits the States with bay tidal waters—Maryland, Virginia, and Delaware—and the District of Columbia to adopt into their State water quality standards as consistent set of bay criteria and designated uses across bay tidal habitats. Whenever there is a proposed change in water quality standards, such as that being undertaken for Chesapeake Bay waters, it is necessary to assess attainment of the designated uses and underlying criteria. Such an assessment is called a Use Attainability Analysis or UAA.

A UAA is used by States to justify changes to their water quality standards by assessing the physical, chemical, biological, economic, or other factors affecting attainment of the designated use. The UAA describes the scientific attributes of the waterbody, both natural conditions and conditions brought about by human contribution. If the attributes of the waterbody make attaining the use impossible, or if there are economic reasons why the use cannot be attained, the UAA is used to clearly document these reasons. Finally, the UAA describes how the proposed standards will protect existing uses. All six bay watershed States—New York, Pennsylvania, Maryland, Virginia, West Virginia and Delaware—along with the District of Columbia and EPA are working together with bay watershed partners to carry out such an

Table 4. Chesapeake Bay Criteria Needed for Protection of the Proposed Tidal Waters Designated Uses

	Dissolved Oxygen	Chlorophyll *a*	Water Clarity
Migratory Spawning and Nursery	✔	✔	
Shallow Water	✔	✔	✔
Open Water	✔	✔	
Deep Water	✔		
Deep Channel	✔		

assessment. A use attainment assessment on a scale as large as the 64,000 square mile Chesapeake Bay watershed has never been carried out.

Adopting Bay Criteria as State Water Quality Standards

Water quality standards combine water quality criteria and designated uses to produce a target numeric value assigned to a waterbody that, if achieved, will maintain healthy water quality.   Through the Chesapeake 2000 Agreement, Maryland, Virginia, and the District of Columbia are committed to adopting the new bay criteria–dissolved oxygen, water clarity, and chlorophyll *a*—along with the refined tidal water designated uses as State water quality standards. Delaware, which shares bay tidal waters with Maryland in its portion of the Nanticoke River watershed, has made the same commitment through the six-State memorandum of agreement. Together, the States and the District must achieve these new bay-specific water quality standards needed to support restored estuarine ecosystem if the Chesapeake Bay is to be removed from the list of impaired waters.

Why New State Standards

Existing State water quality standards are applied broadly across each State's tidal waters, without recognition of the variety of habitats. Each State has different water quality standards applied to the same tidal waters, whereas the aquatic living resources in bay habitats, which do not recognize these jurisdictional boundaries, may have the same water quality needs. Currently, dissolved oxygen is the only numerical water quality standard adopted by all three States and the District of Columbia that addresses nutrient- and sediment-related water quality pollution problems.

So compliance with existing State water quality standards will not fully protect the living resources in the bay waters. In some critical habitats of the bay, specifically migratory fish spawning and nursery areas, existing State water quality standards will not fully protect more sensitive life stages. In other cases, reaching existing standards is not possible owing to natural conditions found in deeper bay waters during the warmer months of the year. Existing State water quality standards do not include measures to protect underwater bay grasses or fully support good quality fish food.

Setting and Allocating New Cap Loads

The Chesapeake 2000 Agreement commits the signatories to determining the nutrient and sediment load reductions necessary to achieve the water quality conditions that protect aquatic living resources. Those load reductions will then be assigned or allocated to each major tributary basin in the form of cap loads. Cap loads are the maximum amounts of pollutants allowed to flow into a waterbody and still ensure achievement of State water quality standards. In this case, the water quality standards will be the new bay criteria and refined tidal water designated uses (currently in draft) to be adopted by the States of Maryland, Virginia, and Delaware, and the District of Columbia into their standards by 2003.

Available Tools and Information

The Chesapeake Bay watershed partners will use the Chesapeake Bay Airshed, Watershed and Estuary Models, the USGS SPARROW model, along with Chesapeake Bay Monitoring Program data, to help determine these cap loads for nitrogen, phosphorus, and sediment. These models are mathematical representations that simulate the real world, interpreting various levels of actions (management scenarios) to reduce different amounts of pollutant loads. These scenarios are run through the models to determine how to achieve baywide attainment of the bay water quality criteria for dissolved oxygen, water clarity, and chlorophyll *a* as applied to the tidal water designated uses.

Cap Setting and Allocation

These models and other available information will be used to allocate loading caps to the nine major tributary basins–Susquehanna, Upper Western Shore, Patuxent, Potomac, Rappahannock, York, James, Upper Eastern Shore, and Virginia Eastern Shore (Figure 4). Each State and the District will bear a proportional burden for achieving and maintaining the cap based on their existing pollutant loadings, progress to date, effectiveness and cost efficiency considerations, and their pollutant loading effects on different tributaries.

For multijurisdictional waters like the Susquehanna, Potomac, and Eastern Shore basins, the linked watershed and bay water quality models will be used to further allocate cap load responsibilities to each State.

Working with their local stakeholders, individual States to further subdivide their major tributary basin load cap allocations into the 37 State-defined tributary strategy sub-basins.

A comprehensive 2-year schedule has been set up to coordinate the efforts of the six watershed States, the District of Columbia, and the many other involved bay watershed partners. The schedule also will ensure direct and continued involvement of local stakeholders and the general public during the entire cap load setting and allocation process. The Chesapeake Bay Water Quality Steering Committee, composed of senior managers from the seven watershed jurisdictions, EPA regional and headquarters offices, Chesapeake Bay Commission, river basin commissions, and involved stakeholders, has the overall responsibility overseeing and reaching agreement on the cap load allocations. Many of the subcommittees and workgroups within the Chesapeake Bay Program committee structure will be carrying out the technical, modeling, data interpretation, economic analysis, policy evaluation, and communication work in support of setting and allocating the nutrient and sediment cap loads.

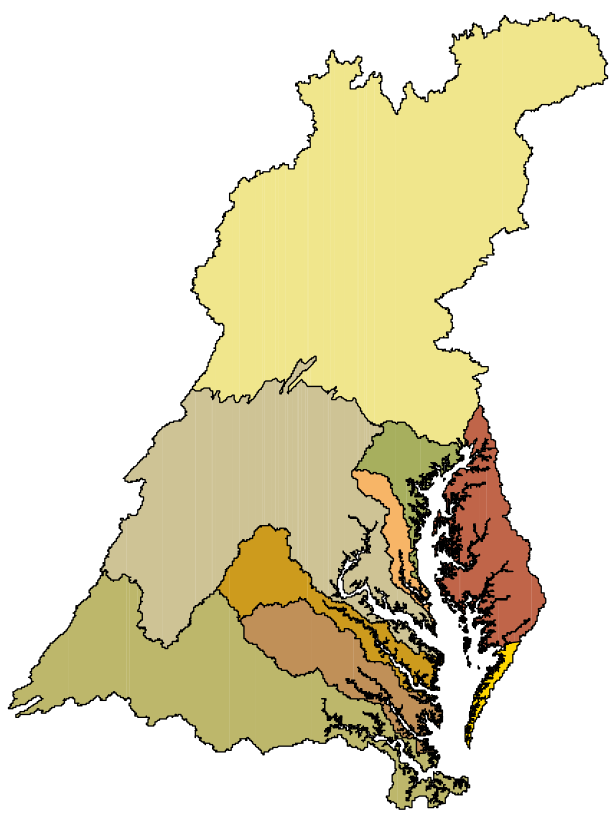


Figure 4. The Nine Major Basins.

Tributary Strategies: Local Watershed Implementation

The Chesapeake 2000 Agreement commits the bay watershed partners to "complete a public process to develop and begin implementation of revised Tributary Strategies to achieve and maintain the assigned loading goals." Tributary strategies are detailed descriptions of planned local actions—riparian forest buffer replanting, wastewater treatment upgrades, nutrient management on farms, stormwater treatment, stream restoration, and many others—and a schedule for undertaking those actions necessary to reduce nutrients and sediment loads from each tributary watershed to reach the assigned loading cap by 2010.

Development of tributary strategies has been a very driven public process with the direct involvement by local governments, watershed associations, regional organizations, and a wide variety of other interested local stakeholders. In creating the strategies, the States, and the District of Columbia work closely with those groups and individuals within each respective watershed who will be directly involved in implementation strategy. Together, they explore and evaluate a wide variety of point and nonpoint source pollution control measures. They then draft a strategy using the most effective reduction options to achieve the cap load allocated to their tributary strategy basin.

The existing Tributary Strategies were designed to achieve the 1987 Bay Agreement goal of a 40 percent reduction in nutrient loads from controllable sources from 1985 levels. Copies of these existing tributary strategies are available on-line through the respective Maryland, Virginia, Pennsylvania, and the District of Columbia tributary strategy web pages.

To restore the tidal water conditions necessary to sustain the bay's fish, crabs, oysters and underwater grasses will likely require greater reductions in nutrients in many areas then called for by the existing tributary strategies. In addition, the water clarity conditions needed to restore underwater bay grasses can not be achieved without significant reductions in sediments loads to the tidal waters. With New York, Delaware and West Virginia joining as bay watershed partners through a six-State memorandum of understanding, new tributary strategies will be developed for these States' portions of the bay watershed not previously addressed under the existing tributary strategies.

The new and revised tributary strategies will now cover all sources of nutrient and sediment pollution, including air sources, across the entire 64,000 Chesapeake Bay watershed. Tributary strategies will address nutrient and sediment loading caps allocated to 37 sub-basins across the bay watershed by the bay watershed partners. Loading reductions required through local stream and river segment regulatory TMDLs will be directly integrated into each respective Tributary Strategy as part of the overall effort to effectively "blend" the regulatory TMDL program and cooperative Chesapeake Bay Program.

Information on Local Watersheds

A wide array of information is available on local watersheds within the Chesapeake Bay basin, including information directly relevant to the overall process for setting and allocating new cap loads through the Chesapeake Watershed Profiles. Through this point and click information system linking the bay watershed partners, one can access information on pollution sources, recent modeling results, status of bay criteria attainment, long-term trends in water quality and living resources, draft cap allocations, and more from the entire bay basin scale to local watersheds.

CASE STUDY
A PERSPECTIVE FROM WASHINGTON STATE

Jan Newton, Washington State Department of Ecology; Randy Shuman, King County Department of Natural Resources; Greg Pelletier, Washington State Department of Ecology

The issue of nutrient control for marine receiving waters in Washington State has its origin in a landmark case concerning freshwater eutrophication and lake restoration. In the early 1950's, Lake Washington, a large lake (85 km2) situated near Seattle, Washington, was showing warning signs of ecological deterioration. Following unregulated dumping of sewage from a growing urban population into the lake, classic signs of eutrophication were observed, including blooms of Oscillatoria rubescens, reduced water transparency, and very low nitrogen to phosphorus concentration ratios. The situation was studied extensively by Dr. W. T. Edmondson, a professor at the University of Washington, who explained that the changes in the lake were directly attributable to nutrient loading from sewage and wastewater (Edmondson, 1991). Edmondson made these facts known not only to the scientific community but also to the public and local government. The case resulted in the diversion of sewage away from the lake and is a classic example of how scientific observations and understanding were used to shape public policy. The sewage diversion and subsequent lake recovery were a success, with current day water quality of the Lake far exceeding that observed during the 1950's-60's.

The solution to this classic case was to divert wastewater from Lk. Washington to nearby Puget Sound, a large inland sea linked to the Pacific Ocean via the Strait of Juan de Fuca. Studies by Dr. G. C. Anderson, an oceanographer at the University of Washington, showed that phytoplankton in the Sound in the vicinity of the outfall proposed to handle the diverted wastewater were limited by light and mixing, not by nutrients. Thus, diversion of effluent from the Lake to the Sound was not "shifting the problem" but rather was an ecologically sound solution. This understanding of both limnology and oceanography laid the conceptual foundation for the formation of a large publicly funded agency (Municipality of Metropolitan Seattle, or "Metro"). It was proposed that Metro would build a new outfall at West Point, on the Main Basin of Puget Sound, divert the sewage from Lk. Washington to West Point, and thus eliminate the nutrient enrichment problem. However, the mandate to create Metro to carry forward these actions had to be approved in a public election first. Much controversy was associated with the process chronicled well in Edmondson's book "The Uses of Ecology" (Edmondson, 1991). Among numerous lines of objection, some public opinion maintained that Puget Sound's ecological health would be destroyed, despite Anderson's observations. The proposed action required two election attempts before it was approved in 1958. It is notable that the election passed before deterioration of Lk. Washington water quality was serious; certainly the local conditions were not as serious as the symptoms seen in lakes in Europe or the Midwest North America. However, deterioration of Lk. Washington conditions did continue during the five years before the Metro diversion construction commenced. The persistent, dense, and obnoxious populations of Oscillatoria galvanized public opinion that the Metro diversion was necessary. Shortly after construction of the diversion, Lk. Washington water quality improved.

The notoriety of this event and the success of the outfall constructed at West Point to not exhibit observable biological changes in Puget Sound resulted in widespread lore that "Puget Sound" cannot be eutrophied because the marine waters are not sensitive to nutrient addition. As Anderson's observations implied, the reason for the success of West Point outfall owes to the deep, well-mixed waters at the site which are flushed with a residence time on the order of days. Density-driven stratification is minimal, the phytoplankton are well mixed, and any depth gradients of oxygen and nutrients do not persist. The outfall, at 71 m depth, diffuses effluent into water that has naturally high concentrations of nitrate and this additional nitrogen burden is thought to not significantly contribute to phytoplankton nutrition.

Ammonium concentrations in excess of normal Puget Sound background levels are observed sporadically (King County 2001) near the site and may be associated with the effluent.

This example remains well-known, but it is important to note that not all of the reaches, bays, and inlets of Puget Sound have the same characteristics of West Point and the Main Basin. Greater Puget Sound is actually composed of several basins: South Puget Sound, Hood Canal, Whidbey Basin, the Main Basin, and Admiralty Inlet with its adjoining waters with the Strait of Juan de Fuca. The first three of these basins have considerable freshwater-induced stratification and are much less well-flushed than the Main Basin. Residence times of these basins range weeks to months. A study funded by EPA to provide a general review of the state of knowledge regarding nutrient-phytoplankton relations and quantify the relative nutrient sensitivity of various areas in the Sound highlighted several areas where nutrients became deplete and N:P ratios suggested nitrogen limitation (Rensel, 1991). Evidence from C-14 uptake experiments have shown that enhancements of primary production over ambient rates due to added nitrogen nutrient can be as high as 300% in South Hood Canal (Newton et al. 1995) and 83% in Budd Inlet, located in South Puget Sound near Olympia (Newton et al. 1998). Based on environmental attributes and human growth indicators, only a few places within Puget Sound were judged to be currently exhibiting signs of eutrophication; however, numerous places, particularly in South Puget, Hood Canal, and Whidbey basins, were assessed to be highly susceptible to future deterioration from eutrophication (Bricker et al. 1999).

Perhaps the first place within Puget Sound to gain wide attention for nutrient enrichment effects was southern Puget Sound, including Budd Inlet, Oakland Bay, Eld Inlet, Henderson Inlet, Case Inlet, and Carr Inlet. The Washington State Department of Ecology sponsored two studies in the 1980s to evaluate the acceptability of secondary-treated wastewater discharges to marine waters in southern Puget Sound (URS, 1985, URS, 1986a). This work identified areas where new or expanded discharges were unacceptable, based on the potential for eutrophication. A simple screening model based on effluent dilution and flushing was developed to identify the most sensitive areas.

Wastewater discharge into Budd Inlet was implicated in causing nuisance blooms of phytoplankton and adding to low dissolved oxygen concentrations noted in the bottom waters at the head of the inlet (URS, 1986b). Studies sponsored by the Washington State Department of Ecology developed the first numerical models to relate the loading of nitrogen to phytoplankton blooms and dissolved oxygen in Budd Inlet. This work was the impetus for construction of advanced wastewater treatment systems to remove dissolved inorganic nitrogen from municipal wastewater from the regional facility that discharges to Budd Inlet. In 1994, the wastewater entering Budd Inlet was treated for nitrogen removal. This has resulted in substantially lower ambient nitrogen concentrations (Eisner and Newton 1997). However, regional growth continues and more capacity to process effluent is needed. Currently, the Lacey-Olympia-Tumwater-Thurston County Wastewater Management Partnership (LOTT) is considering numerous alternatives to meet this need, including one proposal to discharge more effluent in winter when phytoplankton growth is minimal. LOTT undertook a large modeling and observational study: the Budd Inlet Science Study, which will be used to make the final permitting decisions.

Nutrient-induced increases to phytoplankton production with subsequent drawdown of bottom-water oxygen has a unique character in Puget Sound relative to other North American systems because of the influence of upwelled Pacific Ocean water. Upwelling favorable conditions off the Washington coast lead to upwelling of deep ocean waters. These relatively low-oxygenated waters are transported in, landwards, at depth through the Strait of Juan de Fuca, as the estuarine flow of Puget Sound waters flow out, seawards, at the surface. The oceanic waters entering the Puget Sound system through Admiralty Inlet can have oxygen concentrations as low as 5 mg/L, which then spread into the Main Basin and form the bottom-water of the other basins as well. Seasonally low deepwater oxygen concentrations can be found throughout Puget Sound (Newton et al., 1998; King County, 2001). Upwelling is mostly favorable in late summer, when productivity-related oxygen deficits are also maximal. The additive nature of human-caused eutrophication oxygen drawdown to this natural low oxygen quality, results in a smaller margin of error before deleterious effects would be noted.

The success of discharging nutrients into marine waters via regional Puget Sound wastewater plants, including the West Point facility, may be at a scale that has reached capacity. The Seattle metropolitan area continues to grow and new Puget Sound regional wastewater facilities are needed. The King County Council recently approved that a new facility will be required to meet growth demands and is projected for completion in 2010. The new facility will also discharge into the Main Basin, to the north of the West Point facility. King County is currently leading an extensive study to investigate impacts from nutrient loading on the area. These studies include extensive water quality sampling, experiments on the susceptibility of the waters to nutrient additions and modeling experiments to predict future impacts.

The story of nutrient control in Puget Sound continues to evolve. Ecologically sound decisions regarding nutrient management are dependent on two variables: (1) the population producing the effluent, its size, growth rate, and scale relative to the receiving waters; and (2) the sensitivity of the particular region of the Sound where the effluent is to be discharged to nutrients. Unfortunately, both of these attributes are highly variable within the Puget Sound regional area, making nutrient management decisions a challenge and arguing for the utility of careful scientific studies in companion with planning efforts.

APPENDIX A
CONDITIONS FOR BLOOM DEVELOPMENT: INTERPLAY AMONG BIOGEOCHEMICAL, BIOLOGICAL, AND PHYSICAL PROCESSES

Overview

In temperate estuaries, the spring bloom typically is dominated by diatoms and occurs when freshwater delivers adequate amounts of N, P, and Si and other nutrients to the system. In deep estuaries, the spring freshwater inflow also provides for vertical density stratification where enough of the euphotic zone resides near the surface to allow phytoplankton to achieve net biomass production (i.e., total photosynthesis minus respiration is positive). During this time solar insolation and water temperature are on the increase. For a bloom to develop, other conditions must be met. Physical flushing and local dispersion of a water parcel must be less than the doubling time for cells. Biological grazing rate must not be so large as to consume phytoplankton faster than cells' doubling time.

Biogeochemical Processes

Three aspects of N and P biogeochemistry help explain whether N or P dominates nutrient limitation in estuaries (i.e., relative nitrogen fixation rates, denitrification, and sediment regeneration of P). The first aspect involves evidence suggesting that nitrogen fixation is less effective in marine than in freshwater systems in making up nitrogen deficits (Howarth 1988; Schindler 1974). This finding has major implications for long-term coastal and open-ocean nutrient overenrichment, because N fixation is so inefficient that any balance in the N versus P limitation occurs in terms of geological time.

The second aspect involves one of the greatest differences in nutrient biogeochemical cycles between freshwater and marine systems. The superior capacity of freshwater versus marine systems to retain P in sediments through interactions with iron has profound implications. Nearly all the P deposited in marine sediments is remineralized annually (Caraco et al. 1990) and depends heavily on the sulfate concentration, which can be used as a surrogate for salinity. Thus, P in freshwater sediments is bound more tightly, and proportionally less is released back into the water column. Also, P release from marine sediments is temperature dependent, and its maximum release during the summer helps explain the tendency for increasing water column concentrations of P to occur during that season in many estuaries (Nixon et al. 1980). Estuaries with a well-developed tidal freshwater zone might be expected to be more P-limited than estuarine systems with small tidal fresh areas.

More details on nutrient cycling in subtropical and tropical marine waters, systems much less studied than northern temperate estuaries, are provided by Bianchi et al. (1999). The extensive coastal wetland systems (e.g., marshes and mangroves) that border the Gulf of Mexico provide environments where chemical transformations and storage of nutrients occur. Also, extensive seagrass meadows apparently tie up inorganic N and P so that relatively less remains free in the water column. In general, DIN and PO_4 concentrations are much lower in northern Gulf river-dominated estuaries (e.g., Mobile and Apalachicola Bays) than similar U.S. East Coast systems, presumably because of the lower point sources. Local groundwater sources are important and water quality managers should be aware of them. However, local estuarine point sources of N and P may alter nonpoint source patterns (e.g., paper mills and wastewater treatment facilities) (Livingston 2001a).

Denitrification is the third aspect that plays a role in N limitation in estuaries and coastal waters (Nixon et al. 1996). Denitrification is the process whereby nitrate is converted to gaseous N_2 and N_2O and thereby made unavailable. Denitrification provides a sink for N in estuarine systems. Shelf waters generally are too deep to provide enough sediment-water column contact for a quantitatively significant magnitude of denitrification to occur. Bottom-water anoxia limits nitrification and hence denitrification in the high-sulfide sediments where nitrification and denitrification are coupled (Jenkins and Kemp 1984). Knowledge of the magnitude of denitrification can help the water quality manager predict the nutrient overenrichment response of an estuary, because N that is denitrified is largely unavailable to support primary production.

Biological Processes

The relative importance of biological grazing should be assessed, because when a nutrient problem occurs it is evidence that enrichment has exceeded the ability of the system to maintain a steady state in net biomass production at pre-enrichment levels. For example, major changes in the biology of estuaries in terms of particle filtering capacity (e.g., oysters: Newell 1988) and probably filter feeding finfish (e.g., menhaden) can modify phytoplankton primary production, although the quantitative effectiveness of such cropping is scientifically unsettled. Nutrient overenrichment may drive marine waters toward smaller algae and other microbes (Jonas 1992) where organic carbon flows more to the microbial loop (Hassett et al. 1997), versus more direct flow to copepods and higher trophic levels (e.g., finfish). This shift in the food web may be a significant factor in how estuaries "assimilate" increased nutrient inputs (Roelke et al. 1999; Roelke 2000).

Physical Processes and Factors

Conceptual Framework

Smith (1984) argued that there is no inherent difference in nutrient limitation between lakes and the ocean. Abundant evidence indicates that phytoplankton net primary production in north temperate lakes tends to be P limited, and phytoplankton production in the ocean as a whole is potentially moderately P limited but at higher P concentrations than lakes. This conclusion is supported by the observation that TN:TP ratios of the surface ocean are usually well in excess of the Redfield ratio (Guildford and Hecky 2000). Local deviations have been detected. In contrast to lakes and oceans, estuaries and coastal shelf waters tend to be N limited, with some exceptions. Water quality managers may question the reason for this, as the three case studies described earlier, especially temperate estuaries and the coastal shelf, appear to be N limited and not P limited. Such an understanding is basic to arguments about cause and effect and also what ecosystem conditions drive a coastal ecosystem toward N limitation or P limitation. Smith provided an explanation that still has merit.

Smith posited that the apparent difference in limiting nutrient between lakes and oceans lies, in part, with the relative rates for material exchange via physical processes of advection (i.e., transport of water and associated constituents) and eddy diffusion (i.e., local transport of material against a concentration gradient) and biogeochemical processes of N fixation and fixed N loss. The argument is based on field experiments in marine embayments with little or no freshwater input, so advective transport of nutrients from the land simplified nutrient budget development. Smith postulates that if the physical exchange rates are long (e.g., open ocean), then the system would tend toward P limitation because biogeochemical adjustment of the N:P availability ratio is short compared with long physical exchange rates (e.g., months to a year or longer). In other words, nitrogen fixation would balance any losses of nitrogen associated with phytoplankton sedimentation, but P has no atmospheric reservoir or biochemical mechanism for an equivalent P fixation to occur. If the physical exchange rates are faster than biochemical rates (e.g., nitrogen fixation), then net ecosystem production (and by inference net phytoplankton biomass production) of organic material may be N limited; if the biochemical rates are faster, then net production will tend toward P limitation. The ratio of the residence time of the water to the biogeochemical turnover rate indicates the degree to which the hydrodynamic processes dominate or modify estuarine ecosystems (Day et al. 1989). This is an example of the importance of scaling critical processes (Harris 1986). Smith's conceptual model should apply to estuaries and shelf waters. Smith's data, analysis, and synthesis and other empirical data support N limitation in estuaries and coastal shelf environments.

The concept of scale is another element of the conceptual framework. Physical processes that modify the expression of bloom dynamics will best be detected at the ecological level (Figures A-1, A-2). In this context, Geyer et al. (2000) cite many examples supporting the observation that "at virtually every spatial scale, within every component of estuarine ecosystems, physical processes influence the distribution and fate of chemicals (*sic* including nutrients) and organisms." Physical processes are involved in the delivery of nutrients to the biota in estuaries and coastal shelf waters, and also fundamentally influence advective and dispersive processes that transport and retain dissolved and particulate material, including nutrients and plankton in estuaries and on the coastal shelves. The roles of physical processes influencing net biomass production of phytoplankton are explored in the main text in more detail.

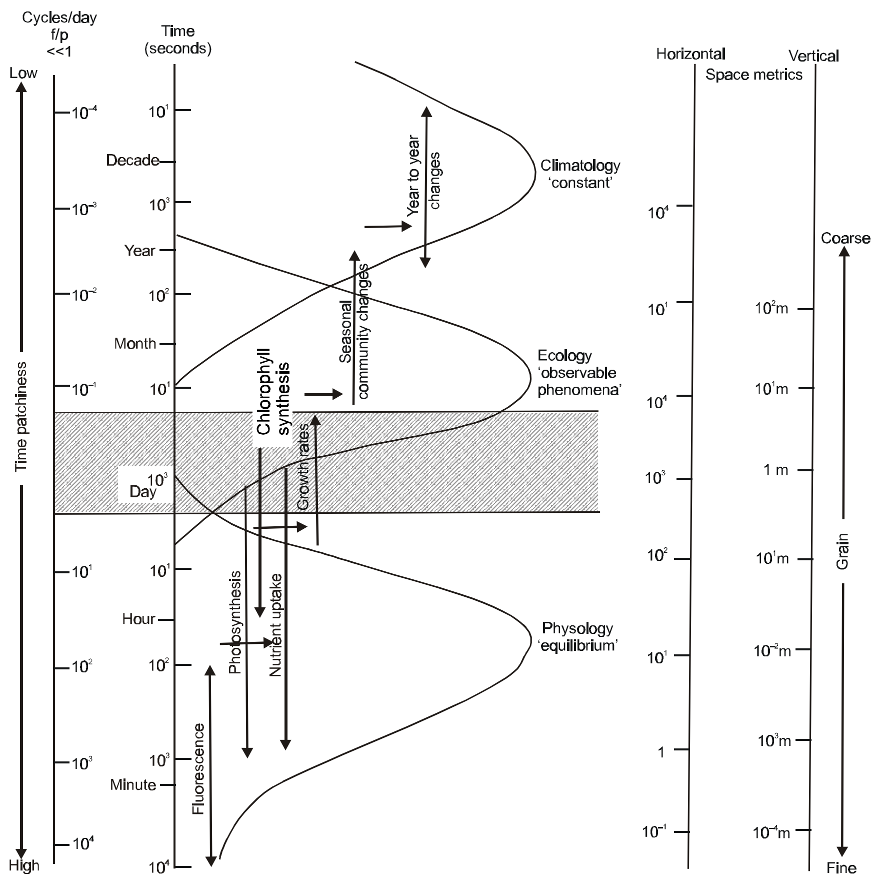
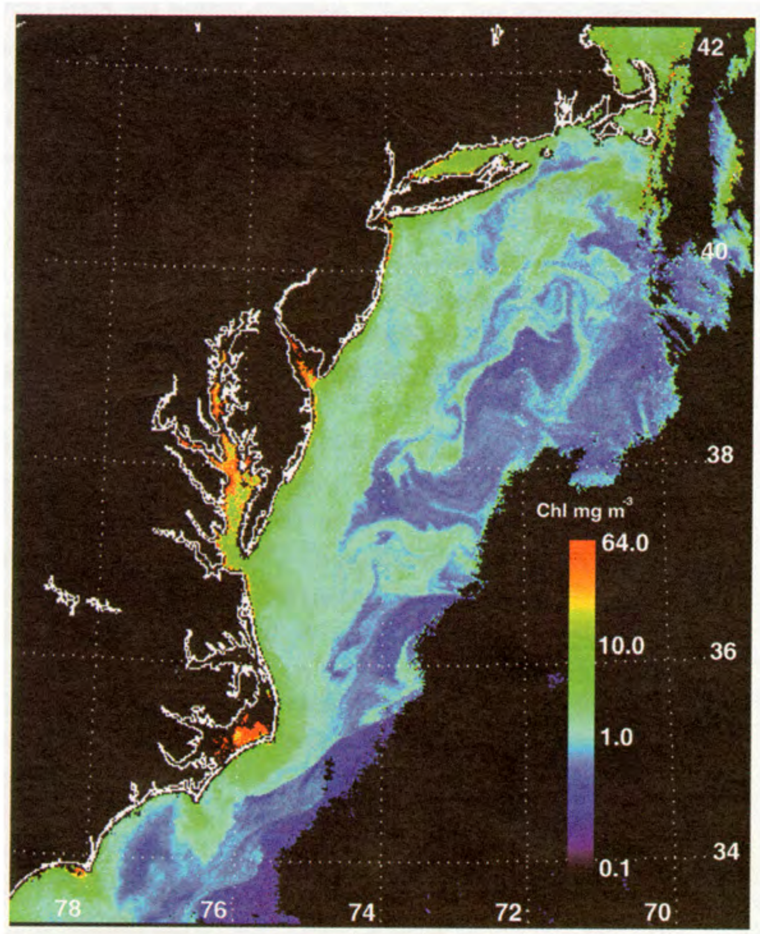


Figure A-1. Scales of phytoplankton ecology. Horizontal and vertical scales are determined by the respective diffusion coefficients K_h and K_L. The time scales for the algae are determined by the scales of growth (shaded band). The processes of importance at each scale are noted. In the past it was always assumed (wrongly) that physiological processes were at equilibrium and that climatological variability could be ignored. Source: Harris 1986.



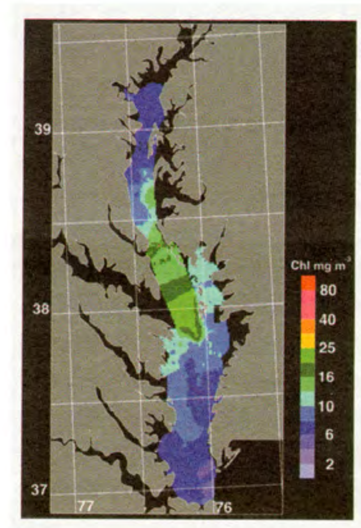


Figure A-2. The left panel shows the distribution of chlorophyll—an indicator of algal biomass—along the east coast of the U.S. from Boston to South Carolina as measured from the ocean color satellite SeaWIFs. Note the higher chlorphyll levels closer to shore, and the much higher levels in enclosed bays, such as Pamlico Sound (latitude 35°) and Chesapeake Bay (mouth at 37° latitude). The above panel shows chlorophyll distributions within Chesapeake Bay in more detail, as measured during a phytoplankton bloom. Both images were taken in April 1998. Source: Howarth et al. 2000.

APPENDIX B
ADDITIONAL INFORMATION ON THE ROLE OF TEMPERATURE AND LIGHT ON ESTUARINE AND COASTAL MARINE PHYTOPLANKTON

Availability in temperature and light values can be plotted as hydroclimographs (e.g., polygons) to picture their relative seasonal change around the coasts. A temperature increase will lower the density of seawater and contribute to density stratification. In particular, rapid changes in water temperature influence the rate of biological metabolic processes, including algal growth rates (Eppley 1972). Some species exhibit various degrees of thermal adaptation when temperature changes are gradual.

Speculation suggests that if sea temperatures continue to rise as a function of the "greenhouse effect," estuarine biotic communities may change over the next several decades as they approach thermal limits. For example, although the seagrasses *Halodule* (shoalgrass) and *Zostera* (eelgrass) now overlap in Core Sound, NC, a northward migration of *Halodule* and a retreat of *Zostera* may occur if water temperatures rise faster than populations can adapt. Such relationships and their potential alteration probably can be documented for other biotic groups along other coasts. For example, if temperature now limits regular flowering of the seagrass *Thalassia* (turtle grass) along the northern Gulf of Mexico, then increased flowering may occur if temperatures warm. Such conjectures notwithstanding, however, little information is available to help assess the consequences of a potential interaction between an increased temperature rise and increased nutrient supply on seagrasses.

Light has a fundamental role in aquatic primary production and is essential in the development of models to estimate phytoplankton primary production (Behrenfeld and Falkowski 1997) and submerged aquatic vegetation (Dennison et al. 1993). Many concepts in aquatic ecology are based on the light gradient (Huisman 1999) (e.g., diel plankton vertical migration, benthic animals migrating out of sediments, depth of euphotic zone, and mixing depth). Phytoplankton growth, nutrient relationships, light, and other physical processes interact in a feedback system. Although light may be adequate and all other requirements met for their formation, blooms may not form if dispersive processes are greater than algal cell doubling time (Kierstead and Slobodkin 1953; Lucas et al. 1999). Tidal ranges greater than approximately 2.0 m apparently disperse phytoplankton faster than cell doubling time, even if nutrient conditions would be supportive of a bloom (Monbet 1992).

Both the vertical distribution of phytoplankton abundance and community composition are frequently changing in the water column. Swimming through use of a flagellum, especially by dinoflagellates, and changes in cell density through physiological mechanisms allow modest vertical mobility against weak mixing forces. If the mixing depth is substantially greater than the euphotic zone depth (e.g., a depth where approximately 1% of surface insolation occurs), then phytoplankton spend too much time in an inadequate light environment and net primary production is limited (Figure B-1) (Huisman et al. 1999). The compensation depth is where water column phytoplankton photosynthesis and respiration are in balance, and this often approximates the 1% insolation depth, or about two times the Secchi disc depth (Parsons and Takahashi 1973). For example, in the lower Delaware Bay the upper-mixed layer often corresponds to the bay bottom, the result of which is that phytoplankton spend too much time below the compensation depth and hence low biomass production occurs (Pennock 1985). In systems where the dominant support of the food web is derived from photosynthesis, net phytoplankton production must be large enough to support the microbial loop (Azam et al. 1983) and higher trophic levels.

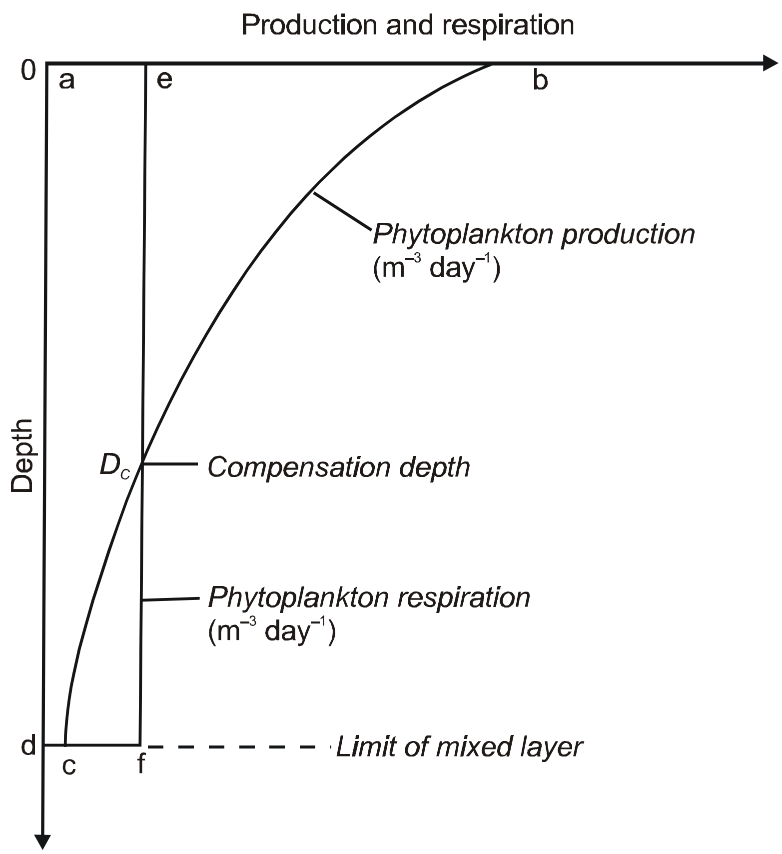


Figure B-1. Diagram illustrating theoretical distribution of phytoplankton production and phytoplankton respiration. After Sverdrup (1953). Source: Mann and Lazier (1996).

In a homogeneous medium, light decreases exponentially with depth and can be represented by the negative exponential equation:

$$I_z = I_o e^{-kz}$$

where I_z is light quantity at depth z, I_o is the light quantity at the water surface, and k represents the vertical light extinction coefficient; the extinction coefficient is more easily calculated as the base $\log_{10}$: $0.434\ kz = \log I_0 - \log I_z$). The light gradient also often extends longitudinally down estuaries, especially those dominated by large volumes of sediment-bearing freshwater. In highly turbid estuaries, deepest light penetration shifts toward the orange end of the spectrum (Champ et al. 1980). The euphotic zone depth generally increases from the tidal head to the coast. Where turbidity is at its maximum level, a localized sharp decrease in euphotic zone depth is typical (Flemer 1970; Pennock and Sharp 1986). Regions with the greatest turbidity typically are light-limited or almost so. In the turbid upper Chesapeake Bay, riverine loading supplied nearly 90% of the particulate organic carbon, but in the clearer waters of the mid-bay primary production dominated supply at 97% (Biggs and Flemer 1972); Smith and Kemp (1995) have updated their original estimate and suggest a lower percentage. However, much of the allochthonous organic matter may not be biologically available. In waters of high humic material content (e.g., Charlotte Harbor, FL), light attenuation can be severe (Dixon and Kirkpatrick 1999). The interaction between turbidity caused by humic materials and nonchlorophyll-bearing particulates complicates the direct application of the Secchi disc as a measure of nutrient overenrichment. Spectral radiometers can to some extent partition the various components of light extinction and are the preferred tools. Modern algal pigment diagnostic tools (e.g., HPLC) can compare water quality responses to varying nutrient and other pollutant inputs across various coastal system types (Jeffery et al. 1997). It is possible that turbidity may mask the impending development of undesirable algae.

The vertical extinction coefficient for estuaries shows wide seasonal variations. Values frequently exceed 0.1 m^{-1}). The extinction in open-ocean waters is often estimated by the relation E.C. = 1.7/ Secchi depth in meters. Holmes (1970) and Keefe et al. (1976) both arrived independently at a constant of approximately 1.46 instead of 1.7 for turbid estuarine and nearshore coastal waters. Walker (1980) suggested that the original Poole and Adkins (1929) Secchi disc constant on average gives results approximately 17% too high and suggested a value of 1.45. These corrections should be made and, more importantly, it is useful for the constant to be checked for each estuary. For more quantitative work, a quantum light meter that measures over a spectral range of 400-700 nM is preferred. Commercial products (e.g., www.licor.com) are now available that can measure the spectral photon flux over a range of interest to aquatic scientists.

The extinction coefficient can be broken down into several components. The total light attentuation, $K_T = K_w + K_c + K_d + K_p$ (Lorenzen 1972; Kirk 1983; Bledsoe and Phlips 2000; Koenings and Edmondson 1991) can be resolved for the effects of water, chlorophyll *a*, dissolved substances, and nonalgal particulate matter. In many estuaries, K_d may contribute between 5% and 50% of the K_T. The Kw usually can be ignored because it is such a minor component. In blackwater estuaries receiving high loads of humic materials the K_d may dominant K_T. In systems such as the "turbidity maximum zone" in upper Chesapeake Bay, K_p may be the dominant component. The EPA Chesapeake Bay Program has sponsored research to calibrate K_T components applicable to SAV beds (www.chesapeakebay.net; search the publications database for "Chesapeake Bay Submerged Aquatic Water Quality and Habitat-Based Requirements and Restorations Goals: A Second Technical Synthesis."

Among some coastal ecosystems, light (i.e., mean photic depth) and nutrient loading appear to be equally good predictors of phytoplankton primary production (see Figure 1b in Cloern 1999). This observation strengthens the proposition that phytoplankton production in these systems can be limited by other resources and processes in addition to nutrient loading. Pennock and Sharp (1994) suggest that the Delaware River Estuary functions analogously as a chemostat during the summer. They point out that high N supplies from upstream advect continually through the brackish-water region into the lower estuary, where high primary production occurs from remineralized N and the advected supply and the phytoplankton biomass is limited by grazing. Their conclusion is especially significant because evidence also suggests that bioassay experiments that isolate the water may lead to misidentification of nutrient limitation. The flushing component in the bay provides the physical analogue to a chemostat, where the nutrient supply and adequate light support high phytoplankton biomass production and grazing and flushing maintain a potentially steady-state phytoplankton biomass, with fluctuations due primarily to physical forcing factors.

APPENDIX C
ADDITIONAL INFORMATION ON FLUSHING IN ESTUARIES

by Edward H. Dettmann, U.S. EPA, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division

A variety of terms such as residence time, flushing time, transit time, turnover time, and age are used to describe time scales for transport and removal of materials that enter waterbodies. Use of these terms in the literature is often inconsistent and sometimes imprecise, so that care must be exercised to determine the meaning of terms being used, to avoid misinterpretation or incorrect comparisons of data. This appendix contains definitions of terms and discussions of factors that affect flushing and empirical and modeling methods for estimating residence times in estuaries. An understanding of residence times is especially important when estimating not only system responses to nutrients, but also the lag phase between management and system improvements.

Definitions of Residence Times

The freshwater residence time (τ_{fw}) is the average amount of time that freshwater, or a conservative tracer introduced with freshwater inputs, resides in the estuary before exiting. It is the mean transit time for a molecule of water or conservative tracer to progress from the point of input to the seaward boundary. This is the definition of residence time most often given in the literature, and is generally the most useful for analysis of eutrophication in estuaries, as most nutrients are usually introduced with freshwater.

Another commonly used concept is that of the mean amount of time required for water (or a homogeneously-distributed conservative tracer) that is in the estuary at a given time (regardless of source) to leave the estuary. This is here called "estuary residence time"(τ_e), although Takeoka (1984) and Zimmerman (1976) use the term "residence time."

The values of τ_{fw} and τ_e may differ. Consider the case of an elongated unstratified estuary, with the seaward boundary at one end and a river as the single source of freshwater located at the opposite end. In this case, the travel distance from the river mouth to the seaward boundary is longer than the average travel distance from other parts of the estuary, so that $\tau_{fw} > \tau_e$. Conversely, if the river enters the estuary near the seaward boundary, τ_e may exceed τ_{fw}. For cases intermediate between these two, the difference between these two residence times becomes smaller, and the times may be equal. Multiple freshwater inputs and other factors such as stratification and estuary shape may complicate matters, but the same principles apply.

These concepts, the related concepts of turnover time and age, and the relationships among these measures are discussed in detail by Bolin and Rodhe (1973) and Zimmerman (1976), and summarized by Takeoka (1984). Miller and McPherson (1991) discuss another concept, "pulse residence time," for water or a conservative constituent introduced as a pulse, that may be useful in some circumstances.

Factors That Influence Residence Time

Water residence times in estuaries are influenced by any factor that affects water movement, including freshwater inflow rates, tides, wind, mixing, stratification, and system topography. Because many of these factors are variable, residence times are also not static. This variability requires attention to the appropriate time interval over which the residence time should be expressed, and the representativeness of the conditions under which a given measurement is made. A long-term (seasonal or annual) average residence time is often most appropriate for analysis of the effects of nutrients.

Freshwater Forcing

Residence time may be quite sensitive to freshwater inflow rate, with larger flow rates associated with smaller residence times. This is illustrated by three Rhode Island estuaries. For Narragansett Bay, τ_{fw} varies between approximately 10 and 40 days; τ_{fw} at the long-term mean freshwater inflow rate is 26 days (Pilson 1985). For the Seekonk River, τ_{fw} varies between 0.4 and 2.9 days, with a value of 1.2 days for the mean freshwater inflow rate, and for the Providence River, τ_{fw} varies between 0.7 and 6.3 days, and is 2.5 days at average inflow (Asselin and Spaulding 1993).

Tidal Forcing

Tides can be a major factor controlling estuary-ocean exchange of water and therefore water residence time. Important factors are the tidal range, tidal frequency (diurnal vs. semidiurnal), and estuary depth. Tidal ranges in U.S. coastal waters range from centimeters to more than 5.5 m. For a given estuary, residence times may vary over the spring-neap tide cycle.

Wind Forcing

Wind may substantially influence estuarine circulation, and therefore water residence time. In a study of two small shallow estuaries, Geyer (1997) found that wind direction (offshore vs. onshore) had a substantial effect on salinity structure and nontidal flow in one estuary, and τ_{fw} varied by a factor of approximately 3 (from 0.8 to 2.7 days) in response to the sea-breeze cycle. Measurements in a nearby and similarly oriented estuary during the same time period found that whereas wind direction influenced salinity distributions, a constriction at the estuary mouth limited estuary-offshore exchange, so that there was no significant relationship between residence time and wind stress. Estuaries bordering the Gulf of Mexico are shallow, highly susceptible to wind forcing, and considered meteorologically dominated (Solis and Powell 1999; Ward 1980).

Determination of Residence Time

A number of empirical and computational methods are used to measure or estimate water residence times.

Empirical Measurements

Empirical measurements of water residence time depend on measurement of the distribution or dynamics of tracers (generally freshwater or introduced dye) in estuaries.

Bolin and Rodhe (1973) show that the transit time of a tracer through a reservoir is given by the turnover time (τ_o) , i.e.

$$\tau_o = \frac{M_o}{F_o}$$

where M_o is the total mass of a constituent in the reservoir and F_o is the total flux through the reservoir. This is the basis for the freshwater replacement method for calculating the mean freshwater residence time in an estuary. The τ_{fw} is calculated as

$$\tau_{fw} = \frac{V_{fw}}{Q_{fw}}$$

where V_{fw} is the volume of freshwater in the estuary and Q_{fw} is the input rate of freshwater. This ratio gives the time required for the inflowing freshwater to replace the freshwater already in the estuary. The volume of freshwater in the estuary is calculated as the amount of freshwater that must be mixed with seawater having salinity (S_s) equal to that entering at the seaward boundary to yield a volume V_e equal to that of the estuary, with salinity equal to the mean salinity of the estuary (S_e). V_{fw} may be calculated as

$$V_{fw} = \left(1 - \frac{S_e}{S_s}\right) V_e$$

Examples of this method are given by Pilson (1985), Geyer (1997), and Solis and Powell (1999).

Fluorescent dye (usually Rhodamine WT) is also used to determine water residence times. To measure freshwater residence time, dye is introduced continuously into the inflowing freshwater at a rate proportional to the freshwater flow rate. Dye concentrations are surveyed at (high or low) slack tide. Dye input is terminated when the mean concentration of dye in the estuary reaches equilibrium (approximately 3 times the freshwater residence time). Further periodic surveys are conducted at the same tide phase to monitor the dye content of the estuary. The average concentration of dye is usually found to follow a decreasing exponential with time,

$$C(t) = C_o \, e^{-k\,t}$$

where C_o is the initial average concentration of dye in the estuary and C(t) is the concentration at time t. This function is fit to the data, and the mean residence time of dye in the reservoir, a surrogate for the mean freshwater residence time, is then $\tau_{fw} = 1/k$. Alternatively, as the concentration one residence time after termination of dye input is $C = C_o\, e^{-1}$, the time required to attain this concentration is sometimes taken as the residence time. The estuary residence time may be measured similarly. In this case, the dye is distributed as uniformly as possible throughout the estuary in a rapid application, and the change in dye content is monitored as above. The residence time determined from the rate of change of tracer concentration is sometimes called the e-folding time. Because of cost and logistical considerations, dye studies are most often done on relatively small estuaries. Examples of such studies are described by Callaway (1981), Dettmann et al. (1989), and Geyer et al. (1997).

Models

A wide range of models, ranging from simple to complex, has been used to calculate water residence time in estuaries. The simplest of these is the tidal prism model, which estimates residence time (in number of tidal cycles) as the ratio V/P, where V is the estuary volume (usually expressed at high tide) and P is the volume of the tidal prism. The model is based on the assumption that water entering the estuary on the flood tide is thoroughly mixed throughout the estuary within a tidal cycle. Most estuaries do not meet this requirement and, for all but very small estuaries, this model generally underestimates estuary and freshwater residence time, sometimes by manyfold.

Ketchum (1951) modified this simple model by dividing the estuary into segments, each having a length that corresponds to the local tidal excursion. Water exchanges between adjacent segments during each tidal cycle are calculated, and complete mixing is assumed to occur only within each segment. The segmented tidal prism model requires data for freshwater inputs, tidal range, and estuary topography. The simple and segmented versions of the tidal prism model are reviewed by Pritchard (1952) and Dyer (1973), and modified by Dyer and Taylor (1973). For most (but not all) estuaries to which the segmented model has been applied, it has given good results (Dyer 1973). Beyond permitting calculation of residence time, segmented tidal prism models permit calculation of the distributions of fresh- and saltwater and other water quality constituents along the estuary, but do not address the effects of wind on flushing.

Another approach to calculating residence times is the box model. Box models also segment the estuary, assume complete mixing within segments, and calculate diffusive and advective exchanges between adjacent segments. These models require data for estuary topography and for freshwater inflows and salinity distribution in the estuary for each set of conditions to which the model is applied. As actual salinity data are used, the effects of wind and tide are implicitly taken into account. Applications of such models are described by Brown and Arellano (1980), Dettmann et al. (1992), Hagy et al. (2000), and Miller and McPherson (1991). Such models also allow calculation of advective and diffusive exchanges among segments and spatial distribution of water quality constituents.

Numerical computer models of hydrodynamics and constituent transport are still more complex. These models are used to simulate movement of water and water quality constituents at fine spatial resolution, and may also be used to calculate freshwater or estuary residence times. These models generally allow consideration of wind stress as well as tidal and freshwater forcing. A few examples of such models are described by Brooks et al. (1999), Signell (1992), and Signell and Butman (1992).

APPENDIX D
NOAA SCHEME FOR DETERMINING ESTUARINE SUSCEPTIBILITY
(SOURCE: BRICKER ET AL. 1999)

The following provides an overview of the NOAA scheme for determining the overall human potential for causing nutrient enrichment of estuaries.

Dilution and Flushing Potential
The length of time that nutrients spend in an estuary potentially affects their opportunity to contribute to overenrichment. The time is a function of dilution potential and flushing rate. The analysis uses physical and hydrologic data to define separately (1) a dilution rating and (2) a flushing rating. In both cases, the higher rating, the greater the capacity to dilute or flush nutrient loads (see Tables D-1 and D-2).

Figure D-1 combines dilution potential and flushing potential. By combining dilution and flushing components, an export potential (EXP) is determined. Estuaries in the upper left portion of the matrix generally have a high EXP that suggests an ability to dilute and flush nutrient loads. Estuaries in the lower right portion of the matrix have the opposite capacity, making them more susceptible to nutrient input.

Nutrient Inputs
NOAA used the USGS Sparrow Model (spatially referenced regressions of contaminant transport on watershed attributes) and other information to estimate nutrient loads and measure nitrogen pressure on estuaries.

Determination of Overall Human-based Nutrient Pressure
A matrix was used to compare the susceptibility to nutrient retention and the level of N inputs to rank the overall expression of human influence on eutrophic conditions in the estuary (Figure D-1). Experts at the National Assessment Workshop reviewed and, where appropriate, modified the assessments based on higher quality data available for some estuaries; expert knowledge also played a role.

Table D-1. Decision rules for dilution potential

Type	IF: vertical stratification	THEN: dilution volume	IF: dilution value	Dilution potential	No. estuaries
A	Vertically homogenous •all year •throughout estuary	$1/VOL_{estuary}$	10^{-13} 10^{-12}	High	30
B	Minor vertical stratification •navigation channels •upper estuary	$1/VOL_{estuary}$	10^{-11}	Moderate	63
C	Vertically stratified •most of year •most of estuary	$1/VOL_{fwf}$ (fwf=freshwater fraction)	10^{-10} 10^{-09}	Low	45

Note: This analysis assumes that a larger portion of the water column is potentially available to dilute nutrient loads in a vertically homogeneous estuary than in a vertically stratified system. The assumption is that for stratified systems, nutrients are most often retained in the upper portion (freshwater fraction) of the water column. In contrast, downward transport (more complete mixing) is likely in vertically homogeneous systems. Type B estuaries are generally vertically homogeneous, although stratification is observed (confined) in narrow navigation channels or the extreme upper reaches of an estuary. In this case, nutrients are assumed to be diluted throughout the entire water column.

Source: Bricker et al. 1999.

Table D-2. Decision rules for flushing potential

Type	Tide range (ft)		Freshwater inflow/estuary volume	Flushing potential	No. estuaries
1	Macro (>6)	and	Large or moderate (10^{00} to 10^{-02})	High	12
2	Macro (>6)	and	Small (10^{-03}, 10^{-04})	Moderate	21
3	Meso (>2.5)	and	Large (10^{00}, 10^{-01})	High	15
4	Meso (>2.5)	and	Moderate (10^{-02})	Moderate	16
5	Meso (>2.5)	and	Small (10^{-03}, 10^{-04})	Low	26
6	Micro (<2.5)	and	Large (10^{00}, 10^{-01})	High	4
7	Micro (<2.5)	and	Moderate (10^{-02})	Moderate	13
8	Micro (<2.5)	and	Small (10^{-03}, 10^{-04})	Low	31

Note: This analysis assumes that a greater capacity to flush nutrient loads exists for estuaries that have large tide and freshwater influences.

Source: Bricker et al. 1999.

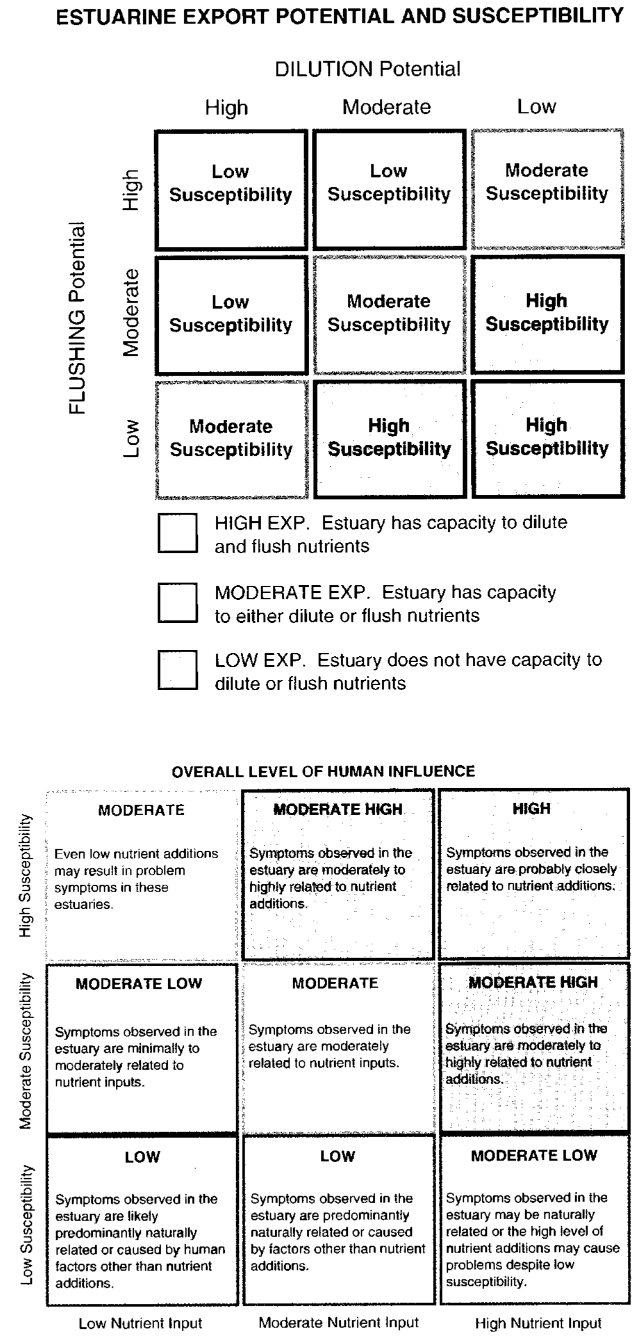


Figure D-1. Estuarine Export Potential Susceptibility.

APPENDIX E
COMPARATIVE SYSTEMS EMPIRICAL MODELING APPROACH: THE EMPIRICAL REGRESSION METHOD TO DETERMINE NUTRIENT LOAD-ECOLOGICAL RESPONSE RELATIONSHIPS FOR ESTUARINE AND COASTAL WATERS

By James Lattimer, USEPA, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division

The empirical regression method can be used to determine the response of estuarine systems to nutrient loading. This method can take two forms: single system and comparative-systems approaches.

In single system approaches, ecological responses and nutrient concentrations are measured over time in a single system. This allows for the development of load-response models for the individual systems that are robust because many of the controlling factors, such as, for example, physiographic setting, primary productivity base, and hypsography, are relatively constant. This approach has been used to develop models of the response of primary productivity to nitrogen load (Figures E-1a,b) (Boynton et al. 1995). However, this approach is only applicable to the system in which it was developed and thus is not considered to be very useful in the development of widely applicable nutrient load-response relationships. Single system empirical studies, however, may be useful in providing data on processes useful for numerical model development.

The alternative approach, using the space for time paradigm (Picket,1988), posits that relationships between nutrient inputs and ecologically meaningful estuarine responses, using multiple systems, have predictive capability, at least for the category of systems used in the model development. This allows for a wide range in nutrient loading and estuarine types to be included. The comparative-systems empirical approach has been used to determine, for example, relationships between nutrient inputs and fish yields (Lee and Jones 1981; Nixon 1992), benthic biomass, production and abundances (Josefson and Rasmussen 2000), summer ammonia flux (Boynton *et al.*, 1995), chlorophyll-a concentration (Boynton et al. 1996; Boynton and Kemp 2000; Monbet 1992), primary productivity (Nixon *et al.*, 1996), and the dominant source of primary productivity (Nixon et al. in press). In many of these cases, important environmental factors such as flushing time and depth, are used to normalize the nutrient loading in a similar way as Vollenweider (Vollenweider 1976) to yield more precise relationships.

The comparative-systems empirical approach has been successfully used in a regulatory framework to develop total maximum loads for 30 subestuaries within Buzzards Bay in Massachusetts (Costa et al. 1999). Using a citizens' monitoring network, many important water quality variables were measured during summer sampling periods over a five to seven year period. Nitrogen inputs were estimated from a land-use model modified from Valiela (Valiela et al., 1997) augmented with literature data on point source and atmospheric inputs. In this study, nitrogen load-response relationships were derived for nitrogen concentration, chlorophyll-a, secchi depth, dissolved oxygen concentration, and eelgrass habitat ratio. Nitrogen loadings were normalized to account for volume and flushing time of each of the systems to improve the precision of the empirical models. An example of the types of relationships determined using this approach are given in Figure E-2. The regional estuarine management program used this method to adopt total maximum annual loads (TMALs) for nitrogen. Specifically, "...the nitrogen management strategy represents a linking of estimates of watershed nitrogen loading...to empirical observations of ecosystem response in a wide variety of Buzzards Bay embayments" (Costa 2000).

The comparative-systems empirical approach does not explicitly consider the processes that produce the observed phenomena; however, factors known to affect behavior are determined in order to reduce the uncertainty in the models. The smaller the variance in the load-response relationship, the more compelling the association. One important question for management is to determine what level of variance is sufficient to convince stakeholders to accept nutrient limits and associated monetary expenditures.

This approach is largely based on statistical associations and is therefore restricted to prediction within the class of systems used in the model development. Applicability improves, however, by the inclusion of systems that encompass a wide range in loading and responses. The comparative-systems empirical approach allows for the direct measurement of important endpoints (e.g., hypoxia, SAV loss, biomass) obtained in the environment. The endpoints are what are important to the general public. So by providing a mathematical relationships between the stressor and important endpoints, managers can convince the public of the importance of regulatory action.

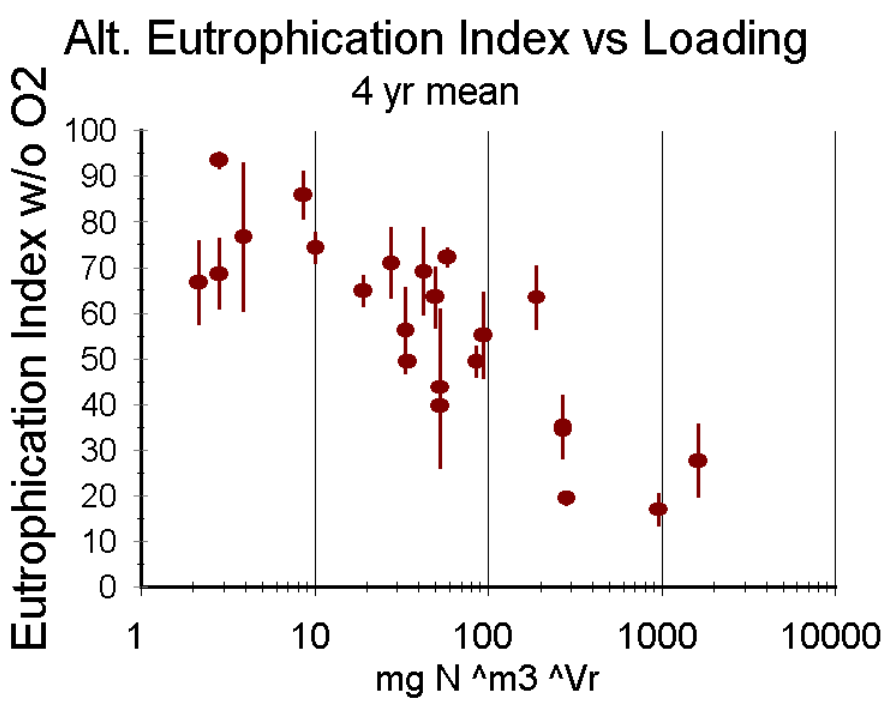


Figure E-1a. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and 92-98 mean +/- std. errors of the Alternate Eutrophication Index scoring (without oxygen scores).

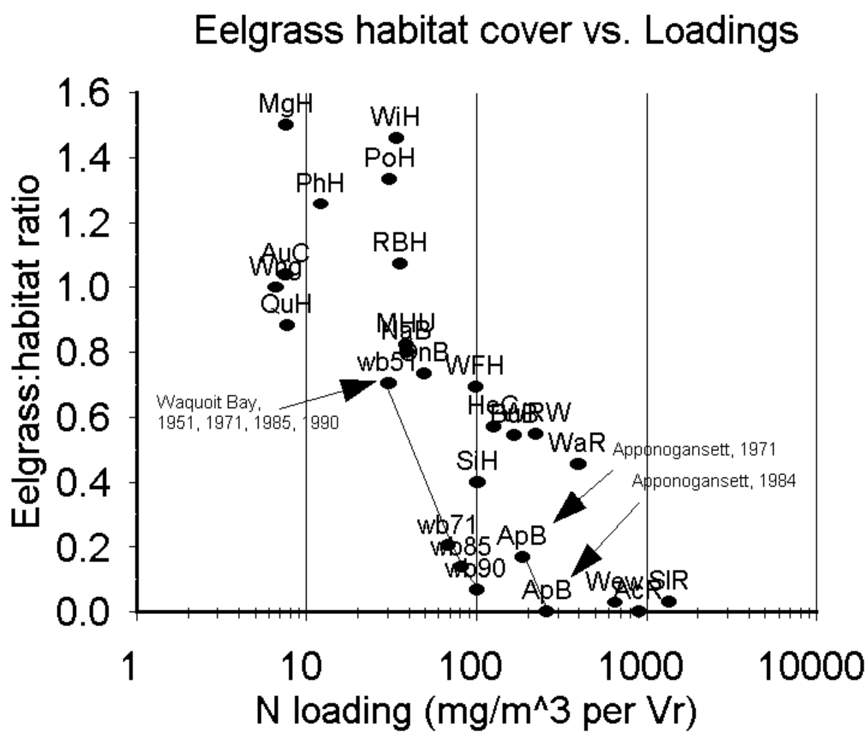


Figure E-1b. Ratio of eelgrass habitat area to potential habitat area versus nitrogen loading, expressed using the volume Vollenweider-term flushing scale (from Costa et al., 1999).

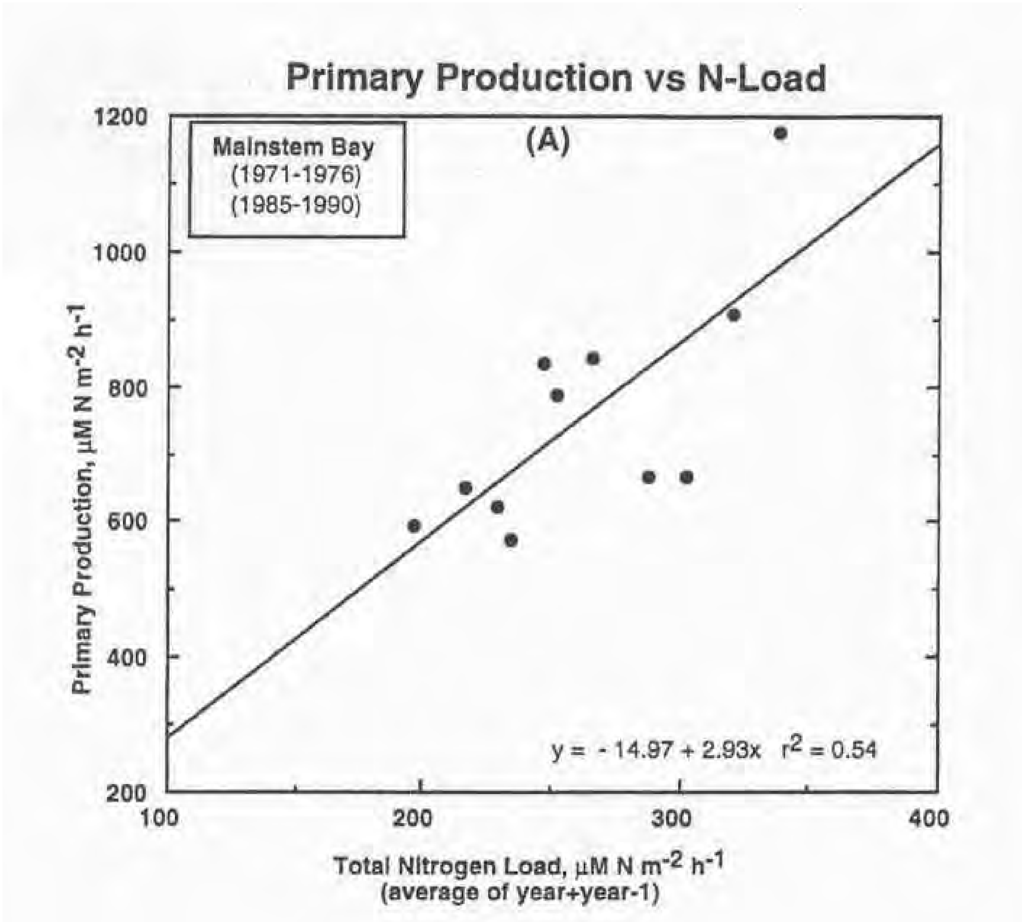


Figure E-2. Plot of annual TN loading rates versus phytoplankton primary production rates at a single station in the Chesapeake Bay from 1971-76 and 1985-90. Source: Boynton et al., 1995.

APPENDIX F
SELECTED THEORETICAL APPROACHES TO CLASSIFICATION OF ESTUARIES AND COASTAL WATERS

Several theoretical schemes have been presented that may facilitate classification of estuaries and coastal waters if the need arises for more theoretical approaches. In most cases, these schemes have potential future value but are not likely to be immediately useful. The information-thermodynamic approach offered by Ulanowicz would seem to provide useful insights to development of an eutrophication index and may be worth an earlier consideration.

Functional Attributes

Odum and Copeland (1974) proposed a classification scheme based on the idea that an ecosystem is a balance between energies that build structure and order, or *ordering energies*, and energies that cause loss of structure and order, or *disordering energies*. Although this approach is of theoretical interest, the data to apply it are still largely unavailable. There are conceptual difficulties. Energy sources and stresses are not always mutually exclusive categories. Day et al. (1989) give several examples of cases where moving water can be either an energy source or a stress, depending on the situation. Moderate currents are a source of energy for seagrass meadows, because they transport organic matter and inorganic nutrients to beds and remove metabolic wastes. If currents are too strong, however, they can disrupt beds and act as disordering energies.

Theoretic Indices

This approach is data intensive and has potential as appropriate estuary data are generated. Ulanowicz (1986, 1997) described an approach based on flow analysis and information theory. The idea is that an ecosystem can be characterized in terms of growth and organization. Growth is defined as an increase in system activity or total system throughput (analogous to total system energy flow). Organization is equated with the mutual information inherent in the trophic flow structure. To apply this approach, one would need to obtain energy flow values (or organic carbon-based equivalents) among trophic compartments. Following Ulanowicz's formulation, one could develop an information index of eutrophication. Compiling such would entail an assessment of the growth and organization status of a eutrophic or nutrient-enriched system compared with a current reference system, a minimally impaired system, or an estimate of pre-eutrophication values from historical data. The ratio of growth and organization of a nutrient-enriched system to its reference value would express the degree of impairment. Such information could help guide restoration priorities. The downside is that the approach requires the availability of a rich ecological database.

APPENDIX G

EXAMPLES OF NUTRIENT CONCENTRATION RANGES AND RELATED HYDROGRAPHIC DATA FOR SELECTED ESTUARIES AND COASTAL WATERS IN THE CONTIGUOUS STATES OF THE UNITED STATES

Atlantic Coast Systems

BOSTON HARBOR, MA (Source: Kelly 1998)
WAQUOIT BAY, MA (Source: Valiela et al. 1992)
SEEKONK-PROVIDENCE RIVER REGION OF NARRAGANSETT BAY, RI
(Source: Doering et al. 1990)
WESTERN LONG ISLAND SOUND AND HUDSON-RARITAN ESTUARY, NY/NJ
(Source: O'Shea and Brosnan 2000)
HUDSON RIVER, NY (Source: Lampman et al. 1999)
DELAWARE RIVER ESTUARY (Source: Lebo and Sharp 1993)
MARYLAND COASTAL BAYS (Source: Boynton et al. 1996)
CHESAPEAKE BAY, MD/VA (Source: Magnien et al. 1992)
CHESAPEAKE BAY (Harding and Perry 1997)
YORK RIVER ESTUARY, CHESAPEAKE BAY, VA (Source: Sin et al. 1999)
NEUSE RIVER ESTUARY, NC (Source: Rudek et al. 1991)
CAPE FEAR RIVER ESTUARY, NC (Source: Mallin et al. 1999)
COASTAL GEORGIA (Source: Hopkinson and Wetzel 1982)

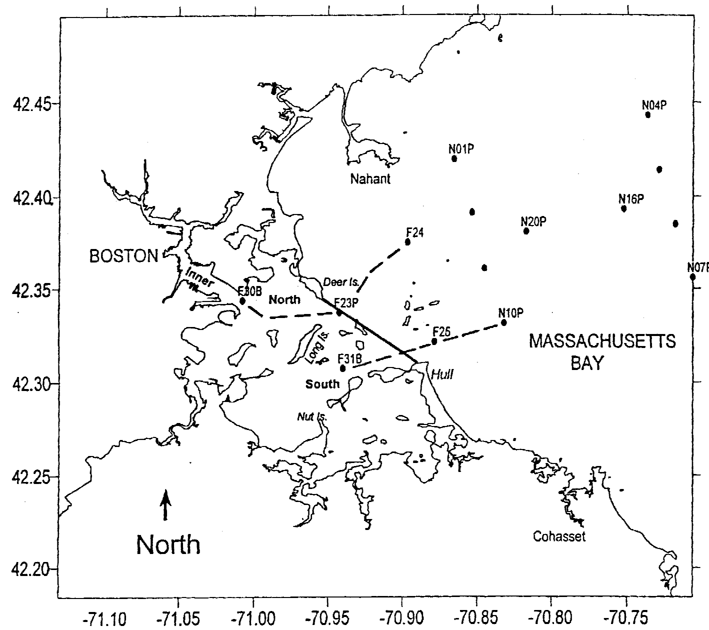
Gulf of Mexico Estuaries

GULF OF MEXICO CASE STUDIES (Source: Bianchi et al. 1999)
GALVESTON BAY, TX (Source: Santschi 1995)

Pacific Coast Systems

EMBAYMENT OF PUGET SOUND, WA (Source: Bernhard and Peele 1997)

Boston Harbor



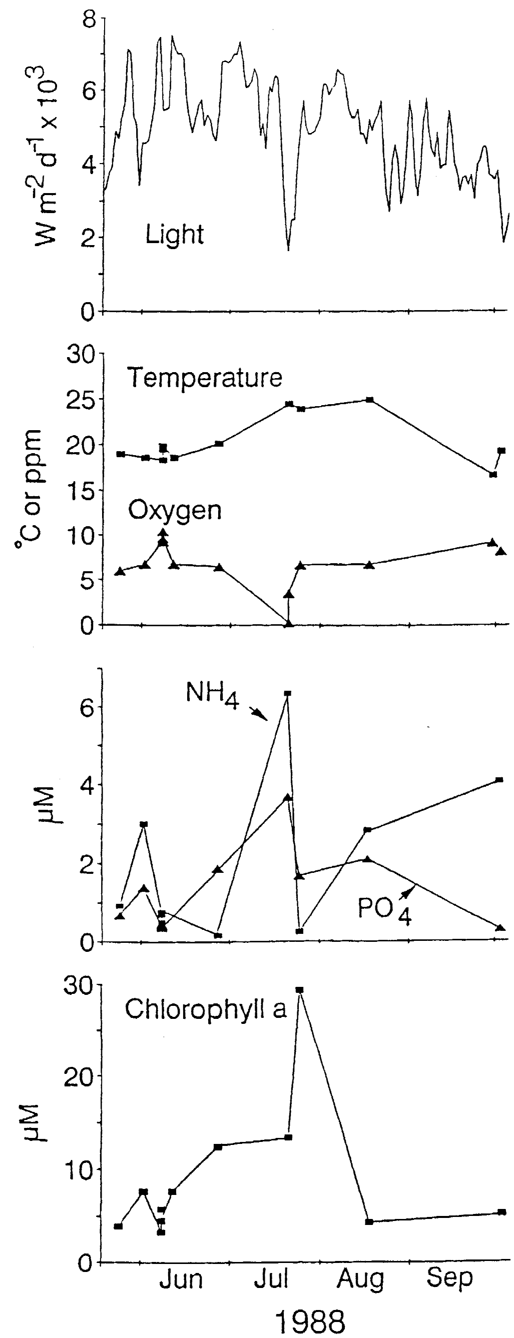
The study area (N latitude and W longitude) in Boston Harbor and western Massachusetts Bay. The boundary between the Harbor and the Bay is defined by the solid line from Deer Island to Hull combined with the dashed lines showing the 2 high-resolution transects, the spatial limits of data for box modeling are depicted. Dots position some watercolumn monitoring stations that were sampled during 1994. Stations prefixed with 'N' surround the future offshore outfall, which is centered between Stns N20P and N16P about 15 km from Deer Island. Data from 2 lines of Bay staions (N01P to N10P and N04P to N07P, each representing about 10 km distance) were used to provide approximate concentrations for the future tidal source region for the Harbor.

Survey	Transect	Bay C_o (to calculate ocean loading) BA[a] (m^{-1})	NH$_4$ (µM)	DIN (µM)	TN (µM)	PO$_4$ (µM)	SiO$_4$ (µM)	Harbor C_i (to calculate Harbor output) BA[a] (m^{-1})	NH$_4$ (µM)	DIN (µM)	TN (µM)	PO$_4$ (µM)	SiO$_4$ (µM)
W9402	North	2.23	3	10.5	20	0.8	8	2.34	4	14	28.5	0.8	12
	South	1.89	2	9.5	18	0.8	8	1.78	3.5	12.5	19	1	9.5
W9403	North	1.23	1	2	nd	0.3	1	1.61	nd	nd	nd	nd	nd
	South	1.07	0.75	1.75	nd	0.3	0.75	1.32	nd	nd	nd	nd	nd
W9404	North	0.85	2	3	16	0.3	1.5	1.29	3	4.5	25	0.3	3
	South	1.11	1	2	14	0.25	1	1.30	2.5	3.25	18	0.25	1.4
W9405	North	1.53	1.5	3	nd	0.5	2.75	1.84	nd	nd	13	0.5	nd
	South	1.46	1.5	3	nd	0.5	2.75	2.00	nd	nd	11	0.8	nd
W9407	North	1.55	1.5	2	14	0.7	1.5	2.15	2.25	2.6	17.5	0.6	1.5
	South	nd	nd	nd	nd	nd	nd	nd	2	3	14.5	0.8	2.5
W9409	North	2.04	5	6	nd	1	4.5	3.26	2.1	nd	11	0.7	nd
	South	1.50	5	6	nd	1	4.5	1.64	1.4	nd	10.5	0.55	nd
W9411	North	1.51	7	11	21	1.1	6	2.07	10	15	26	1.3	8
	South	1.78	4	7	16	1	5	1.90	11	16	24	1.3	7
W9412	North	2.13	6	8	nd	1	3	2.66	2.25	nd	20	1.5	nd
	South	nd	6	8	nd	1	3	nd	nd	nd	2.1	1.5	nd
W9413	North	1.94	2	3.5	nd	0.5	2.75	2.06	15	20.5	29.7	1.7	nd
	South	1.86	2	3.5	nd	0.5	2.75	2.06	18	20.5	29.7	1.7	nd

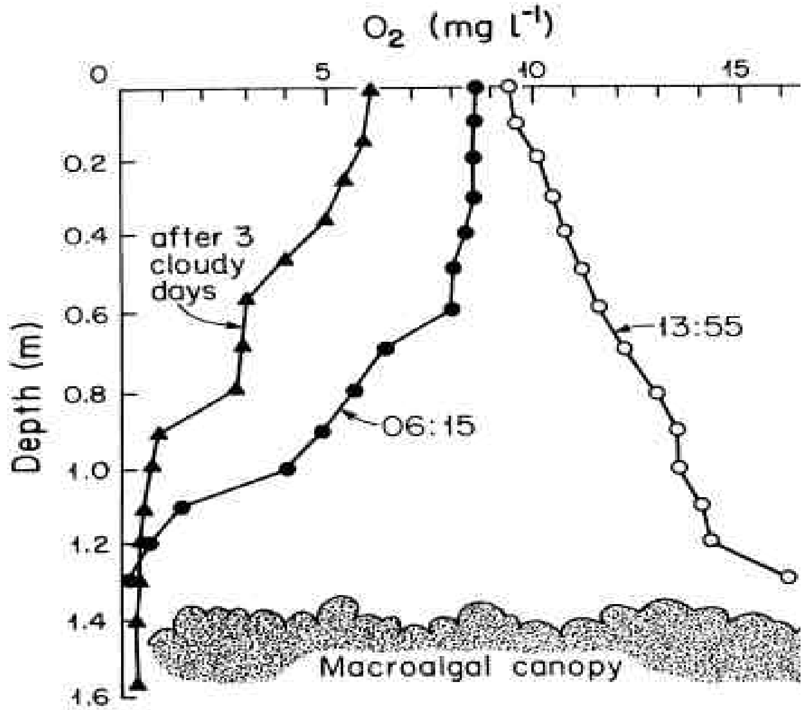
[a]From high-resolution, *in situ* sampling. Converted to TSS as described in the text

Bay (C_0) and Harbor (C_1) concentration data used in calculating Harbor-Bay exchange. BA: beam attenuation from transmissometer readings; nd: not determined.

Waquoit Bay

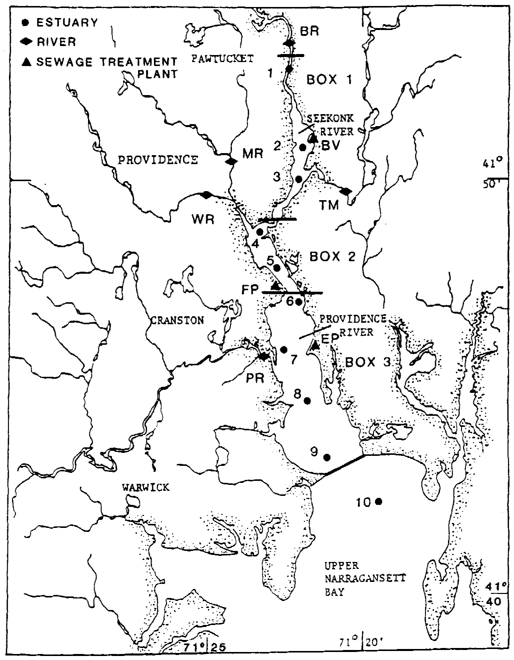


Time course of light, temperature, oxygen, ammonium, phosphate, and chlorophyll in water of Waquoit Bay, summer 1988. Modified from Costa et al. (in press).

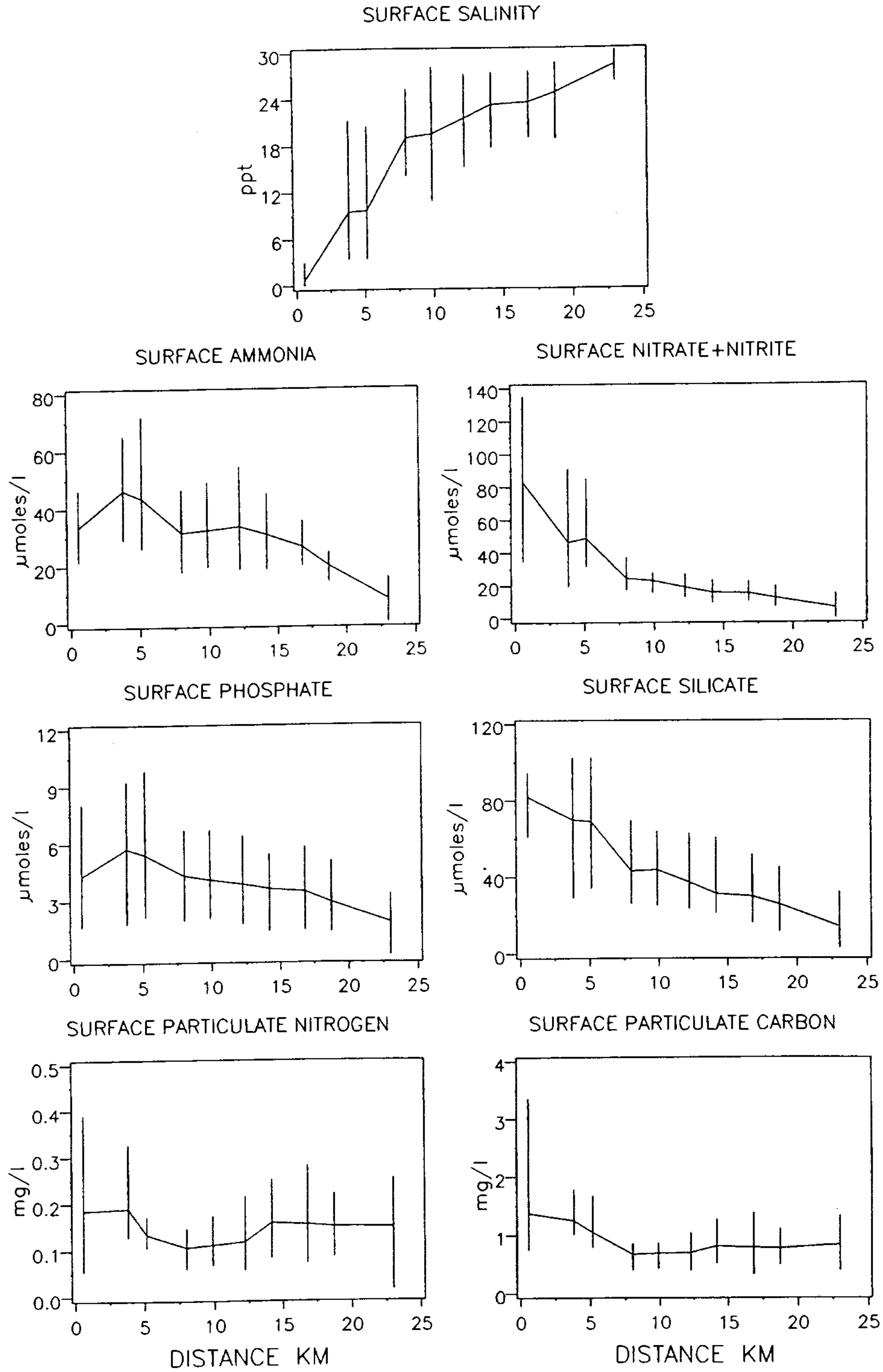


Vertical profiles of oxygen in Childs River during dawn and midafternoon of a sunny day (circles), and during afternoon after three cloudy days in a row. Data from C. D'Avanzo.

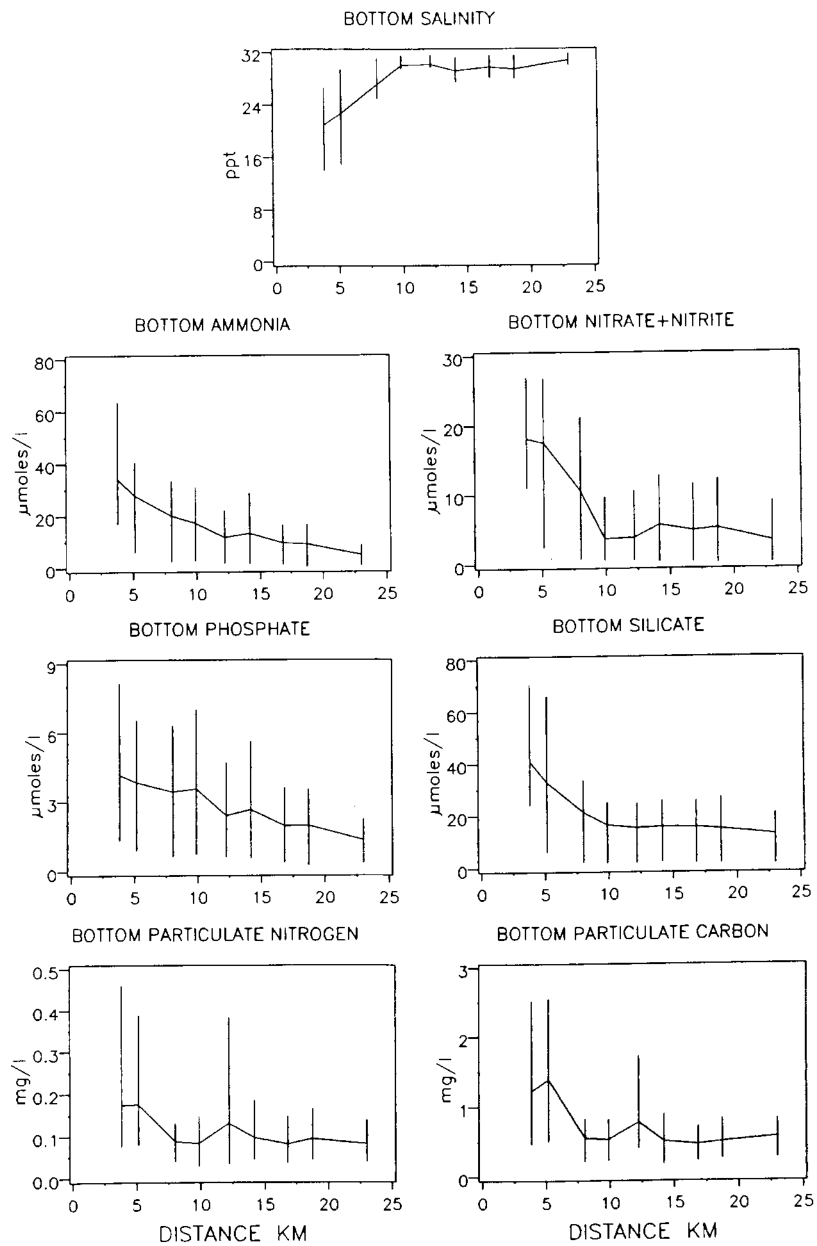
Seekonk—Providence River Region, Narragansett Bay



Station locations, Solid lines delimit boxes used in modeling effort. BR=Blackstone River, BV= Blackstone Valley Sewage Treatment Plant (STP), TM= Ten Mile River, MR Moshassuck River, WR=Woonasquatucket River, FP=Field's Point STP, EP=East Providence STP, PR= Pawtuxet River



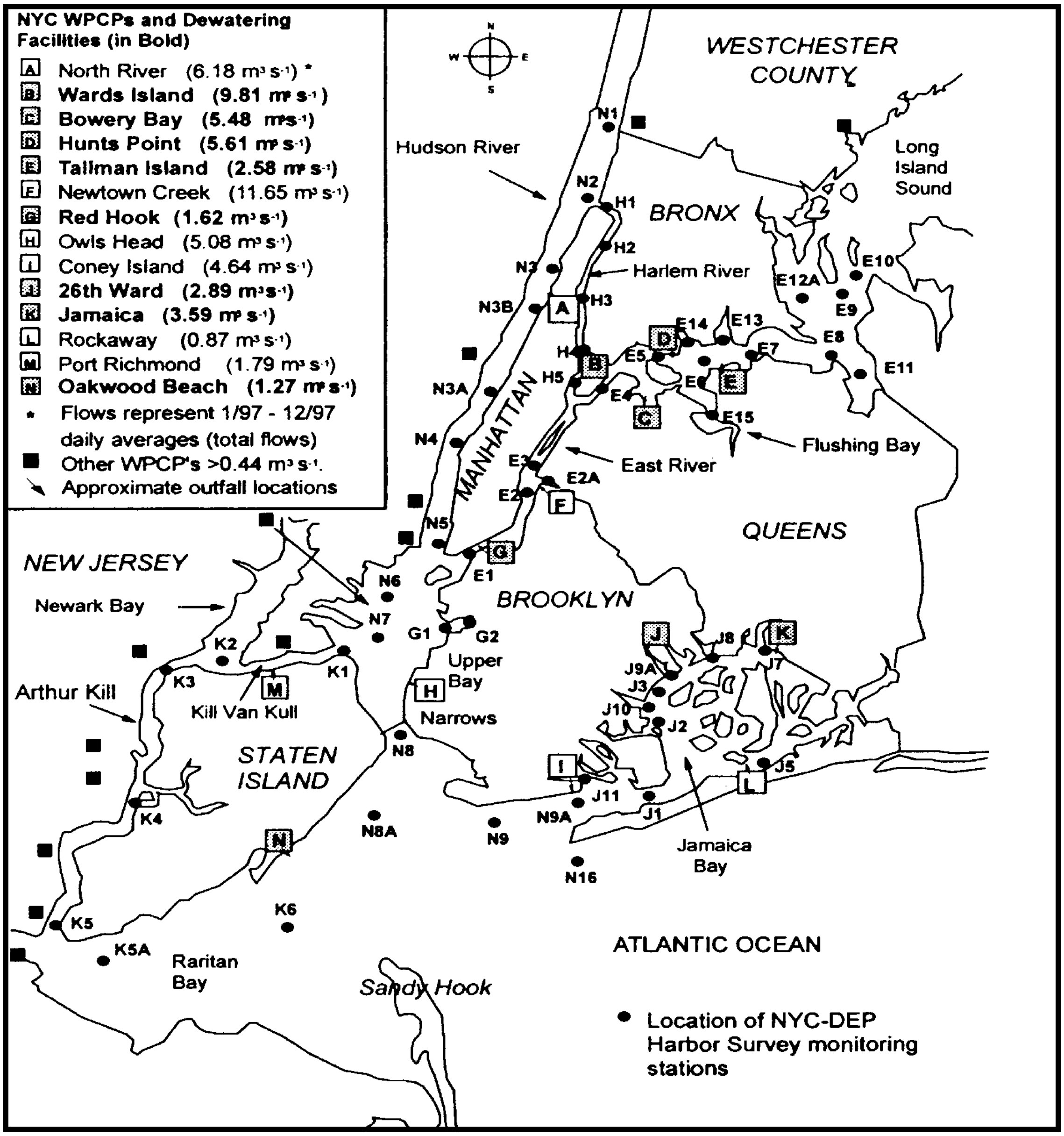
Mean and range of salinity and nutrient concentrations observed in surface waters (depth=1.0m) during the six cruises, versus distance from the Main Street Bridge, Pawtucket, Rhode Island at the head of the Seekonk River.



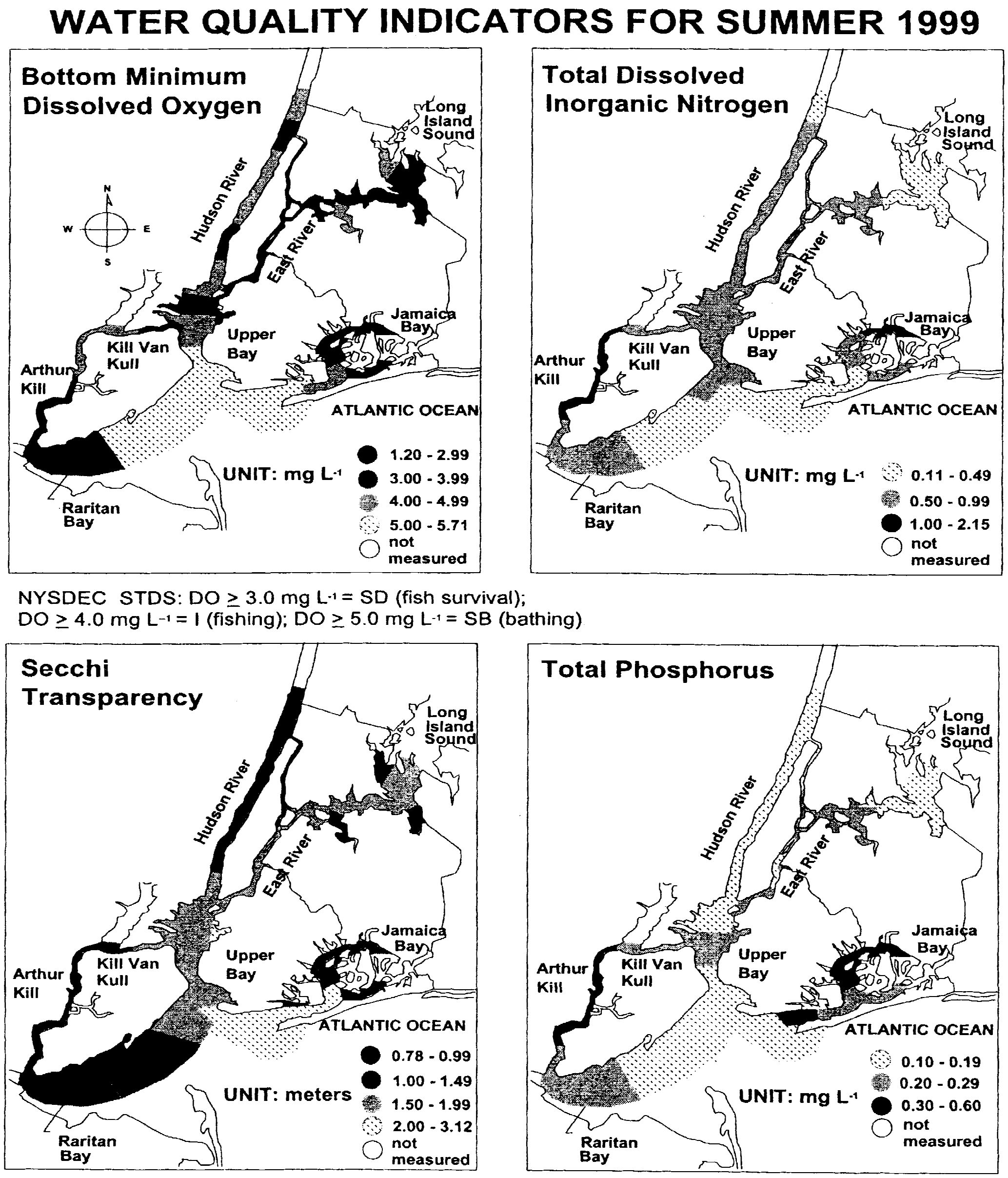
Mean and range of salinity and nutrient concentrations observed in bottom waters (1.0 m from bottom) during the six cruises, versus distance from the Main Street Bridge, Pawtucket, Rhode Island at the head of the Seekonk River.

Western Long Island Sound
Hudson-Raritan Estuary

LOCATION OF WPCP's AND SAMPLING STATIONS IN NY HARBOR

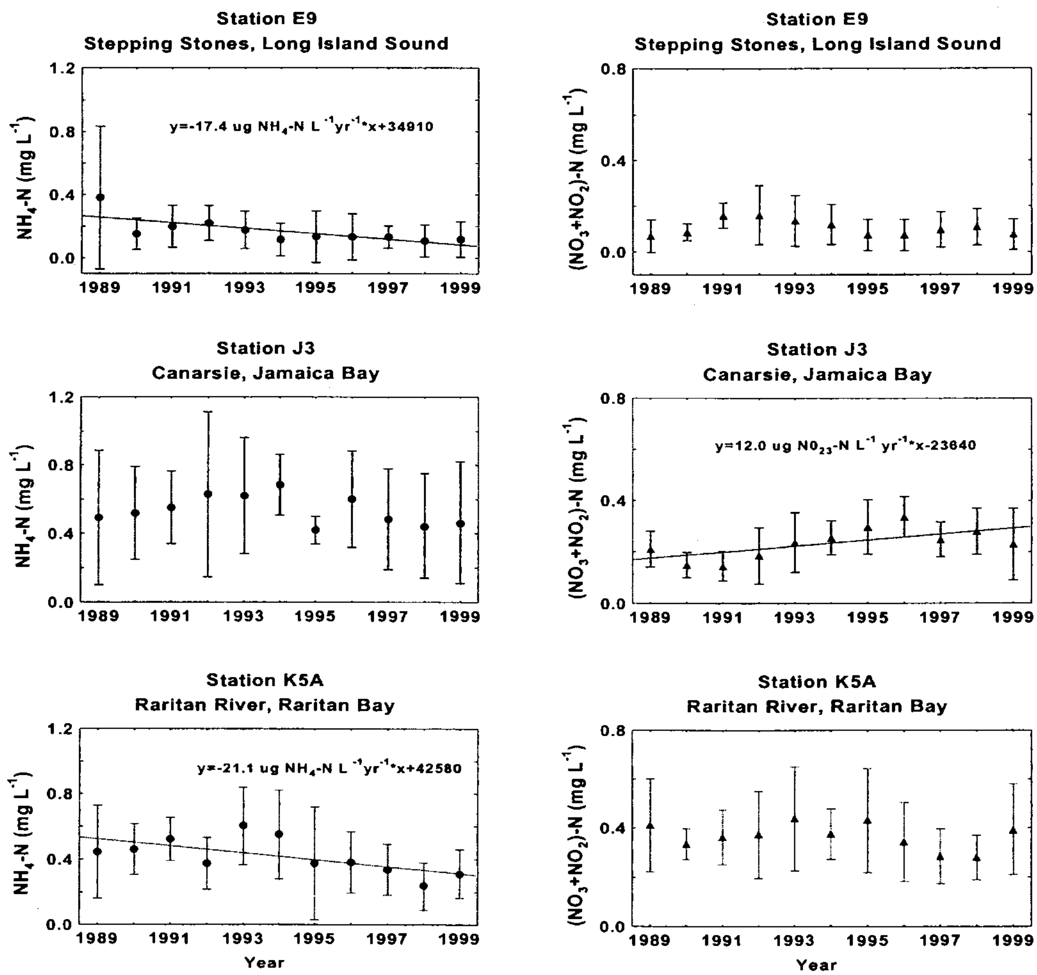


Location of New York City Department of Environment Protection's Harbor Survey Water Quality Monitoring Stations. Also depicted are the 14 New York City Water Pollution Control Plants and the 8 New York City sludge dewatering facilities (shaded boxes). Sandy Hook, lower center, is located at approximately 40°30'N, 74°W.



Key water quality indicators for summer (June–September) of 1999. Depicted are: bottom minimum dissolved oxygen (upper left) in mg l^{-1}; summer average dissolved inorganic nitrogen [NH_3–N+(NO_3+NO_2)–N] (upper right) in mg l^{-1}; summer average Secchi transparency (lower left) in m; summer average total phosphorus (lower right) in mg l^{-1}.

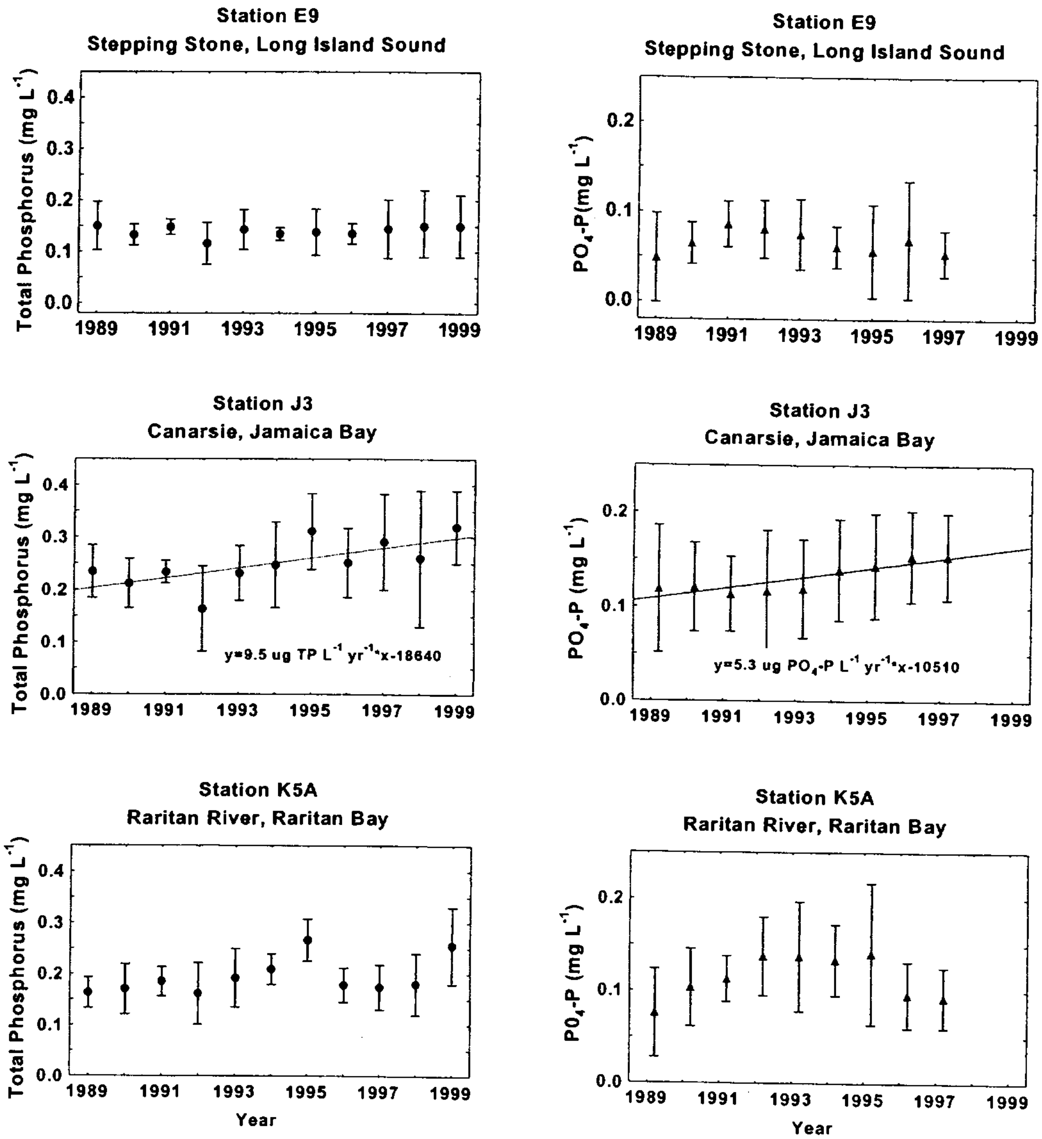
Dissolved Inorganic Nitrogen Trends
Summer Means, 1989-1999



Whisker=Mean±SD

1989–1999 summertime (June–September) average ambient dissolved inorganic nitrogen [dissolved ammonium-nitrogen (NH_4-N) and dissolved nitrate- and nitrite-nitrogen (NO_3)-N] concentrations (mg l^{-1} for western Long Island Sound, Raritan Bay, and Jamaica Bay stations. Trends in summertime average concentrations are noted where significant ($p < 0.05$).

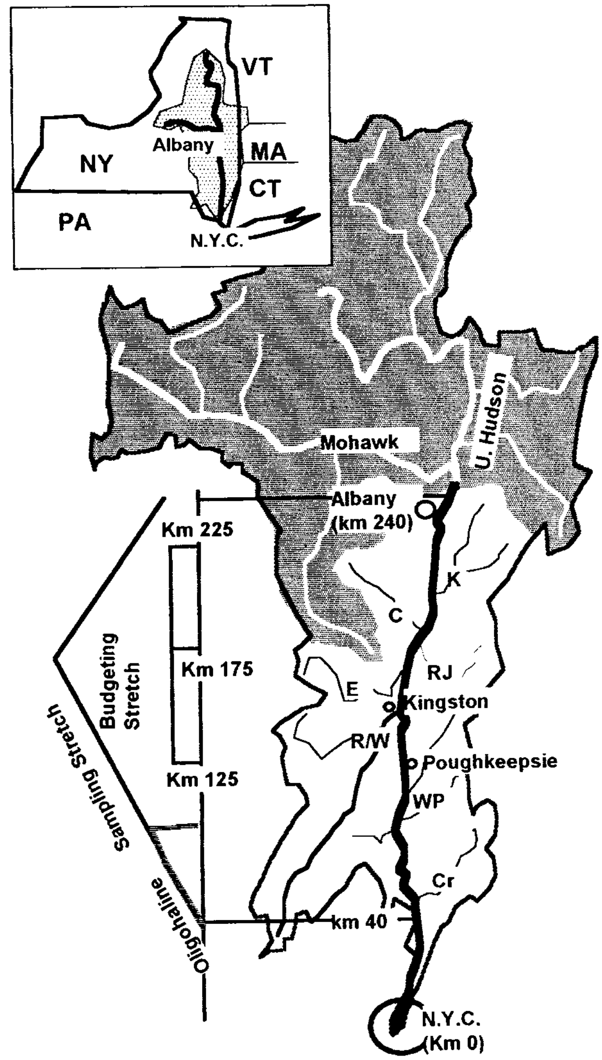
Phosphorus Trends
Summer Means, 1989-1999



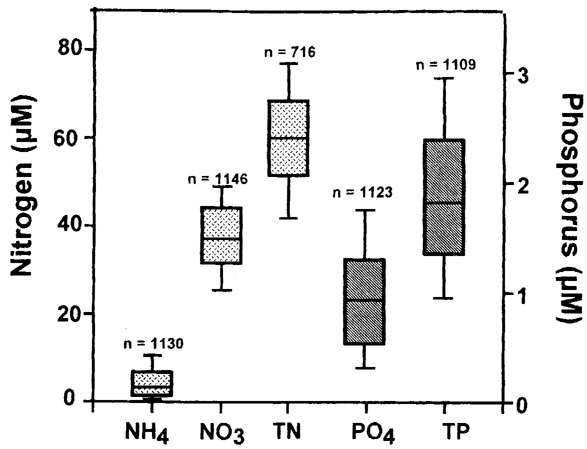
Whisker=Mean±SD

1989–1999 summertime (June–September) average ambient total phosphorus (TP) and dissolved orthophoshpate (PO_4-P) concentrations (mg l^{-1}) for western Long Island Sound, Raritan Bay, and Jamaica Bay stations. Trends in summertime average concentrations are noted where significant ($p < 0.05$).

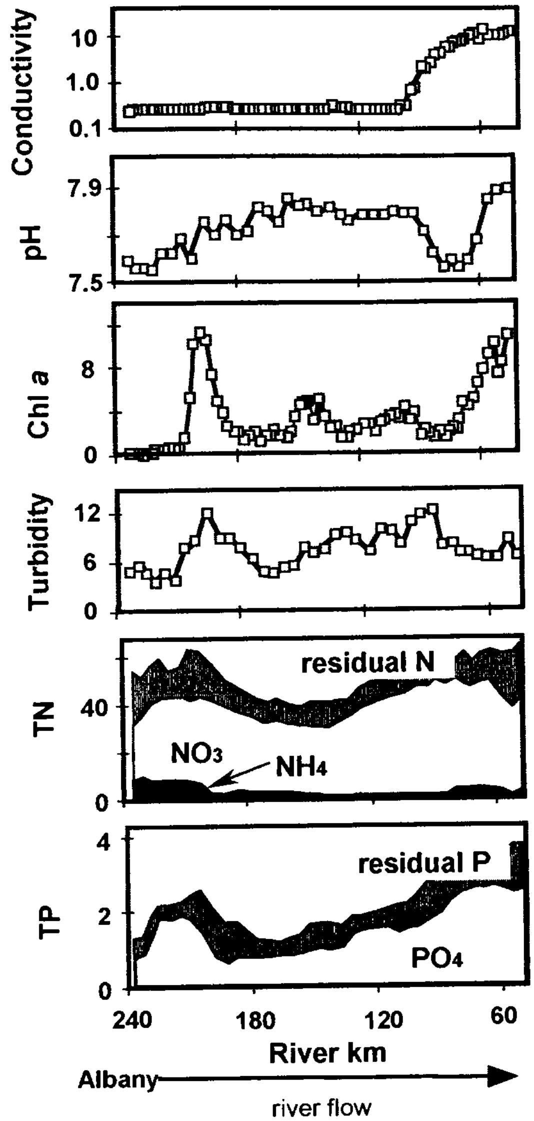
Hudson River, NY



Map of the tidal freshwater Hudson River, its watershed and major tributaries. The inset shows the location of the watershed of the watershed of the Hudson, primarily in eastern New York State. The main panel shows the watershed of the Hudson. The upper (non-tidal) tributaries are in white and their surrounding watershed is shaded. The remaining, unshaded, section of the watershed delivers water to the tidal Hudson River (heavy black line) primarily through 7 tributaries. These tributaries (creeks) are labeled: K for Kinderhook; Cfor Catskill; RJ for Roeloff Janson Kill; E for Esopis; R/W for Rondout/Wallkill; WP for Wappingers; Cr for Croton. The first 5 of these tributaries enter within the budgeting reach (labeled) between km 225 and 125. The section of the river that was sampled during our transect sampling (labeled sampling stretch) is also shown.

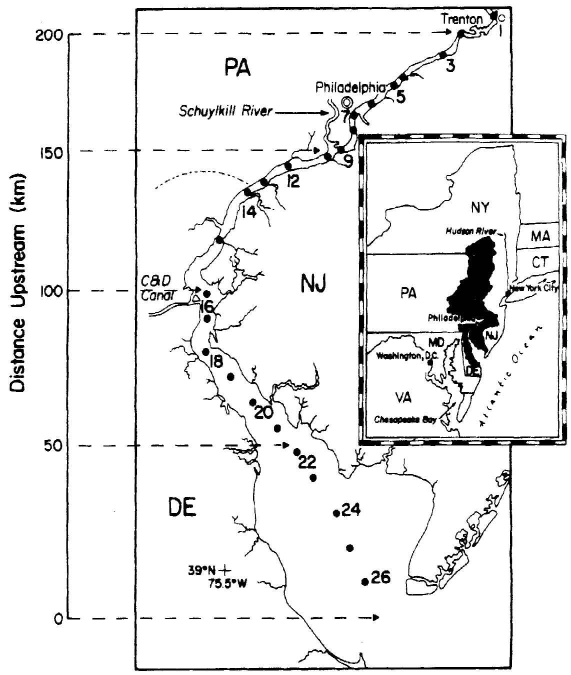


Box and whisker plots for forms of N and P for the entire data set of the Hudson. The data include all samples taken from January 1992 to December 1996 and combine seasonal and spatial variation. Shown are the medians, upper and lower quartiles and 90% inclusion lines. The number of samples for each analysis is labeled near the boxes.



Spatial variation in selected variables for a representative transect taken in early September 1996. Transects run km 240 (Albany downstream to km 40. For each variable samples were taken every 2 to 4 km. Units are, conductivity—(note log scale); pH—normal pH units; chlorophyll *a*— 1^{-1}; turbidity— NTU; N and P—μM. For both N and P the per line represents total N and toal P. "Other N" is TN minus DIN (NH_4 plus NO_3); For P, "Other P" is TP minus phosphate.

Delaware River Estuary

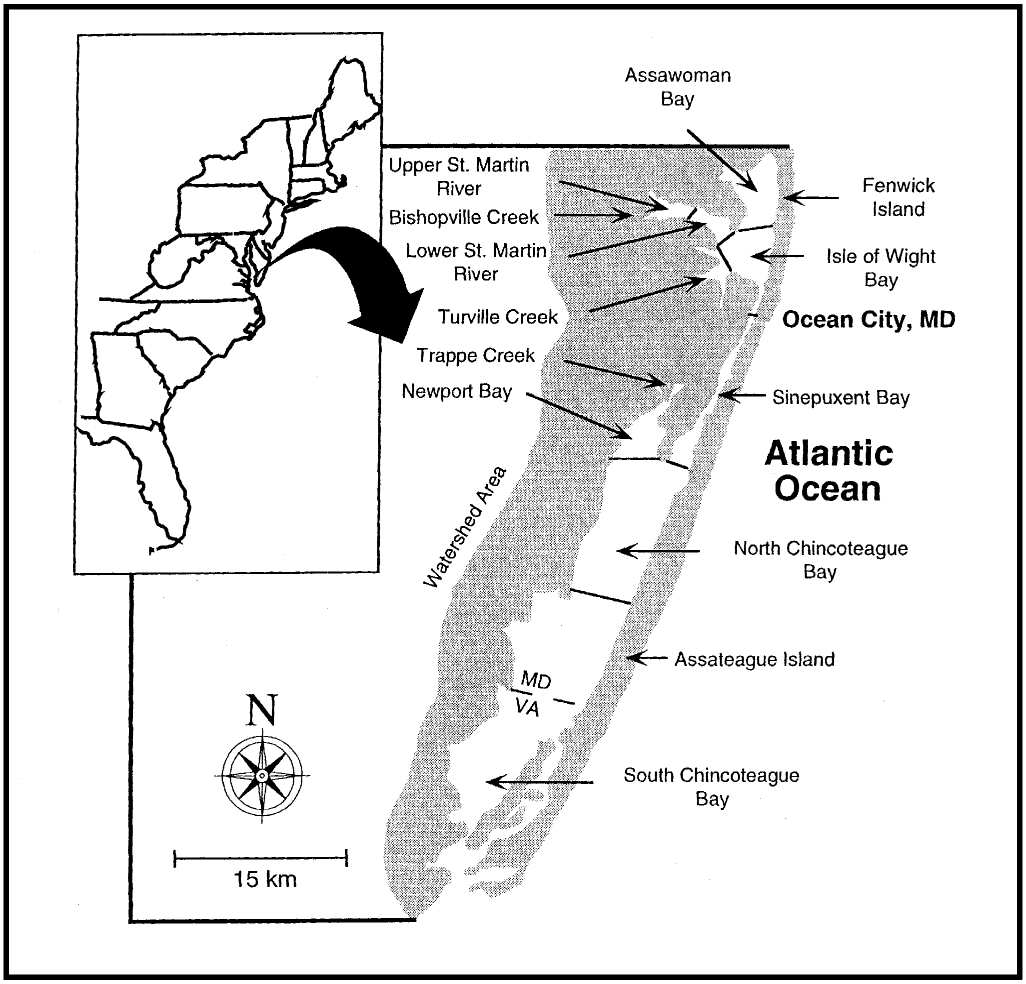


The Delaware Estuary with insert showing relation to the mid-Atlantic coast of the United States. Stations (1-9, 11-26) were located along the main axis of the estuary, every 5-15 km. Station 10 was located in the Schuylkill River (not shown). The river is tidal from the mouth of the bay up to the fall line at Trenton, New Jersey (240 km). Salinity increases from 0.1% at station 14 (130 km) to 30% at the bay mouth. The majority of TP inputs to the river are clustered between stations 6-9.

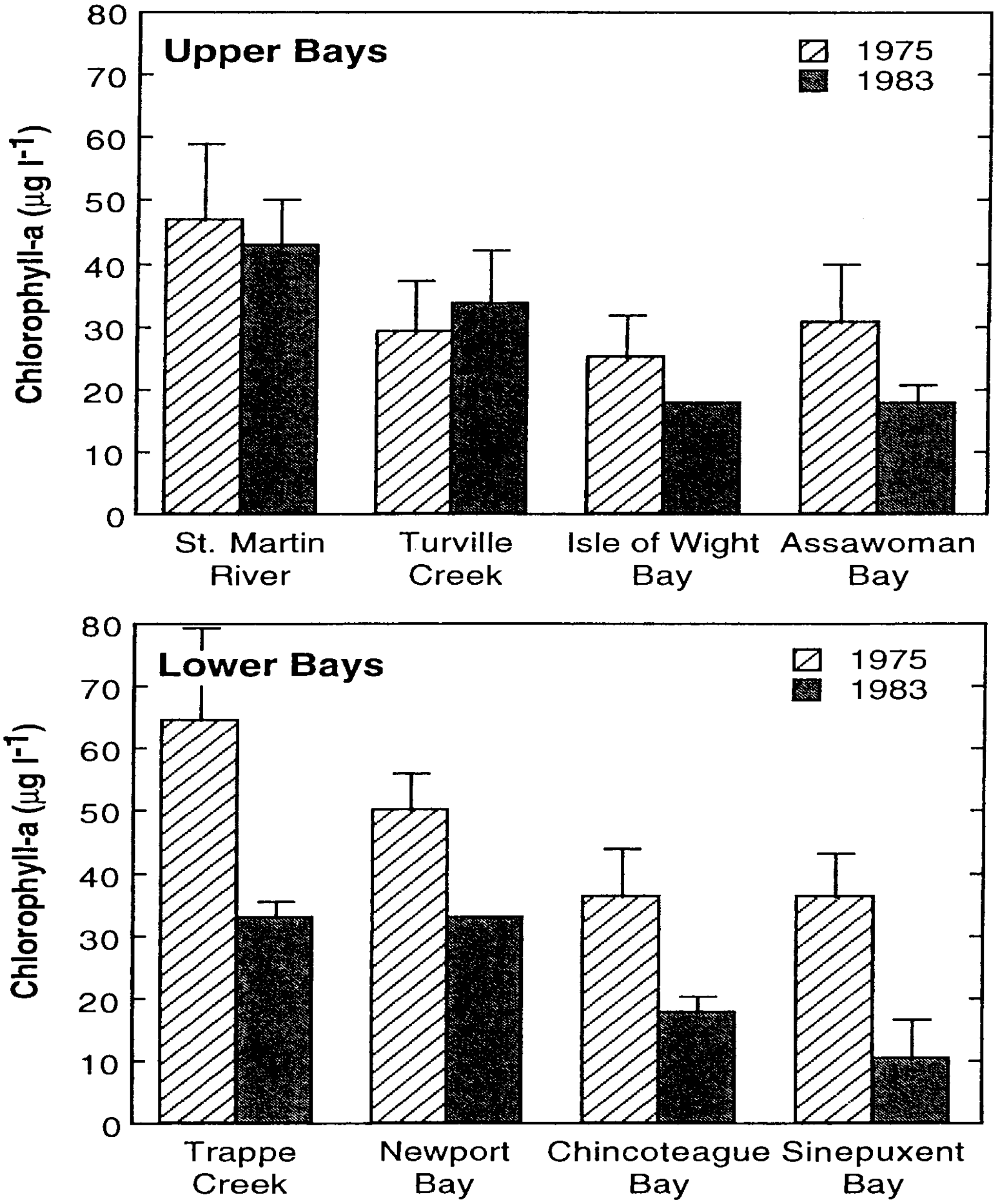
Parameter	Units	2	7	14	16	20	23	RSD
Distance	km	197	161	127	101	66	39	—
Salinity	ppt	<0.1	<0.1	0.1	1.5	11.2	20.4	—
pH	NBS Scale	7.1	6.8	6.9	7.1	7.7	8.0	4
Dissolved oxygen	Percent of saturation	87	73	76	82	96	105	16
Nitrate	μM	74	88	121	122	72	29	35
Total nitrogen	μM	98	126	163	169	94	41	27
Dissolved organic carbon	μM	253	283	287	285	277	208	25
Particulate carbon	μM	36	39	52	138	69	54	47
Secchi depth	cm	129	124	79	33	64	120	36
Phytoplankton production	mmol C m^{-2} d^{-1}	54	40	23	12	47	102	119

Figure 16 Average biological, chemical, and physical parameters for Delaware Estuary 1986-1988. Values are the nonweighted mean of 17 to 24 discrete samples taken at each location (Lebo et al. 1990). Data are shown for locations near the Delaware River (station 2), Philadelphia (station 7), beginning of the salinity gradient (station 14), turbidity maximum (station 16), downstream of the turbidity maximum (station 20), and in the lower bay (station 23). In addition, average relative standard deviation (RSD) for all stations is shown. The average standard deviation for station location and salinity was 1.1 km and 2.1%, respectively.

Maryland Coastal Bays

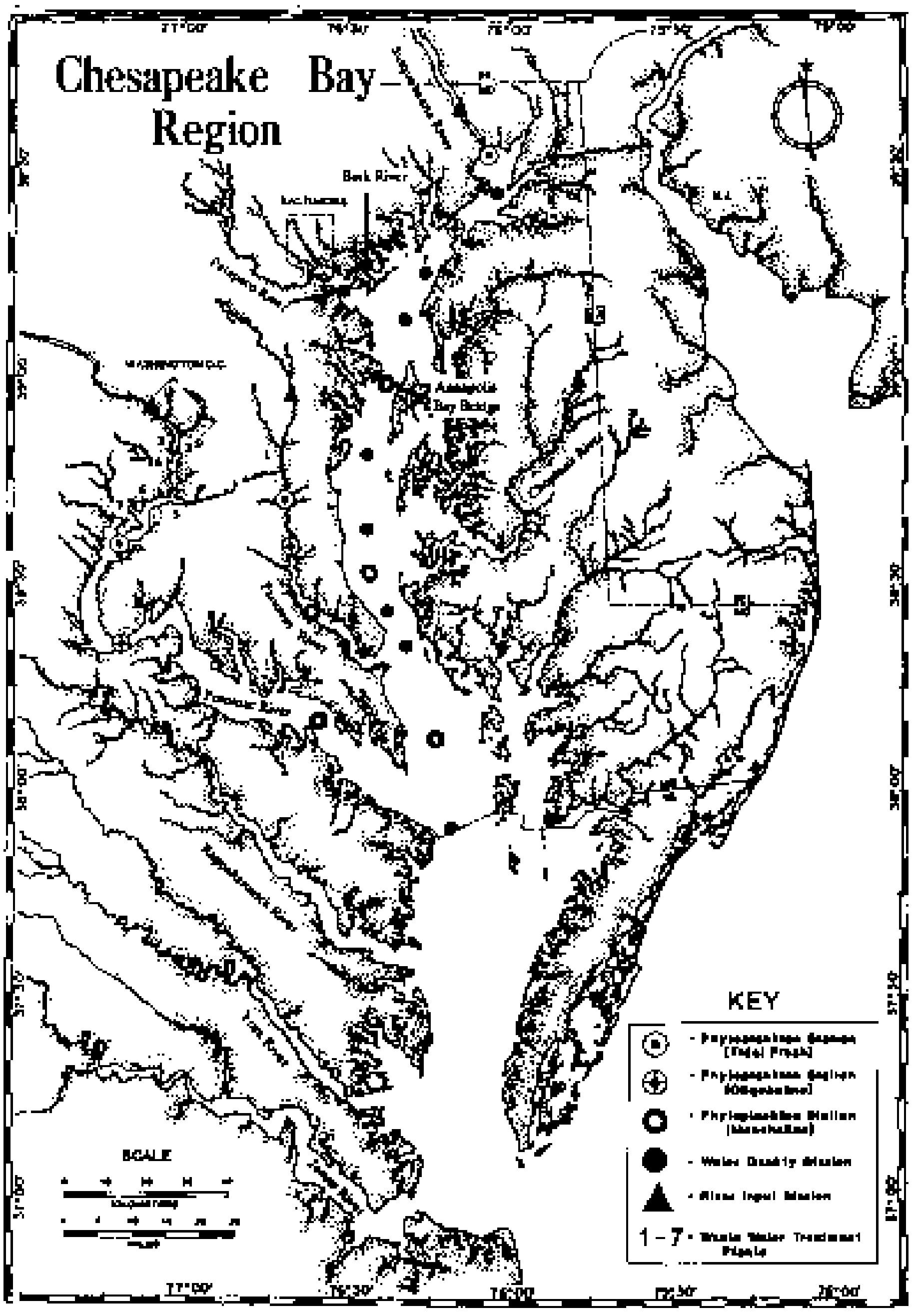


Map of the Maryland coastal bays complex indicating the boundaries of the watershed and a subsystems for which nitrogen inputs were estimated.

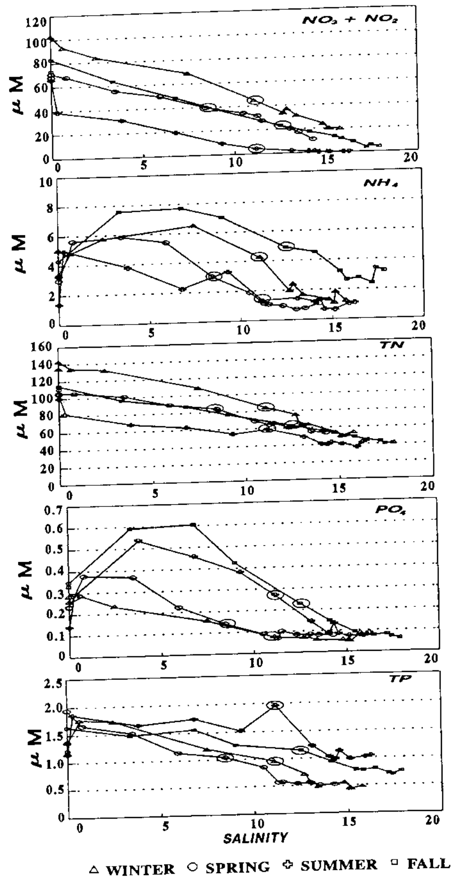


Summer average chlorophyll *a* concentrations for representative regions of the Maryland coastal bays based on samples collected during 1975, 1983, and 1991. Data are from Fang et al. (1977a,b), Maryland Department of Health and Mental Hygiene (1985), and National Park Service (1991).

Chesapeake Bay
Magnien et al.



Map of Chesapeake Bay region showing tributaries, sampling sites, and major wastewater treatment plants for systems examined or referenced in this paper. The major wastewater treatment plants are as follows: 1) Western Branch, 2) Arlington, 3) Blue Plains, 4) Alexandria, 5) Piscataway, 6) Little Hunting Creek, and 7) Lower Potomac.



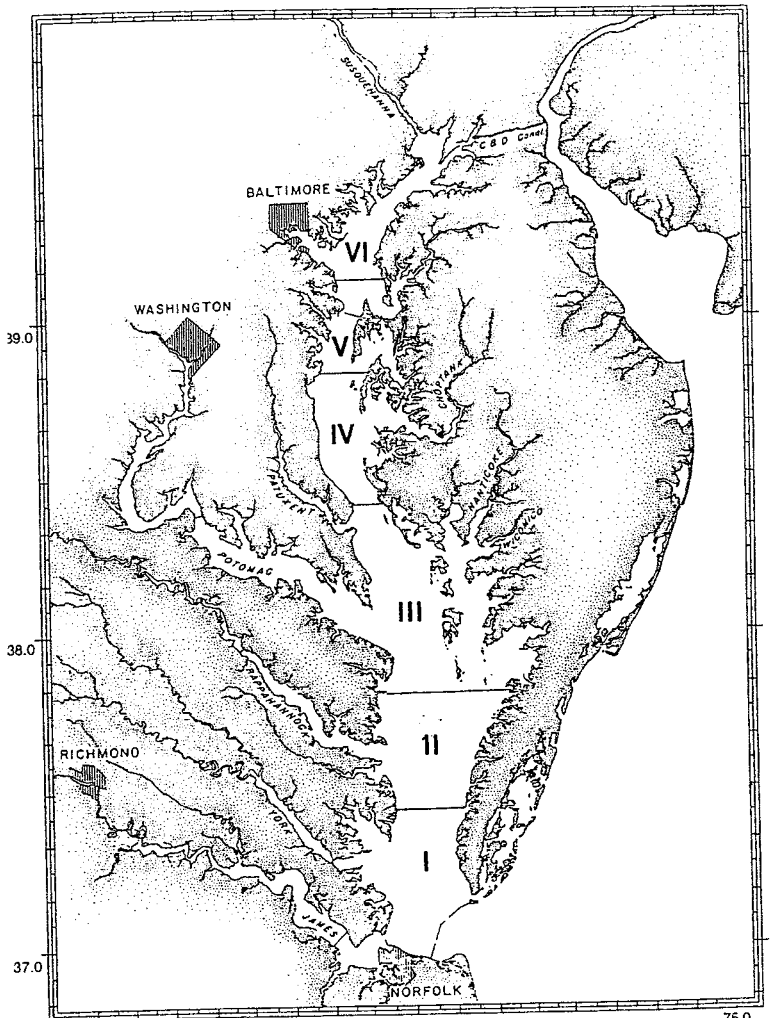
Salinity dilution plots of nitrate plus nitrite, ammonium, total nitrogen, orthophosphate, and total phosphorus for surface samples at all Chesapeake Bay Mainstream stations. Each point represents a single station at which median values were calculated for each seasonal time frame (see Fig. 2) over the entire 195-1989 period. All stations identified in Fig. 1 were plotted in longitudinal order starting at the head of the estuary. The circled point for each seasonal plot is the upper mesohaline plankton station at the Annapolis Bay Bridge.

Concentrations of ammonium (µM) in Back River, and ammonium concentrations, primary production estimates (mg C m^{-2} d^{-1}) and temperature (°C) for the Patapsco River(Baltimore Harbor). Values presented are surface mixed layer medians by month from continuous monitoring at a frequency of one to two times per month over the period 1985-1989. Station locations are centrally located in each system (see Fig. 1)

	Back River	Patapsco River		
Month	**NH_4^+**	**NH_4^+**	**Primary Production**	**Temperature**
January	257.0	8.9	299	3.2
February	414.0	48.2	1,384	3.0
March	112.4	24.3	640	6.2
April	439.0	33.6	663	12.3
May	151.7	11.7	1,520	16.8
June	35.7	9.7	1,840	24.3
July	82.1	13.0	2,263	27.2
August	87.5	30.3	1,950	26.6
September	107.1	18.8	1,744	23.8
October	103.5	21.3	953	17.7
November	232.0	15.4	751	11.3
December	317.7	13.1	930	6.5

Chesapeake Bay

Harding and Perry



Chesapeake Bay showing locations of the 6 regions.

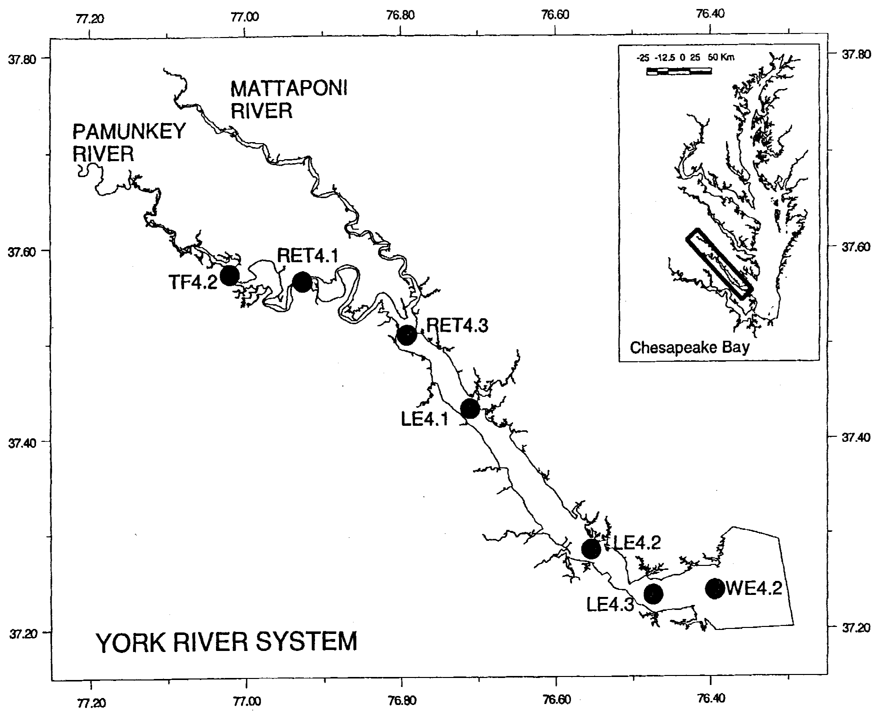
Regions in the Chesapeake Bay by latitude and geographic location.

Region	**Latitude range**	**Geographic location**
I	36.95-37.40^{N}N	Mouth of Bay to Mobjack Bay
II	37.41-37.80^{N}N	Mobjack Bay to Rappahannock River
III	37.81-38.40^{N}N	Rappahannock River to Patuxent River
IV	38.41-38.80^{N}N	Patuxent River to South River/Annapolis
V	38.81-39.10^{N}N	South River/Annapolis to Bay Bridge/Magothy River
VI	39.11-39.66^{N}N	Bay Bridge/Magothy River to Susquehanna Flats

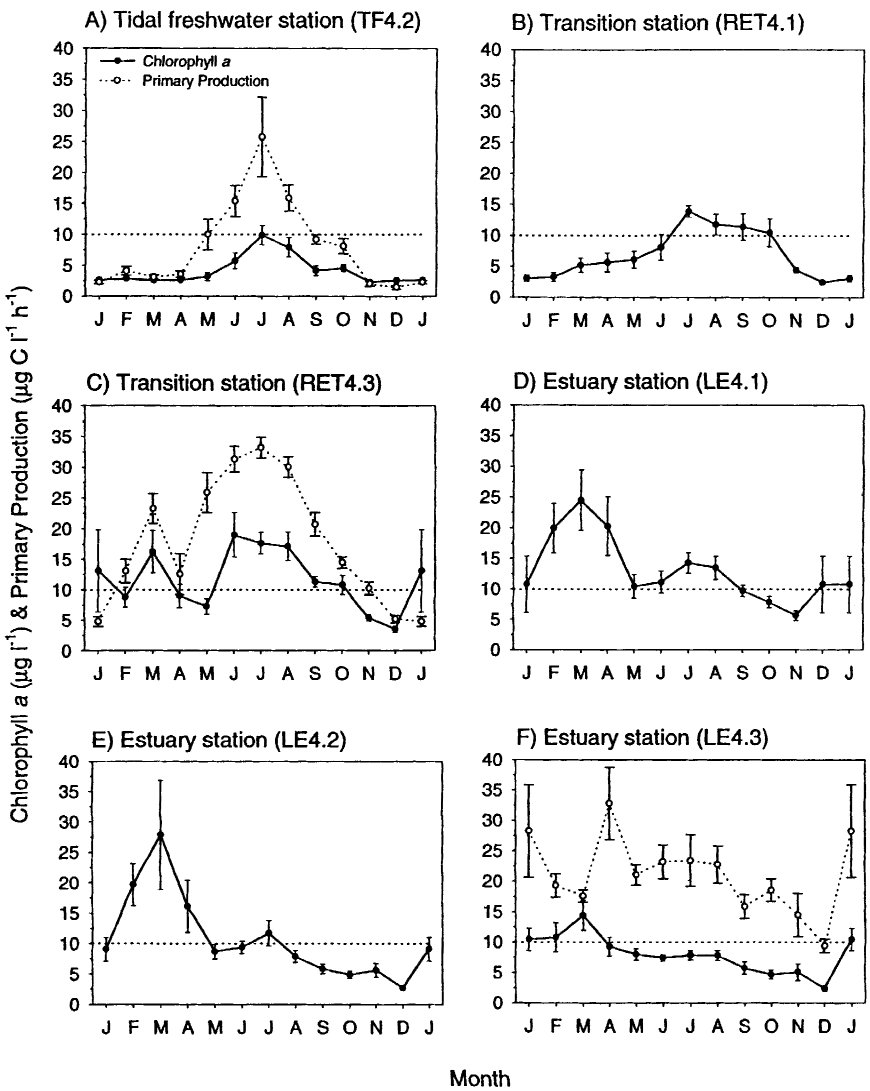
Year	Region	n	LS mean	95% LCI	95% UCI
1985	I	227	5.17	4.63	5.75
	II	254	5.77	5.22	6.38
	III	149	7.19	6.32	8.17
	IV	142	7.84	6.88	8.92
	V	124	10.5	9.17	12.0
	VI	106	7.17	6.15	8.34
1986	I	240	5.21	4.68	5.78
	II	243	6.79	6.13	7.50
	III	161	7.42	6.56	8.38
	IV	141	7.81	6.85	8.89
	V	143	10.6	9.39	12.1
	VI	118	6.74	5.82	7.78
1987	I	265	8.91	8.12	9.78
	II	249	10.6	9.61	11.6
	III	173	9.86	8.79	11.1
	IV	148	10.5	9.26	11.8
	V	159	12.1	10.8	13.7
	VI	128	7.60	6.62	8.71
1988	I	251	5.05	4.55	5.60
	II	241	8.78	7.95	9.67
	III	160	10.3	9.17	11.6
	IV	124	8.42	7.33	9.66
	V	126	10.9	9.53	12.4
	VI	109	6.04	5.18	7.03
1989	I	243	5.98	5.40	6.62
	II	234	7.51	7.08	7.95
	III	252	8.39	7.62	9.23
	IV	116	8.93	7.74	10.3
	V	230	10.9	9.85	12.0
	VI	223	5.69	5.10	6.33
1990	I	218	6.31	5.67	7.02
	II	234	10.4	9.41	11.4
	III	130	9.90	8.66	11.3
	IV	129	9.49	8.30	10.8
	V	146	10.8	9.53	12.2
	VI	118	4.76	4.08	5.54

Summary statistics on surface chlorophyll concentrations (mg m^{-3}) from Monitoring Program cruises on the Chesapeake Bay, 1985-1990.

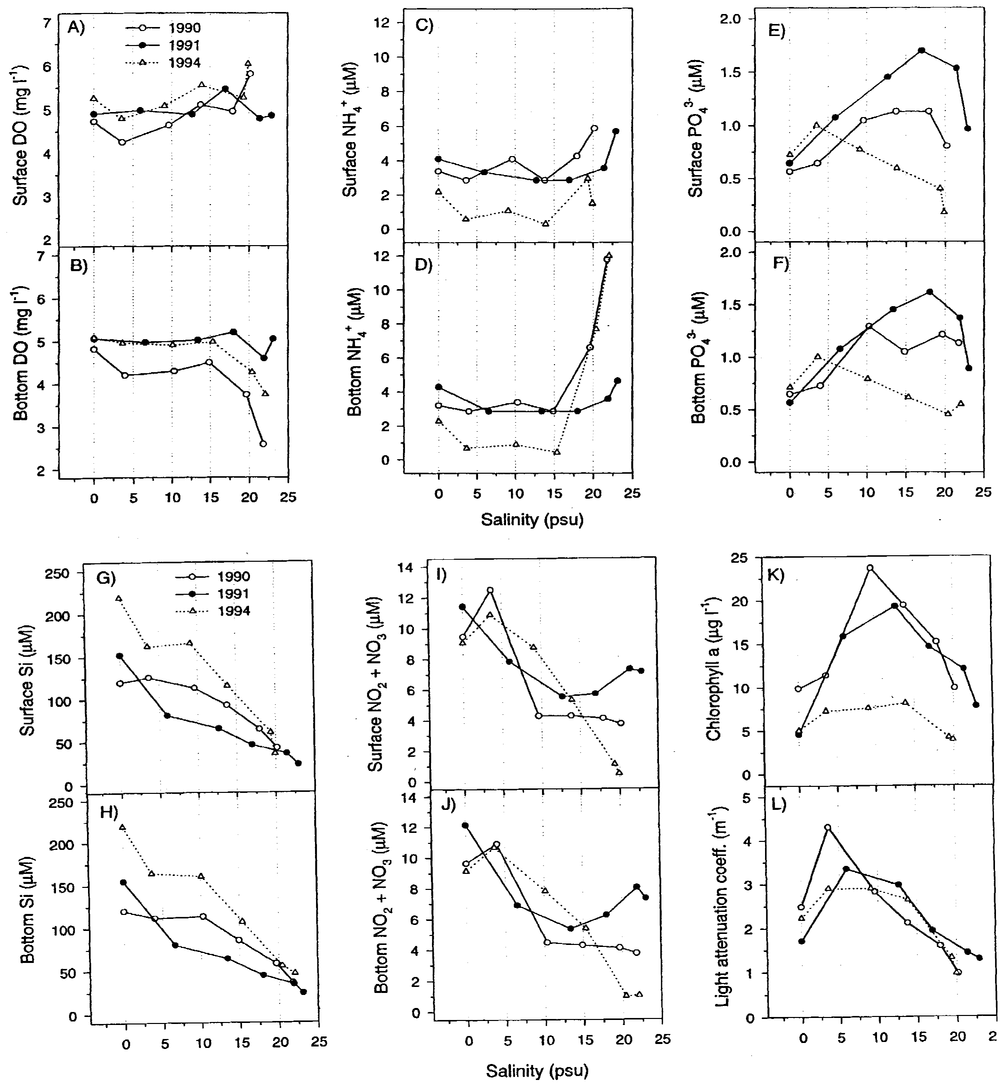
York River



The Environmental Protection Agency Chesapeake Bay Monitoring stations in the York River estuary.

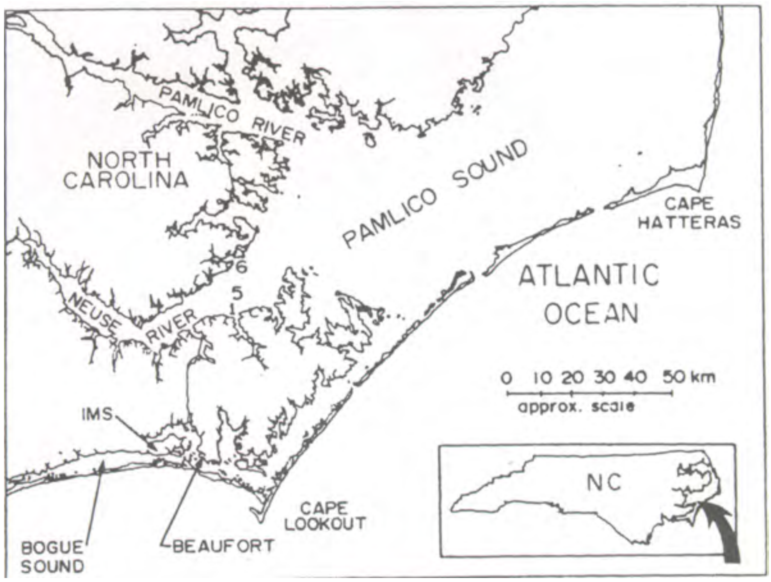


Seasonal distributions of chlorophyll *a* and primary production in the York River system; monthly means and standard errors were calculated from the 10 years data (1985-1994) for chlorophyll *a* and from 7 years data (1988-1994) for primary production. Dashed line at 10 µg 11^{-1} indicates our criterion for algal blooms and primary production shown in Fig 3F was measured at WE4.2.

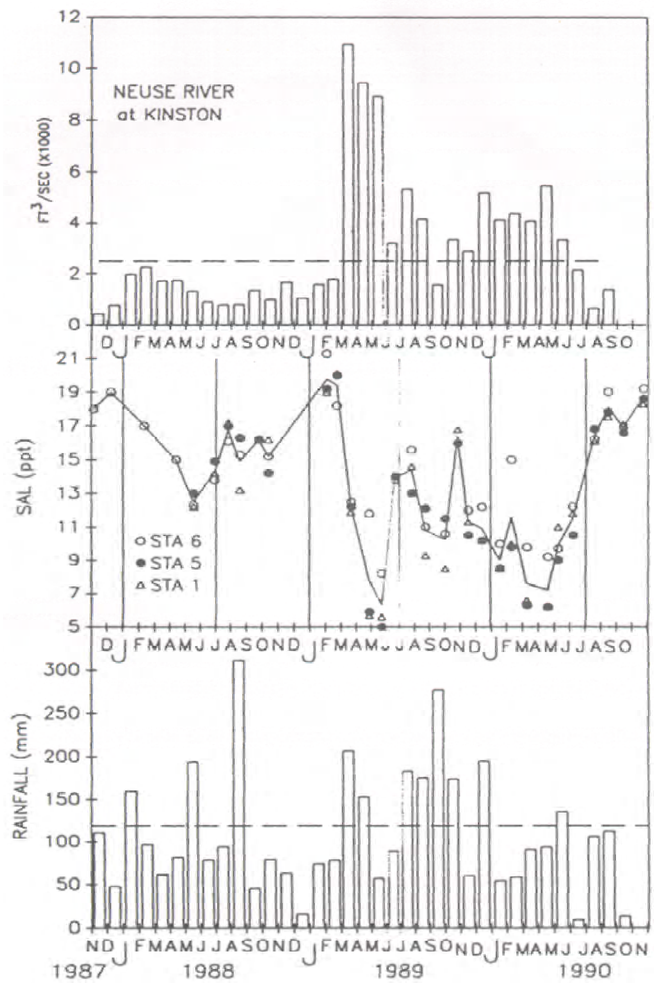


Salinity dilution curves of DO, ammonium, orthophosphate, silicate, nitrite + nitrate, chlorophyll a and light attenuation coefficient in the water column of the York River system for low (1991), mean (1990) and high (1994)flow years during the summer-fall period (June through October).

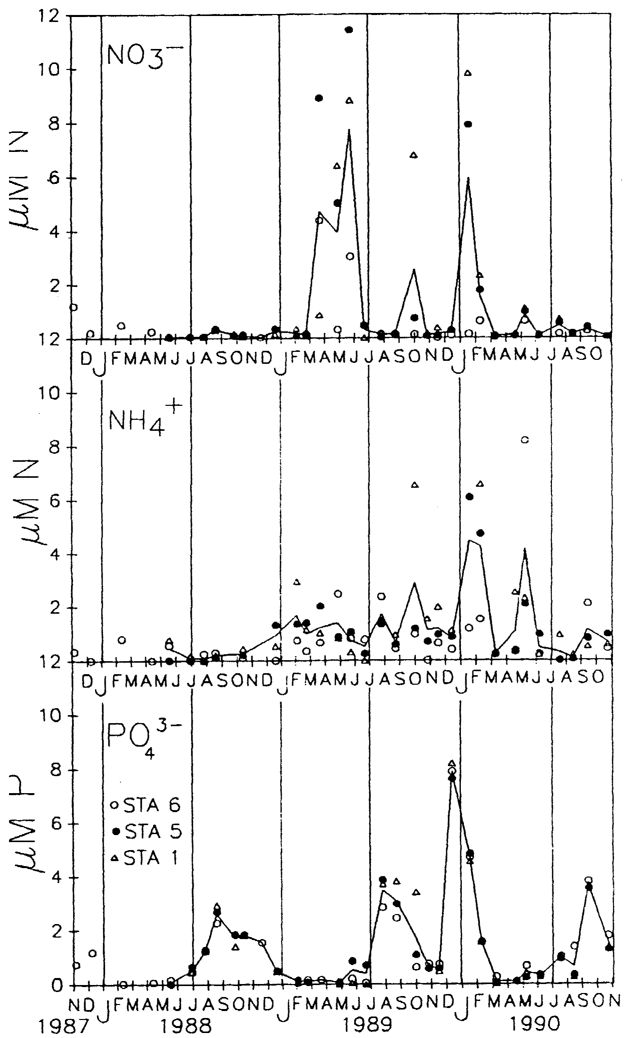
Neuse River Estuary, NC



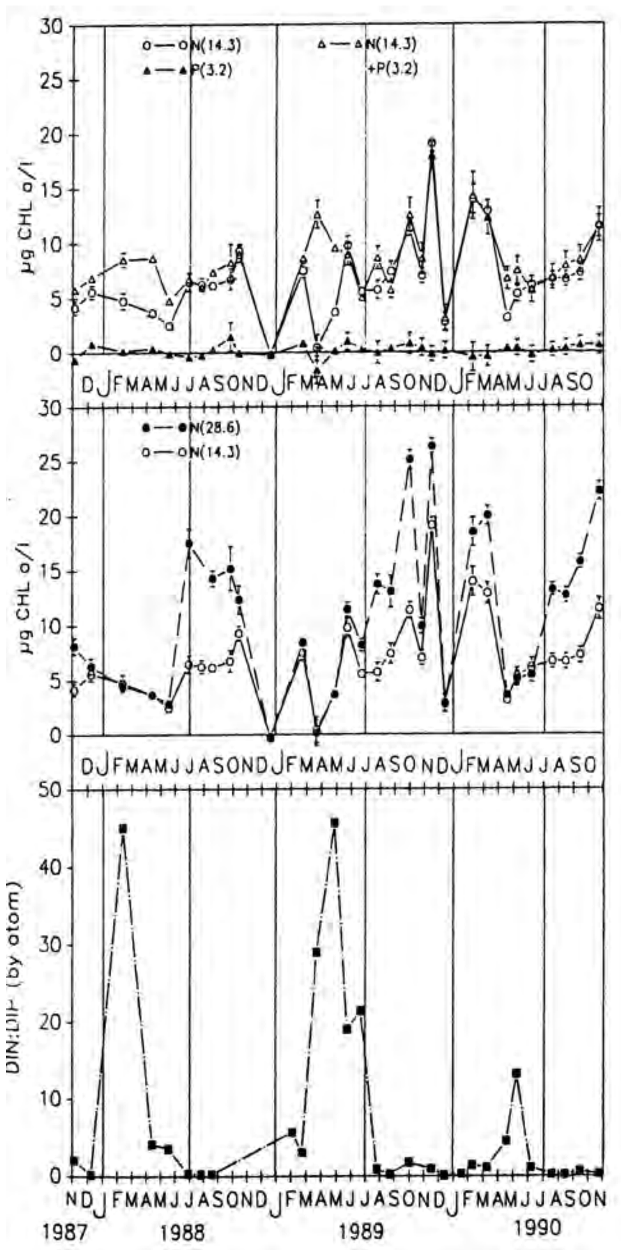
Sampling locations (Stns 1, 5, and 6) in the lower Neuse River Estuary, North Carolina, USA.



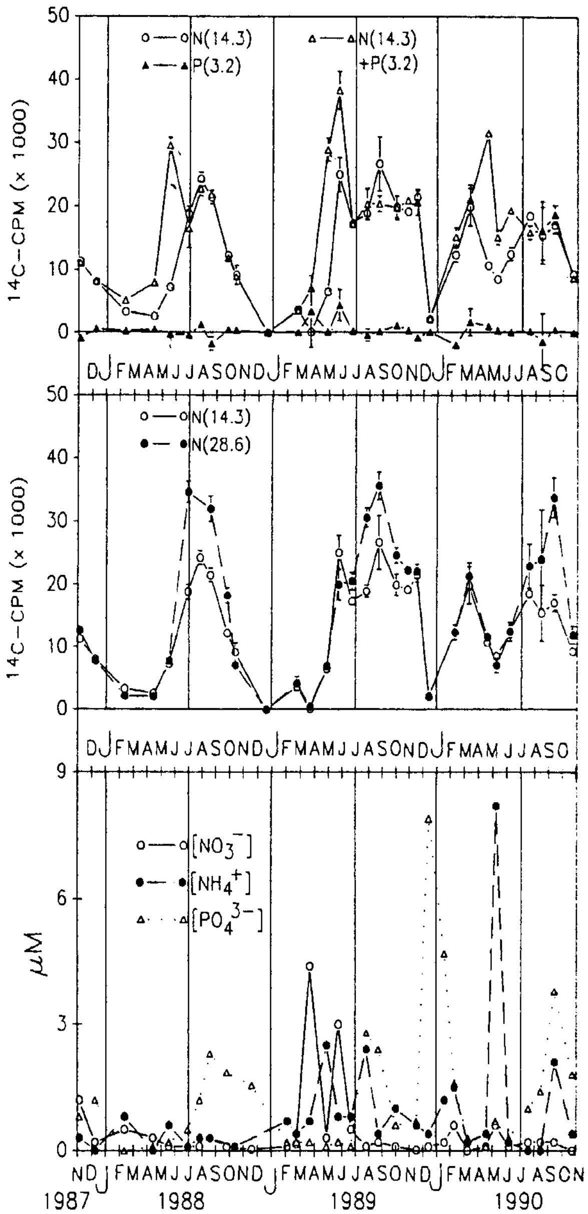
Neuse River mean monthly flows (× 1000 ft^3 s^{-1} or × 28.3 m^{-3} s^{-1}), at the US Geological Survey gauging station in Kinston North Carolina. Dashed line represents 60 yr average flow (J.D. Bales, US Dept of Interior, Geological Survey, Water Resources Division, Raleigh, North Carolina). (Middle) Surface measurements of salinity at Stns 1, 5, and 6 (see Fig. 1). Continuous line represents means of stations measure. (Bottom) Monthly rainfall totals at the Institute of Marine Sciences, Morehead City, North Carolina. Dashed line represents average monthly precipitation for the southern section of the Albermarle-Pamilico estuarine system area (118.5 mm mo^{-1}) (H. Porter, University of North Carolina Institute of Marine Sciences, Morehead City).



Nitrate, ammonium, and phosphate concentrations in surface waters at Stns 1, 5, and 6 (see Fig. 1). Continuous line represents means among stations.

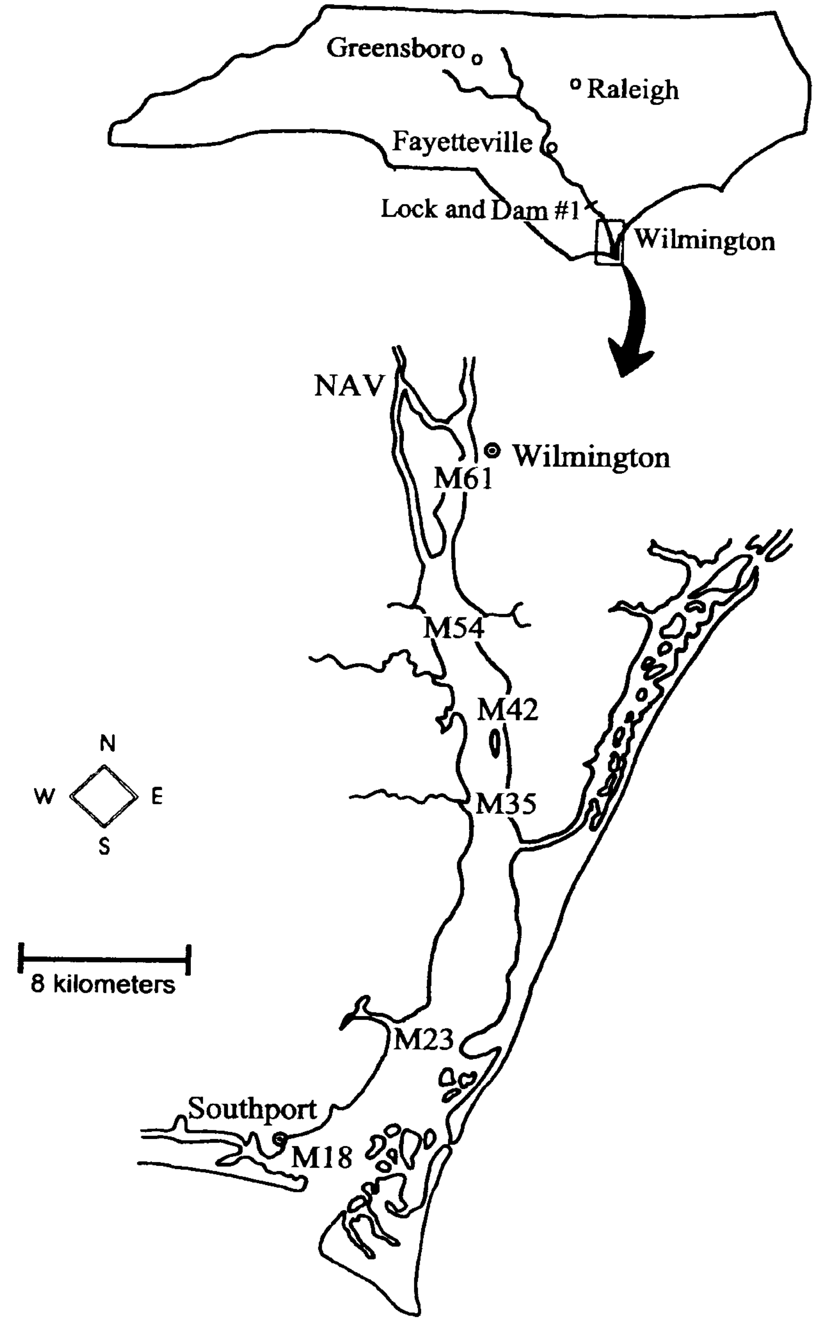


(Top and middle) Chlorophyll *a* concentration of selected nutrient addition treatments minus controls, averaged over the 4 d of each bioassay. N(14.3) and N(28.6) respectively indicate addition of 14.3 and 28.6 µm NO_3- P(3.2) indicates addition of 3.2 µm PO_4^{3-}. Error bars not visible are smaller than symbol. (Bottom) Dissolved inorganic nitrogen: dissolved inorganic phophorus (DIN:DIP) ratio (by atoms). No error bars plotted.

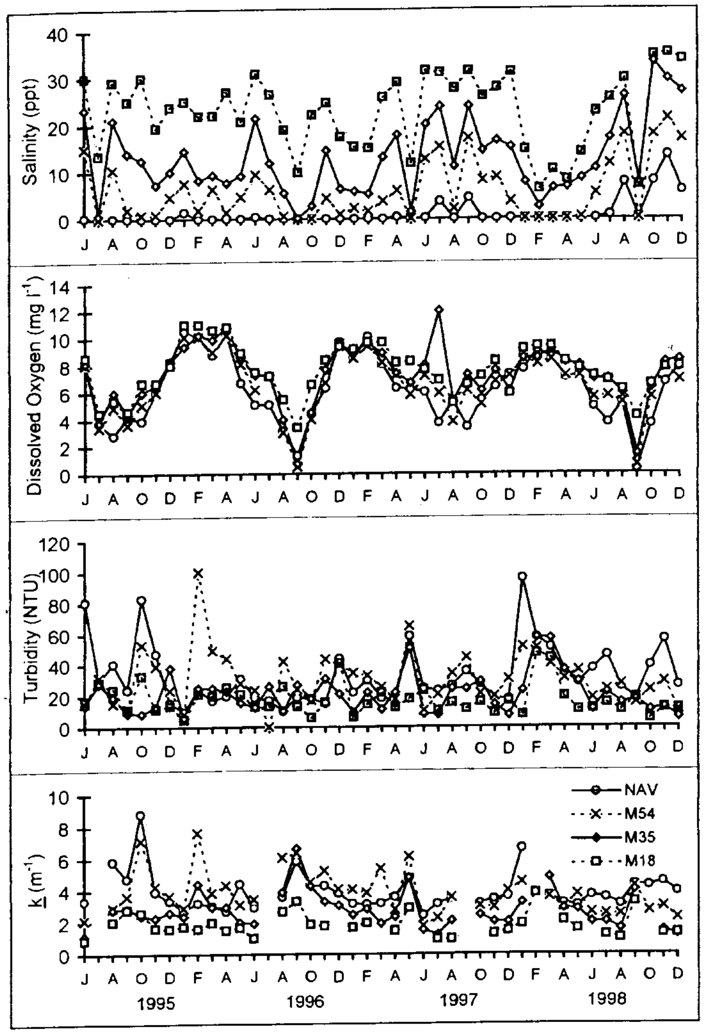


(Top and middle) ^{14}C assimilation of selected nutrient addition treatments minus xontrols, averaged over the 4 d of each bioassay. Symbols as in Fig. 4 (Bottom Nitrate, ammonium, and phosphate concentrations in surface waters at Stn 6 (see Fig. 1). Data are compiled from Fig.3 and presented here for comparative puposes.

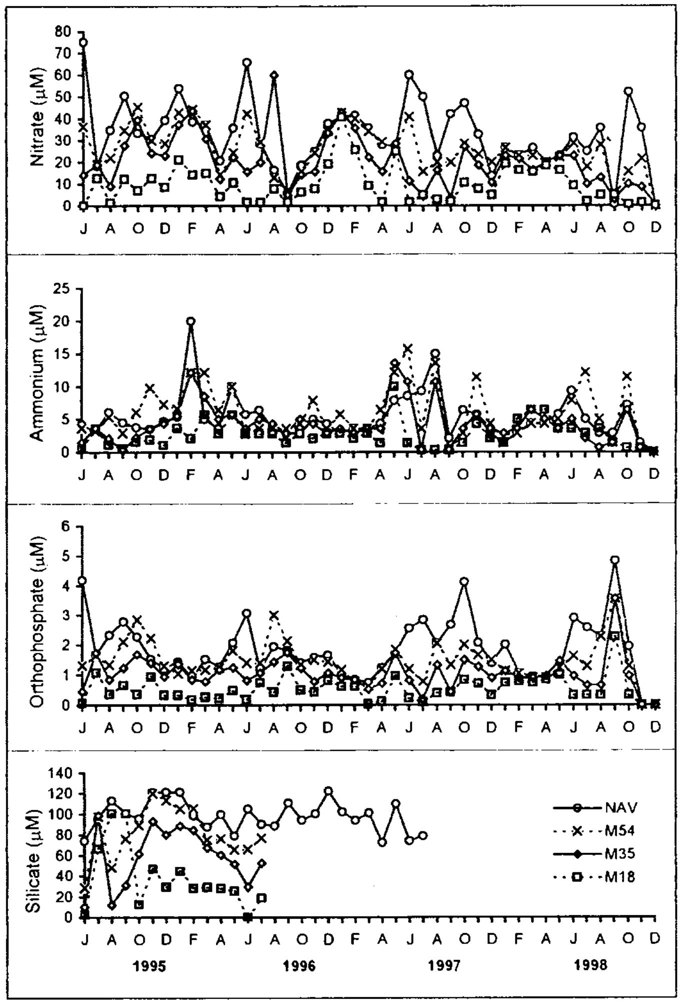
Cape Fear River Estuary



Sampling stations along the Cape Fear River Estuary, North Carolina, United States. The lower estuary is centered in 33°56′N, 77°58′W.



Physical parameters for selected stations in the Caper Fear River Estuary, June 1995-November 1998.



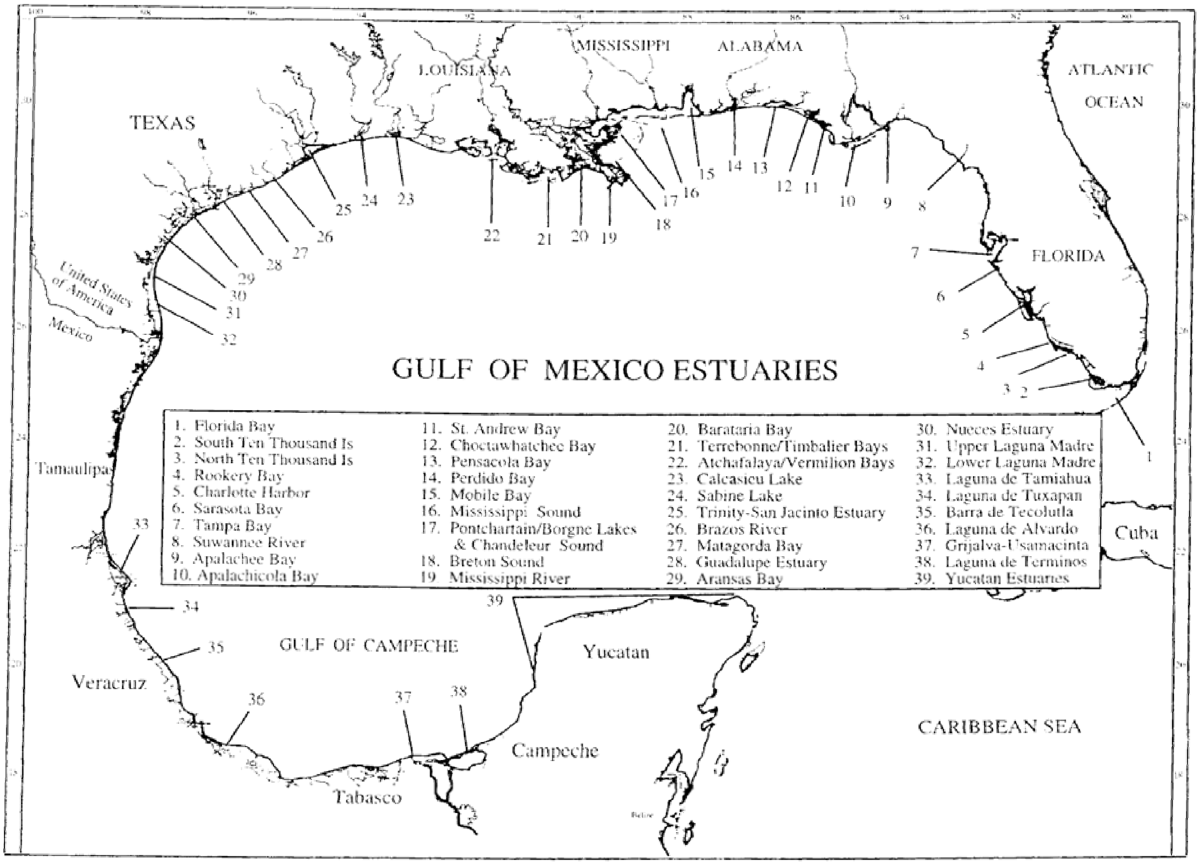
Inorganic nutrient concentrations for selected stations in the Cape Fear River Estuary, June1995- November 1998.

Coastal Georgia

Nutrient exchange (μg-atom m^{-2} d^{-1}) across the sediment/water interface and initial nutrient concentrations (μg-atom 1^{-1}) in the overlying water of the nearshore environment. Values given as release (+) or uptake (-) of the nutrient by the sediment.

Constituent	**Initial Concentration μM**
O_2	6.5
NH_4^+	1.1
$NO2^-$	0.11
$NO3^-$	0.16
DON	24.3
$PO4^{\equiv}$	1.4
DOP	0.0
Σ Inorganic nitrogen	1.26
Σ N	25.56
Σ Inorganic phosphorus	1.4
Σ P	1.4

Gulf of Mexico Estuaries



Map showing the distribution of estuaries in the Gulf of Mexico.

Site	Area (km^2)	Volume (10^6 m^3)	Mean Depth (m)	Maximum Length (km)	Maximum Width (km)	
Florida Bay, FL	1800	3200	2.0	70.0	40.0	Carbonate system low freshwater input
Tampa Bay, FL	896	3490	3.9	61.1	16.1	Marine dominated estuary
Apalachicola Bay, FL	593	1600	2.7	12.9	33.8	River dominated with micro-tides
Mobile Bay, AL	1060	3200	3.0	48.5	40.0	River dominated with seasonal coastal plume
Fourleague Bay, LA	56	72.8	1.3	16.0	3.5	Shallow, wetland dominated salinity variable usually low.
Nueces River Estuary, TX	538	1290	2.4	28.2	12.9	Event driven estuary, low river flows
Celestun Lagoon, Mexico	28	33	1.2	20.7	2.1	
Chelum Laggon, Mexico	13.6	16.3	1.0	14.7	1.8	Freshwater spring-fed lagoons.
Dzilam Lagoon, Mexico	9.4	11.2	0.8	12.9	1.6	
Rio Lagartos Lagoon, Mexico	96	76.8	0.8	80.0	1.5	

Table 5-1. Physical Characteristics of Gulf Estuaries Examined in Case Studies.

Site	Salinity (g liter^{-1})	NO_3 (μM)	NH_4 (μM)	PO_4 (μM)	SiO_2 (μM)
Florida Bay					
Eastern Region	29 (0.2–45)	0.62 (0.01–10.0)	3.19 (0.03–82.1)	0.03 (0.01–0.51)	16.0 (0.18–122)
Central Region	33 (9–63)	0.25 (0.01–5.70)	5.31 (0.01–120)	0.04 (0.01–0.84)	65.6 (0.06–109)
Western Region	35 (25–51)	0.12 (0.01–7.25)	0.12 (0.01–7.25)	0.03 (0.01–0.39)	18.6 (0.13–57.1)
Tampa Bay	— (20–35)	—	—	—	—
Apalachicola Bay	16.6 (0–37.3)	7.6 (1–35.7)	1.4 (0.1–7.1)	0.16 (0.1–0.62)	51.8 (3.0–136)
Mobile Bay					
Upper Bay	5.2 (0–19)	6.6 (0–40)	3.2 (0–17)	0.58 (0.05–1.4)	02.1 (13–131)
Mid Bay	9.1 (0–23)	4.1 (0–19)	2.4 (0–13)	0.46 (0.05–1.7)	59.7 (8–110)
Lower Bay	15.3 (0–32)	3.3 (0–18)	1.9 (0–12)	0.38 (0.05–1.6)	40.5 (2–104)
Fourleague Bay	—	42	2.5	0.8	—
Nueces River Estuary					
Nueces Bay	18.8 (3–30)	4.6 (0.5–23)			
Corpus Christi Bay	32.1 (28.6–37.9)	3.2 (1.5–11.3)	4.63 (0.5–19)	1.99 (0.6–4.6)	38.2 (10.5–60.8)
Celestun	25 (5–37)	4.82 (0.9–1.5)	7.82 (1–90)	0.82 (0.02–7.2)	54.4 (5–220)
Chelem	36 (27–43)	1.89 (1–6)	7.31 (1–38)	0.41 (0.1–6)	36.8 (4–50)
Dzilam	31 (30–37)	7.07 (1–10)	2.5 (1–15)	1.45 (0.2–8.1)	163 (12–210)
Rio Lagartos	57 (20–100)	0.7 (0.2–5)	8.5 (2–21)	1.55 (0.3–11)	24.7 (5–75)

Annual Mean and Range of Salinity and Nutrient Concentrations for Gulf Estuaries Examined in the Case Studies.

Site	Chlorophyll (μg liter^{-1})	Volume (mg C liter^{-1} d^{-1})	Primary Production Area (mg C m^{-2} d^{-1})	Annual (gC m^{-2} yr^{-1})
Florida Bay				
Eastern Region	0.76 (0.04–11.3)	—	—	75
Central Region	2.04 (0.21–11.6)	—	—	400
Western Region	1.83 (0.23–11.6)	—	—	250
Tampa Bay	7.5 (0–45)	—	—	—
Apalachicola Bay	5.6 (0–37.6)	21.4 (0.2–137.1)	771 (96–1812)	240 (188–300)
Mobile Bay	—	—	—	242 (194–325)
Upper Bay	6.5 (0.2–41)	—	642	—
Mid Bay	8.0 (0.2–55)	—	683	—
Lower Bay	5.5 (0.2–20)	—	661	—
Fourleague Bay	—	—	—	—
Nueces River Estuary	—	—	—	370 (270–400)
Nueces Bay	21.92	2.2 (0.2–7)	—	—
Corpus Christi Bay	8.4 (5.3–11.2)	1.2 (0.1–6)	—	—
Celestun	5.8 (0.5–28.5)	1.2 (0.21–2.1)	1440 (300–2520)	525 (109–919)
Chelem	2.8 (1.4–9)	0.42 (0.3–0.55)	420 (300–500)	153 (109–182)
Dzilam	2.7 (2–4)	0.53 (0.057–0.922)	420 (40–730)	153 (14–266)
Rio Lagartos	4.9 (2–10)	0.74 (0.054–0.875)	540 (40–700)	215 (14–255)

Annual mean and range of Chlorophyll-a Primary Production of Gulf Estuaries Examined in the Case Studies.

Galveston Bay

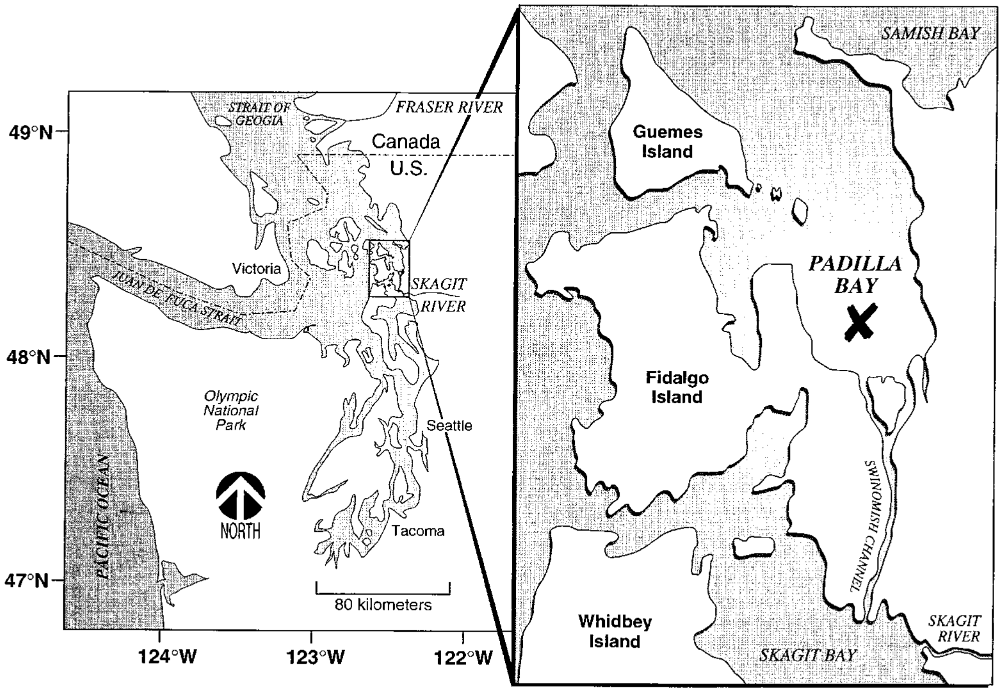
Parameter	*Trinity Bay* (*St.2422.02*) (*n* = *35*)	*Smith Point/Eagle Point* (*St.2439.025*) (*n* = *104*)	*Morgan's Point* (*St.2421.0075*) (*n* = *57*)	*East Bay* (*St.2423.01*) (*n* = *34*)	*West Bay* (*St.2424.01*) (*n* = *87*)
Ortho-phosphate (mg P l^{-1})	0.21 ± 0.001	0.22 ± 0.01	0.42 ± 0.02	0.07 ± 0.008	0.07 ± 0.007
Total phosphate (mg P l^{-1})	0.26 ± 0.02	0.27 ± 0.01	0.51 ± 0.02	0.105 ± 0.013	0.10 ± 0.009
Nitrate (mg N l^{-1})	0.10 ± 0.025	0.08 ± 0.01	0.43 ± 0.05	0.03 ± 0.007	0.04 ± 0.01
Ammonia (mg N l^{-1})	0.06 ± 0.01	0.08 ± 0.01	0.20 ± 0.02	0.08 ± 0.02	0.05 ± 0.01
Nitrate (mg N l^{-1})	—	0.035 ± 0.01	0.13 ± 0.02	—	—
Total Kjeldahl nitrogen (mg N l^{-1})	—	1.4 ± 0.06	—	—	—
Salinity (o%)	9.1 ± 1.2	17.3 ± 0.6	15.3 ± 1.1	15.9 ± 0.8	24.8 ± 0.8
Total suspended solids (mg l^{-1})[a]	15 ± 1.7	20 ± 1.5	27.7 ± 5.7	—	—
Total organic carbon (mg C l^{-1})	—	7.4 ± 0.5	4.4 ± 0.7	—	—
Chlorophyll-*a* (μg l^{-1})	3.8 ± 0.9	8 ± 1	14.6 ± 2.4	5.0 ± 1.4	2.9 ± 0.5

[a] Called 'total residue concentration' in TWC database. The arithmetic averages are for illustrative purposes only, and do not claim true statistical meaning; Morgan's Point has higher salinity due to the fast transport of seawater in the ship channel, and lower freshwater input from the San Jacinto River.

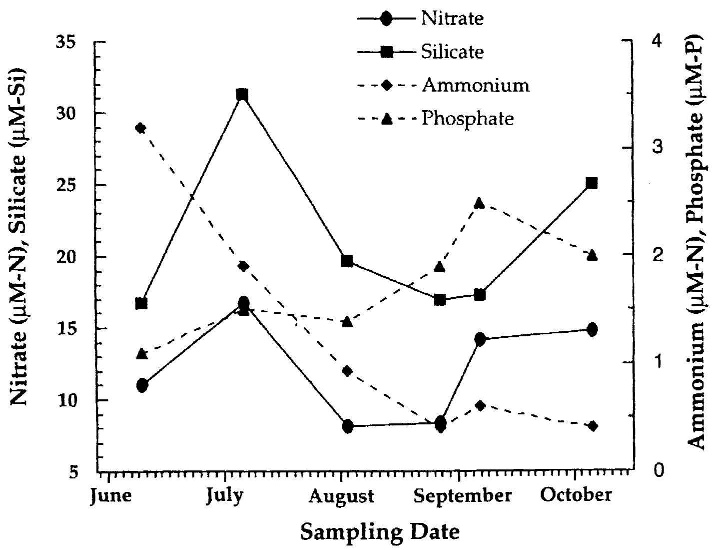
Average Concentrations of Nutrients and Other Chemical Parameters in Galveston Bay, Calculated from the TWC Database (± 1σ).

Puget Sound

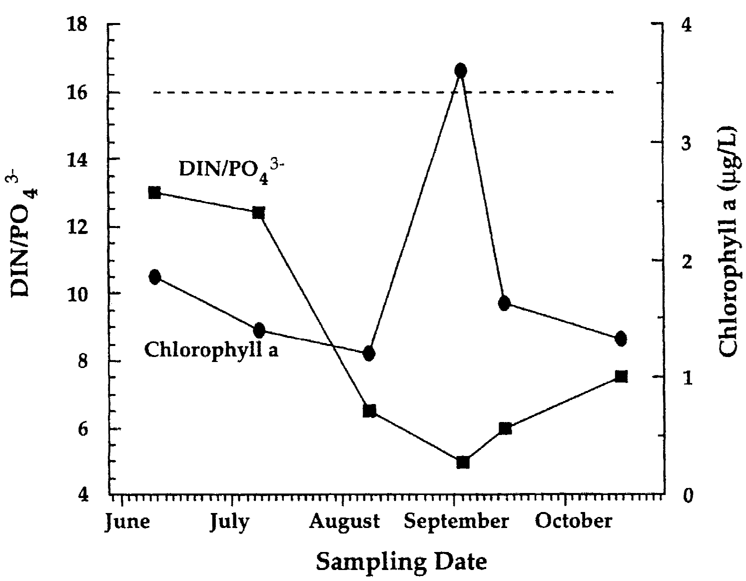
Padilla Bay



Map showing location of Padilla Bay in relation to Puget Sound (left) and the study site (X) in Padilla Bay.



Dissolved inorganic nutrient concentration in surface water at the study site from June to October 1992.



DIN:PO_4^{3-} ratios and chlorophyll a concentrations in surface water at the study site from June to October 1992. DIN is $NO_3^- + NO_2^- + NH_4^+$. PO_4^{3-} is soluble reactive phosphate. The dotted line represents the Redfield ratio of 16:1.

APPENDIX H
PRELIMINARY STATEMENT OF PROPOSED NEAR COASTAL MARINE NUTRIENT SAMPLING AND REFERENCE CONDITION DEVELOPMENT PROCEDURE

Synopsis of the National Nutrient Criteria Program, Coastal Marine Sampling Design Planning Meeting, 4-5 June 2001, USEPA Environmental Science Center, Ft. Meade, MD.

Near Coastal Marine Nutrient Sampling and Reference Condition Development Committee Members: Barry Burgan, USEPA; John Fox, USEPA; Laura Gabanski, USEPA; Jeroen Gerritsen, Tetra Tech Inc.; George Gibson, USEPA*; William Muir, USEPA; Kent Price, Univ. Del.; Don Pryor, NOAA; Greg Smith, GLEC, Inc.; Val Smith, Univ. Kansas; and Jack Word, MEC Analytical Systems, Inc.

*Contact: USEPA Laboratory, 701 Mapes Road, Ft. Meade, Md. 20755-5350. Phone: (410) 305-2618
E-mail: Gibson.George@EPA.gov.

Background and Purpose

Cultural eutrophication is an established water quality management concept and concern reaching as far back as the 1600s in America (Capper, Power and Shivers 1983). However, extensive public recognition of this form of pollution in coastal waters is relatively recent. The publication “Eutrophication, Causes, Consequences, and Correctives” (NAS 1970) is often perceived as the technological beginning of American nutrient pollution awareness and is centered on the understanding and abatement of this problem primarily in freshwater lakes and reservoirs. We have since come to better understand the problem in streams, rivers and estuaries with the publicity and public involvement in the Chesapeake Bay studies of the 1980s. Vollenweider, Marchetti, and Viviani published “Marine Coastal Eutrophication” in March of 1992 and this volume may be considered the coastal equivalent of the land mark NAS freshwater publication a decade earlier.

In response to this growing awareness, the EPA National Nutrient Criteria Program is preparing technical guidance for nutrient reference condition determination and related criteria development to be used by States and Tribes in the reduction of cultural eutrophication of the Nations’ surface waters. This report concentrates on coastal marine waters and the effort to identify relatively natural nutrient water quality conditions, which can be used as a benchmark to evaluate cultural eutrophication or overenrichment. A preliminary literature investigation and data search indicate that insufficient data exist to derive the reference condition information suitable for the needs of the Program without going to primary data collection.

Because data gathering is likely to be a preliminary concern as well as an ongoing requirement, this meeting of coastal marine nutrient research and management specialists was called to design a standard protocol that the EPA can recommend for use in U.S. marine coastal waters. Coastal marine waters are defined as those waters within 20 miles of shore along the East, West, and Gulf coasts of the United States as well as Alaska, Hawaii, and the U.S. Trust Territories. Emphasis is on the three mile limit State waters, although interest may devolve to the 12- mile U.S. limit as well. Nutrient loading from cultural land run off sources are not presently expected to be a serious problem beyond this limit. The general design and protocol are applied here to a case study example, the coastal waters of the mid-Atlantic Bight from New Jersey to the Virginia Capes. Many of the design elements, in particular the number, size and placement of spatial elements (strata and cells; see below) would need to be modified for specific applications in different parts of the U.S. coastline.

A coastal transect of fixed stations exists for most of the Mid-Atlantic Bight, which has been used by EPA and NOAA for several years to collect nitrogen, phosphorus, Secchi depth (SD), and chlorophyll-*a* (Chl-*a*) data. This procedure and data base will be the prototype presented and discussed to develop the recommended protocol.

Objective

To determine a simple, cost effective, scientifically defensible and standardized method to sample for marine enrichment variables to use in determining reference condition for nutrient criteria derivation.

Premise of the National Coastal Nutrient Criteria Program

Offshore marine and onshore, near-coastal sites removed from point and estuarine discharges can be identified as reference sites reflecting the least culturally impacted nutrient water quality of a region. "Region" in this case is a geographically similar portion of the coastline such as the Mid-Atlantic Bight. Such regions can also be described from the coastal portion of the Level III nutrient ecoregion map of the continental United States (which is consistent with the rest of the National Program and is similar to the ORD Provinces used by EMAP).

Nutrient water quality is established from representative sampling of the coastal waters at these a priori reference sites. The other elements of nutrient criteria, i.e., historical trends, modeling of the data for additional insights, and attention to the consequences down-current of any proposed nutrient criteria, and assessment of all of this information by a Regional Technical Assistance Group (RTAG) are applied to the initial reference condition values to develop nutrient criteria for total phosphorus (TP), total nitrogen (TN), Chl-*a*, and SD.

These criteria can then be used by States and Tribes to manage and monitor the nutrient quality of their coastal marine waters. While this concept was developed and initiated by the USEPA beginning with the Biological Criteria Program in 1989 (EPA-440/5-90-004, EPA-440/5-91-005) and further refined and applied to nutrients by the EPA National Nutrient Criteria Program in 1995 (EPA 822-R-96-004, EPA 822-R-98-002), the idea has also been independently developed by the Swedish Environmental Protection Agency using the reference condition approach on a regional basis and employing the same indicator variables (Report number 5052, 2000).

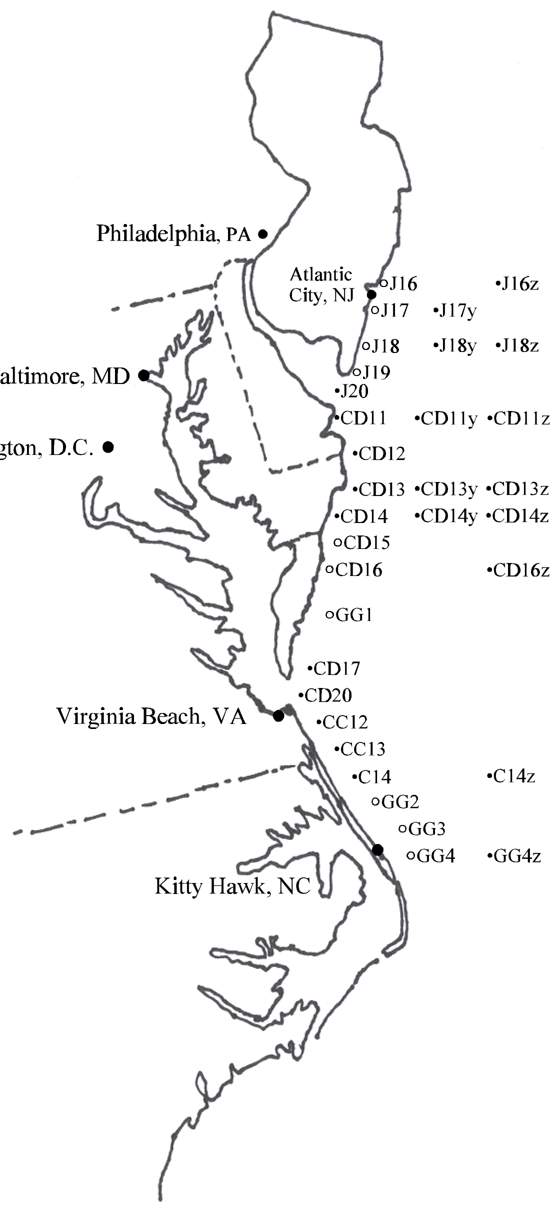
Importance

All other waters of the continent drain to the coast and these coastal marine areas are the recipient of any nutrients not intercepted from that cumulative runoff. Globally, conditions of many coastal areas have shown several-fold increased levels of nitrogen and phosphorus since industrialization (Smith 1998, Smith, Tilman and Nekola 1999), and preliminary assessments of empirical data collected from the Mid-Atlantic Bight between New Jersey and North Carolina by USEPA Region III since 1987 have suggested an upward trend in the concentration of both dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) at stations approximately 1 to 5 nm off shore (Muir, pers com, 2001). An extensive baseline of region specific coastal nutrient data, regularly and consistently collected, is needed to establish criteria against which future loading conditions may be compared. Potential predictive models may also be developed from this information relating algal booms and other biological responses to these nutrient levels in coastal waters.

The Transect Sampling Design

As illustrated below (Figure H-1, of coastal sampling stations between Atlantic City, NJ and Kitty Hawk, NC) a series of transect sampling stations have been located between one and five miles from shore along a designated coast line to measure ambient water quality as reflected in TN, TP, Chl-*a* and SD. Data collected from those stations determined to be remote from significant cultural impacts such as sewage

Figure H-1. Example of an initial coastal monitoring project conducted off the mid-Atlantic shelf just north of Delaware Bay and south of the mouth of the Chesapeake Bay to obtain nutrient reference condition values. Open circles are reference sites and the inshore references are compared to the offshore ones for confirmation.



discharges, industrial activities, major port facilities, or estuarine discharges constitute the reference condition. The values measured can be compared to marine conditions further offshore (e.g., 20-25 nm) reflecting, at least for TN and TP, the unimpacted condition.

This comparison, together with attention to prevailing currents including upwellings, would establish the seasonal values for TN, TP, Chl-*a*, and SD. Salinity would also be an important variable to document local constancy of the waters and to avoid discharge plumes from rivers and estuaries that are not part of the defined coastal waters of concern. These estuarine and riverine waters should have their own criteria.

Data have been collected from the surface 1 meter, mid depth, and the bottom 1 meter of the water column at each station. Measurements are made on a seasonal basis, essentially mid-summer and mid-winter and with sufficient data, criteria can be established for each season. While the protocol as described above has been in use for about twenty years, most of those data are for dissolved inorganic nitrogen and phosphorus rather than TN and TP, Chl-*a*, and SD. The TN, TP and off shore stations are a recent addition and only two summers of data are available which is reported here.

Data Collection
Operations are during daylight hours only in order to include SD measurements with surface nutrient collections and to maintain a consistent nutrient depth profile relative to photo periodicity. Additional data collected with a CTD includes salinity, pH, temperature, depth, conductivity, and DO.

Data sampling points presently include the discharge plumes of estuaries, rivers, and point dischargers to monitor impacts and design management plans, but not as part of the proposed coastal reference condition sampling system per se.

The Pilot Project
The area approximately between Atlantic City, NJ and Kitty Hawk, NC has been studied by EPA Region III for about 17 years and includes a mix of nutrient, chemical, biological, and physical measurements. This data has been processed and will provide trend information about the area.

For the last two summers and one winter, nutrient data have been collected from this area in the manner described above. This is the initial basis for a reference condition determination and is presented in Figures H-2 a-d, below.

If the pilot project is judged successful, it is expected that the process, training, and funds for similar equipment will be provided to each of the coastal EPA Regions for comparable operations to develop their ecoregional coastal nutrient criteria. This area more than any other because of the proximity of State and Federal waters will lend itself to joint data gathering and criteria development.

Methods and Materials
Forty-nine sampling stations are located along the 200 nautical mile (nm) stretch of coastal waters. There are twenty stations roughly 10 miles apart situated one to five miles off shore (total of 39 consisting of either single stations or sets of two or three), there are five intermediate (ten miles off shore) stations, and there are eight stations located about 20 nm offshore (Figure H-1).

Sampling was conducted from the OSV Peter W. Anderson using a Sea-Bird brand CTD and rosette sampler with 30 L Niskin bottles to produce a continuous water column profile and discrete water samples from the surface one meter, the mid-depth, and the bottom one meter of the water column at each station. One liter of sample was filtered using a Millipore Corporation apparatus and 0.7 um fiberglass filters (Whatman GF/F). Ten ml sub-samples each of water were taken for TN and TP analysis using the

Figure H-2a. Two-year summer nutrient survey results using a sampling design as illustrated in Figure 1. Potential reference condition for summer conditions is 0.025 mg/L TP (NM = nautical miles).

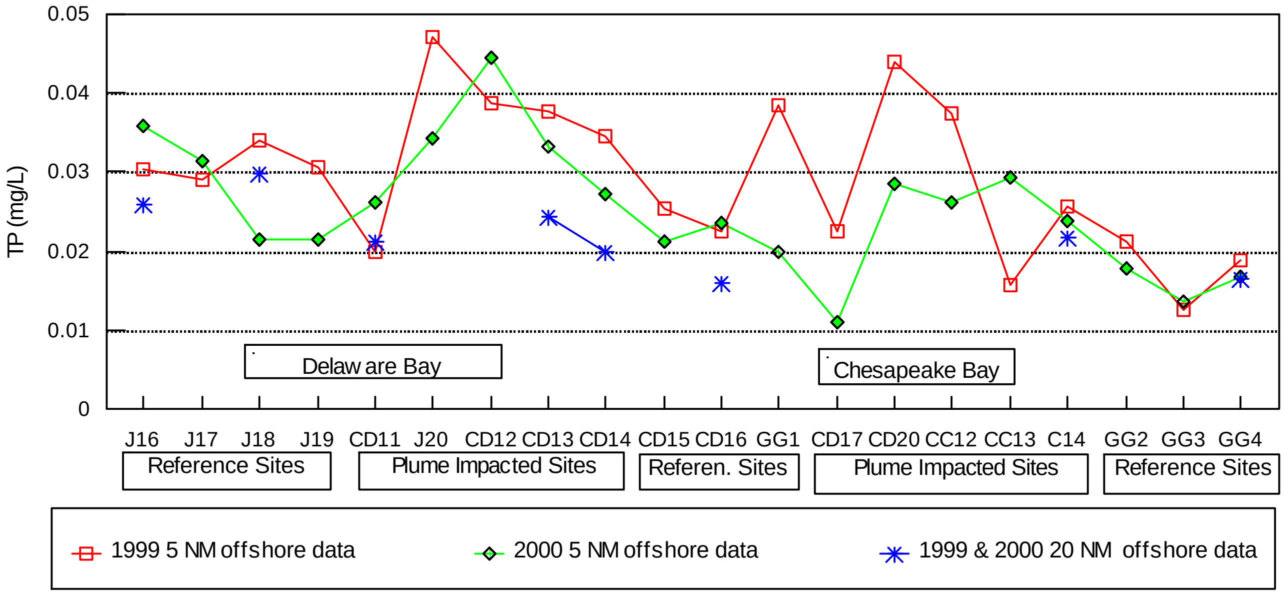


Figure H-2b. **Two-year summer nutrient survey results using a sampling design as illustrated in Figure 1. Potential reference condition for summer conditions is 0.175 mg/L TN (NM = nautical miles).**

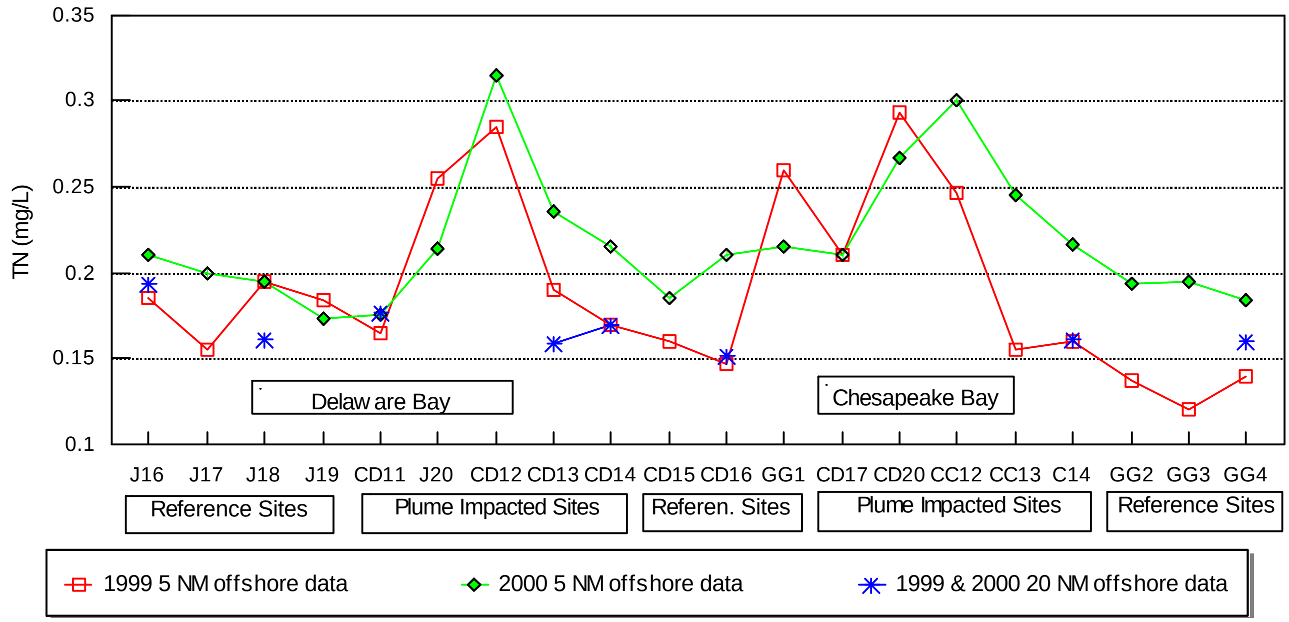


Figure H-2c. **Two-year summer nutrient survey results using a sampling design as illustrated in Figure 1. Potential reference condition for summer conditions is 0.09 µg/L chlorophyll-a (NM = nautical miles).**

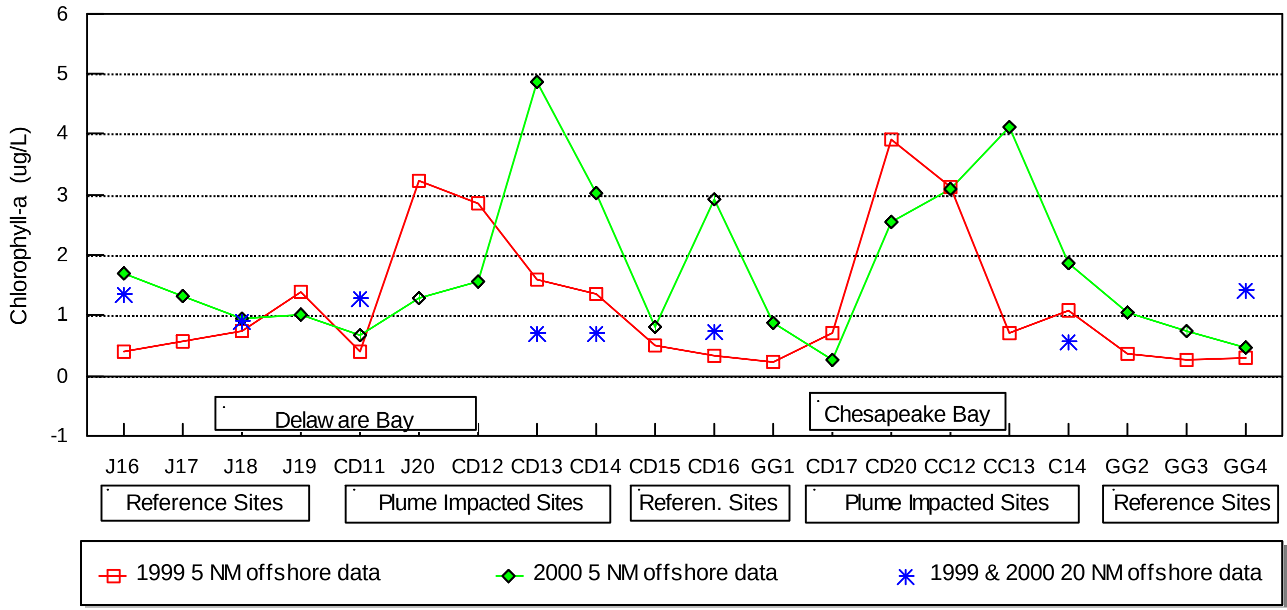
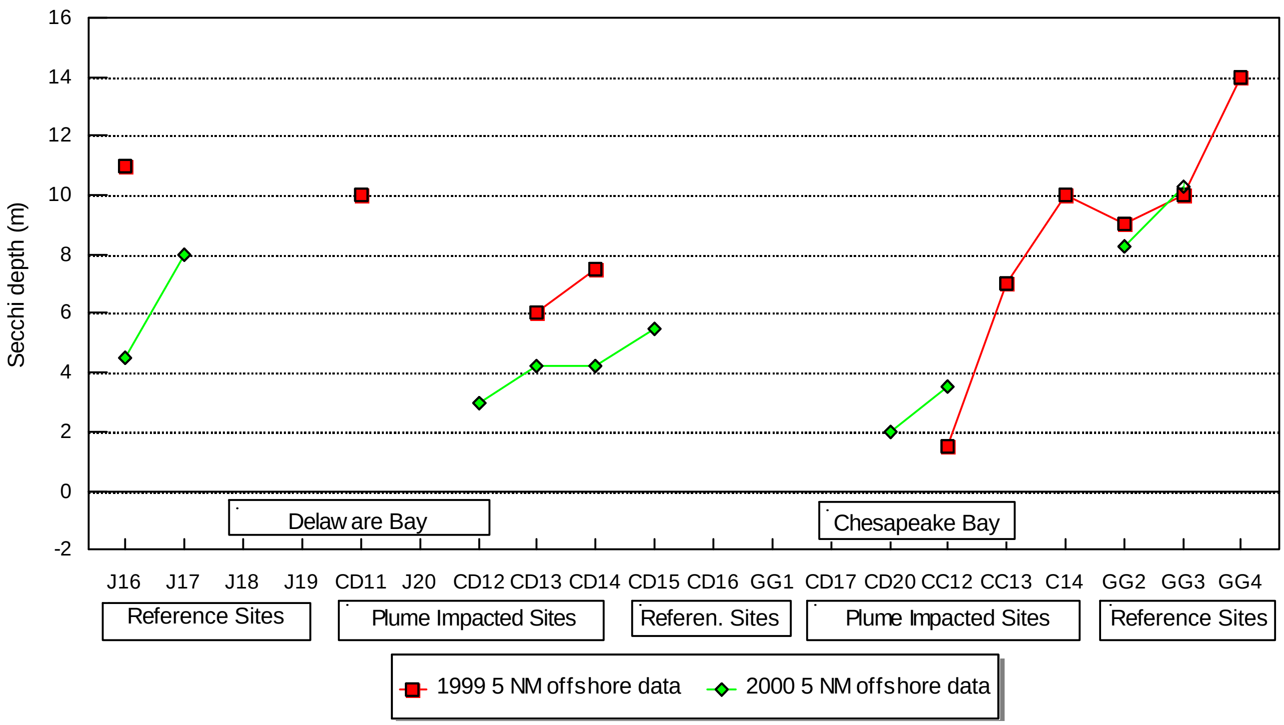


Figure H-2d. **Two-year summer nutrient survey results using a sampling design as illustrated in Figure 1. Secchi depth data are incomplete because of missing observations during night-time operations (NM = nautical miles).**



Standard Methods persulfate digestion method. All samples were frozen on board for later analysis at the University of Maryland Chesapeake Biological Laboratory. Secchi depth was determined using a 35 cm diameter white Secchi disc on 0.5 m marked line.

Results

Preliminary data analyses indicated that the groupings of two or three stations (originally intended to reflect progressive off shore encroachment of nutrient runoff) are not significantly different. Similarly, initial assessment of the surface, mid-depth, and bottom water column samples showed no significant differences for the nutrient criteria indicators except for DO. Salinity did show the expected surface to bottom variations expected of near coastal waters. Consequently horizontal and vertical sampling distances for each station were combined to produce the nutrient results presented here.

The following figures illustrate the results of two summer surveys in 1999 and 2000. Data indicated by an asterisk represents the mean of combined two year data from each 20 nm off shore station. Gaps in the Secchi depth results are because of the night time intervals during 24-hour operations. The stations located in the vicinity of the estuarine discharges of Delaware and Chesapeake Bays are indicated by the appropriately labeled open boxes, i.e. J18 through CD12 for the Delaware and CD17 through CC13 for the Chesapeake.

Discussion

- The survey technique appears to faithfully reflect the nutrient conditions of the coastline in that higher nutrients were found in the waters just off each estuary. Further, when surface station salinity was plotted, a mirror image of the nutrient data results was presented indicating a freshwater correlation with higher nutrient concentrations. The biological response of the waters was also evident from the correlation of Chl-*a* and SD data with the TN and TP levels.

- Given the variability in only two years of data, it is interesting that the offshore stations appear to be relatively consistent in all three nutrient parameters. The incongruity of the TP and TN data at GG1 for 1999 and for chlorophyll-*a* at CD16 for 2000 is also noted, but unexplained.

- The two consecutive sampling years demonstrated similar trends among the stations indicating no major weather variability during this period and also suggesting that the proposed reference sites respond comparably to the discharge sites to inter-annual climatological events.

- The stations that consist of sets of two or three sites within a few miles of one another were determined upon review of the data to have very little distinction. This suggests that perturbation, if any, originating from the coastal land mass has had an impact over all three sites.

- It appears that the Delaware plume drifts more southward from the mouth of its bay than does the Chesapeake. This may reflect the lower discharge volume of the Delaware and the influence of the Hudson River discharge and Longshore current, both displacing the plume southward. Similarly, the slight northward offset of both the Delaware and Chesapeake plumes in 1999 relative to 2000 may be a response to changing coastal current dynamics from year to year.

- There was no significant difference between the eight offshore sites (20 nm offshore) and their inshore counterparts. Further comparison of offshore stations to those counterparts within the estuarine discharge plumes will help determine the sensitivity of the comparison. The intermediate 10 nm stations do not appear to add substantial information and can be discontinued except in the vicinity of discharge plumes to help define these margins.

While the data is limited, it is encouraging that the offshore controls are comparable to the expected inshore reference sites. Candidate inshore reference sites are initially selected on the basis of an apparent physical absence of local cultural impact, i.e. tributary discharges, municipal discharges, ports or marinas, or other commercial enterprises. Because of the potentially high variability in the existing data, a concurrence between offshore and inshore reference stations is not necessarily confirmation of the quality of the inshore reference sites, but when consistent with observed physical indications, this information adds confidence to the selection. Conversely, significant differences would be cause for suspecting the inshore site selection as of reference quality.

- In this regard, an interesting trend was noted among the three groups of reference sites located north of Delaware Bay; between Delaware and Chesapeake Bay; and south of Chesapeake Bay (Figures 2a-d). The mean ambient concentrations of TP, TN and Chlorophyll-*a* at these reference sites trend downward from north to south. The same trend appears in the eight stations 20 nm offshore presumably reflecting a broad scale process affecting this area. This further supports the importance of establishing relatively close spaced reference sites when preparing coastal marine criteria The mean TP values for the nearshore reference sites at the northern terminus of the transect of stations is significantly higher than those at the southern end ($p = 0.0006$) even though the region is presumed to be geologically homogeneous.

- Secchi depth data were inconclusive because not enough data points were generated as a consequence of 24-hour sampling when Secchi depth could not be determined during night-time hours. Additional future sampling will be conducted during daylight so all parameters may be evaluated.

Committee Discussion of the Prototype Methodology

Fixed Station Sampling vs Stratified Random Sampling

Inferences derived from fixed-station and fixed-transect sampling, while common in oceanographic research and monitoring, are potentially confounded by unintentional and unknown biases, and by the inability to extend statistical inferences to the entire sample space desired. Alternatively, fixed stations tend to reduce the amount of unknown physical variability associated with interpreting climatic factors upon a given site such as when attempting to assess hurricanes, upwelling or acid deposition effects on a particular coastline. Although there is no reason to suspect that the existing stations of the mid-Atlantic coastal nutrient study are biased, the design group thought that the design should allow data inference to the entire coastal sampling space. It therefore proposes a change to a probability-based design, of equal sampling cost, to avoid the potential pitfalls of a fixed-station design.

The sample space for this project is open marine waters of the U.S. coastal zone, with emphasis on state waters within the 3 nautical mile state limit. Sampling will be carried out in three sampling strata for which nutrient conditions are to be estimated:

1. Reference areas within the 3-mile limit, outside the influence of major estuary plumes (e.g., Hudson River, Delaware Bay, Chesapeake Bay);

2. Nutrient influenced areas within the 3-mile limit, affected by the estuary plumes and other discharges; and

3. Offshore waters beyond the state 3-mile limit.

These three regions will define the sampling strata. The sampling design will be to define longshore "cells" in each of these three zones. During each sampling event, one site will be selected randomly within each sampling cell. The overall design can be described as stratified-systematic-random; where the strata are the three areas defined by estuary influence and distance from shore, the systematic component is the cells that define each stratum, and the random component is the random sampling location selected on each cell during each sampling event (Figure H-3).

The first task in developing the sampling plan will be to define areas of presumed estuarine influence, from existing physical oceanographic research on water movement, in this case in the mid-Atlantic coastal area, and from existing water chemistry data showing elevated nutrients and other constituents such as salinity and conductivity in the estuary plumes. Because plumes will vary with estuarine discharge, Gulf Stream eddies, and other events, precise definition of the plume area is not possible. Instead, plume areas should be defined as where the estuarine influence is likely to occur. An understanding of local estuarine hydrology will help in this regard, but extensive and expensive physical investigations should not be a prerequisite for the determination of likely plume influence.

Coastal reference site determinations and reference condition derivation should be, at least initially, established by EPA. Data collection would be continued as an EPA function because the offshore stations are the purview of the Federal government and because the related reference sites can be incorporated in the effort.

Estuarine and riverine plume monitoring are more likely accomplished by the coastal States using existing budgets and vessels already at their disposal. A coordinated effort relating State and Federal sampling and data exchange should be promoted. The nutrient quality of the discharge plumes can then be expeditiously compared to the proximal reference condition(s) to assess impact upon the near coastal marine waters.

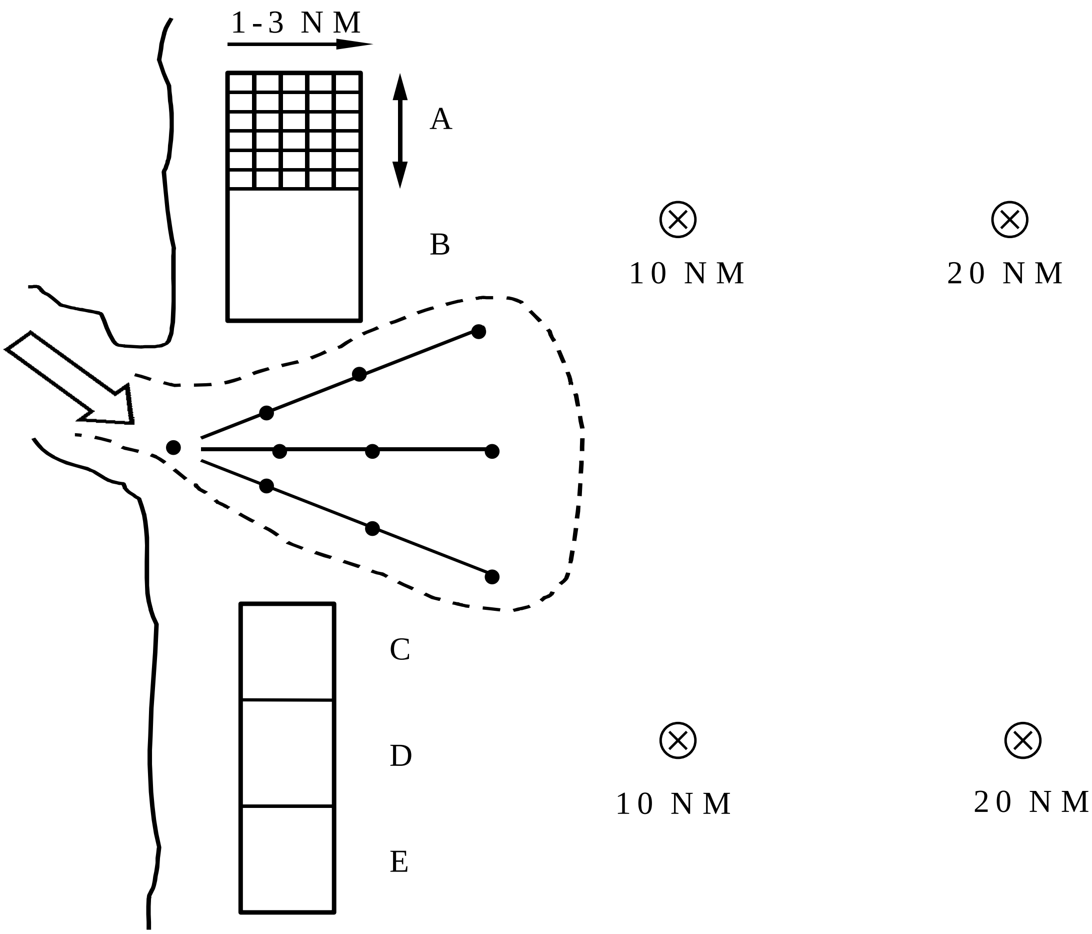
Regional coastal characteristics will determine both the reference site cell structures and placement and the estuarine or riverine plume sampling designs.

Sampling Times and Depths
The committee recommended that sampling be conducted during the period of optimal marine vegetative growth. In temperate areas this is Spring and Summer, generally May, June and July, August. Other interval options depending on locale might be wet and dry periods during the growing season.

As a cost effective approach producing scientifically valid results, sampling depth is recommended as always at the surface, i.e., top meter of water accompanied by either a composite sample from the remainder of the water column or sampling from just below the thermocline and at one meter above the bottom. Some members of the committee advocate the surface, mid-depth, and bottom sampling technique where the mid-depth sample is usually below the thermocline and/or where inshore waters often fail to demonstrate a thermocline.

Variables
The primary four variables of TN, TP, Chlorophyll-*a*, and SD are recommended because they are the best early indicators of causal and biological response indicators to nutrient loadings. Other measures of clarity or transparency may also be used, but Secchi depth should always be included because such a large body of information is already available in this form; it is inexpensive and reliable; and continued Secchi depth measurements provide a continuity with much historical data. In an independent

Figure H-3. Illustration of stratified random grid sample design for reference condition cells as related to an estuarine discharge. Nutrient quality and spatial extent of the grid can be compared to a value derived from the measurements in reference sites (cells) A-E. Values at 10 and 20 nm stations are a check against reference values found at A-E.



investigation, the Swedish Environmental Protection Agency selected the same four primary variables (Report 5052, 2000).

Other recommended variables are: dissolved oxygen as an important secondary response variable almost always measured by investigators because of the significance of respiration to the biological community, and planktonic species composition as a refined diagnostic indicator of the nature and extent of enrichment.

Geographic Application of the Protocol

Members of the committee are familiar with both the East, West, and Gulf coasts of the continental U.S. and conclude that the method described above, with allowances for regional modifications such as relative distance from shore to shelf break and the magnitude of upwelling, can be successfully applied in all three coastal environments to identify reference conditions for criteria development.

Summary Conclusions

1. The basic protocol as described, but modified to include a probability-based sampling design, a variation on the surface, mid-depth, and bottom sampling profile, and sampling emphasis on twice during the growing season, is a scientifically defensible and broadly applicable method for establishing regional reference conditions to support coastal marine nutrient criteria development.

2. The TP, TN, Chlorophyll-*a*, and perhaps Secchi depth variables are responsive and together with salinity measurements are descriptive of both estuarine discharge plumes and near coastal reference quality waters. These measurements can be used to assess the concentration and area changes of discharge plumes over time.

3. The comparison of marine nutrient water quality to inshore reference sites is valuable as confirmation of “natural” reference conditions and as a graphic descriptor of the reference concept for the public. However, for cost effectiveness, these off shore stations need not be monitored every time the inshore reference sites are sampled.

4. But it is important to note that the distinction of cultural from inherent nutrient discharges by estuaries and other tributaries to coastal waters is not possible just by comparison to reference conditions. The other elements of nutrient criteria development should be incorporated to help make this distinction. The reference conditions and criteria will reveal exceedences of the “natural” background levels to be preferred and the relative extent and magnitude of the problem, but source identification and cause and effect studies will be required for an effective management response to this identified concern.

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