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EPA

Nutrient Criteria Technical Guidance Manual

Estuarine and Coastal Marine Waters

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.

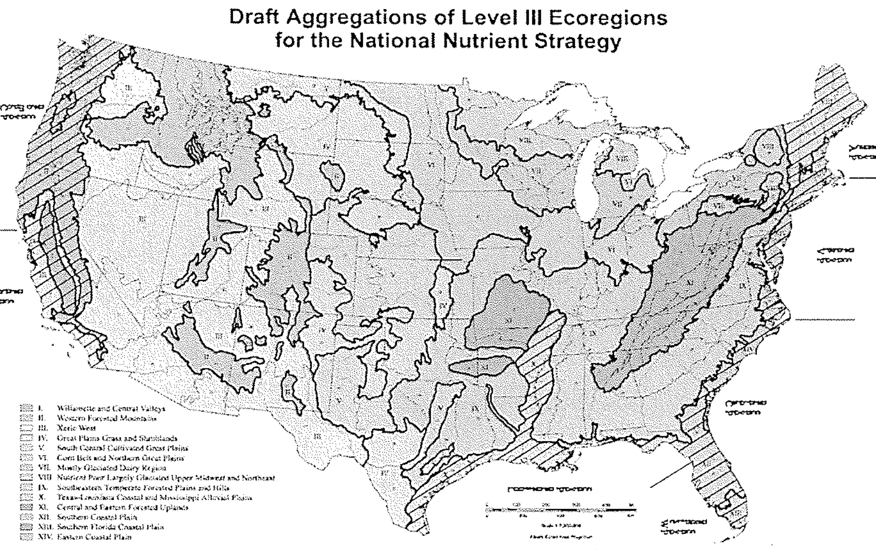

Figure 1-1a. Draft aggregation of Level III ecoregions for the National Nutrient Strategy illustrating those areas most related to coastal and estuarine criteria development.

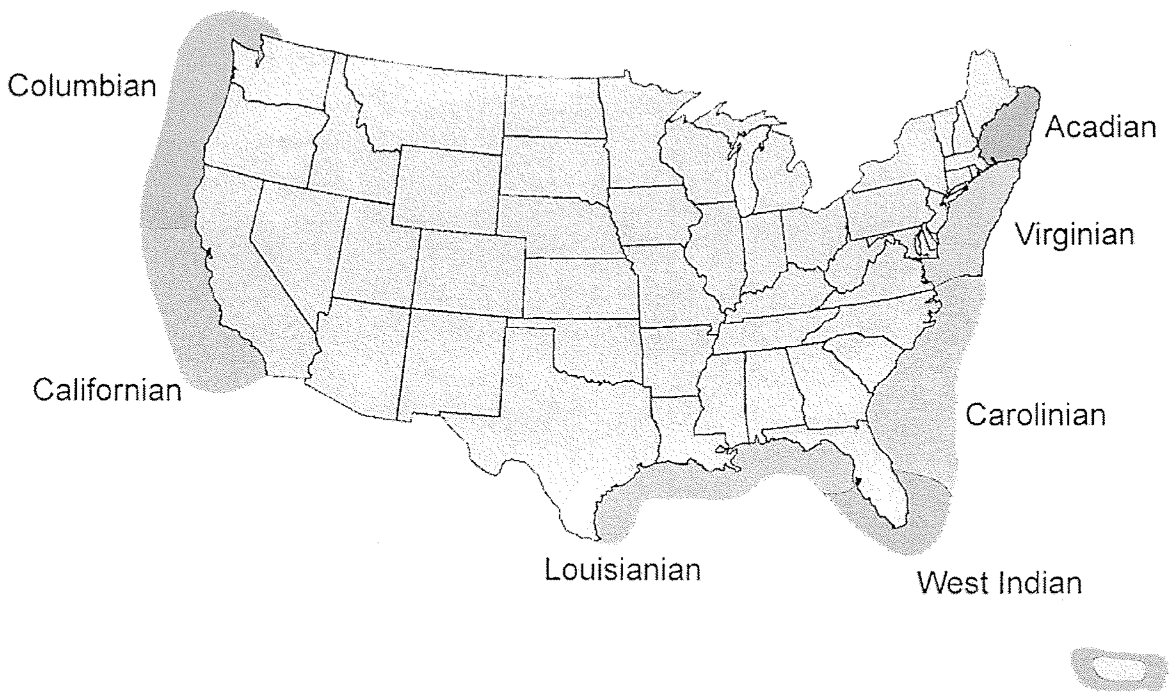

Figure 1-1b. Coastal provinces.

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.

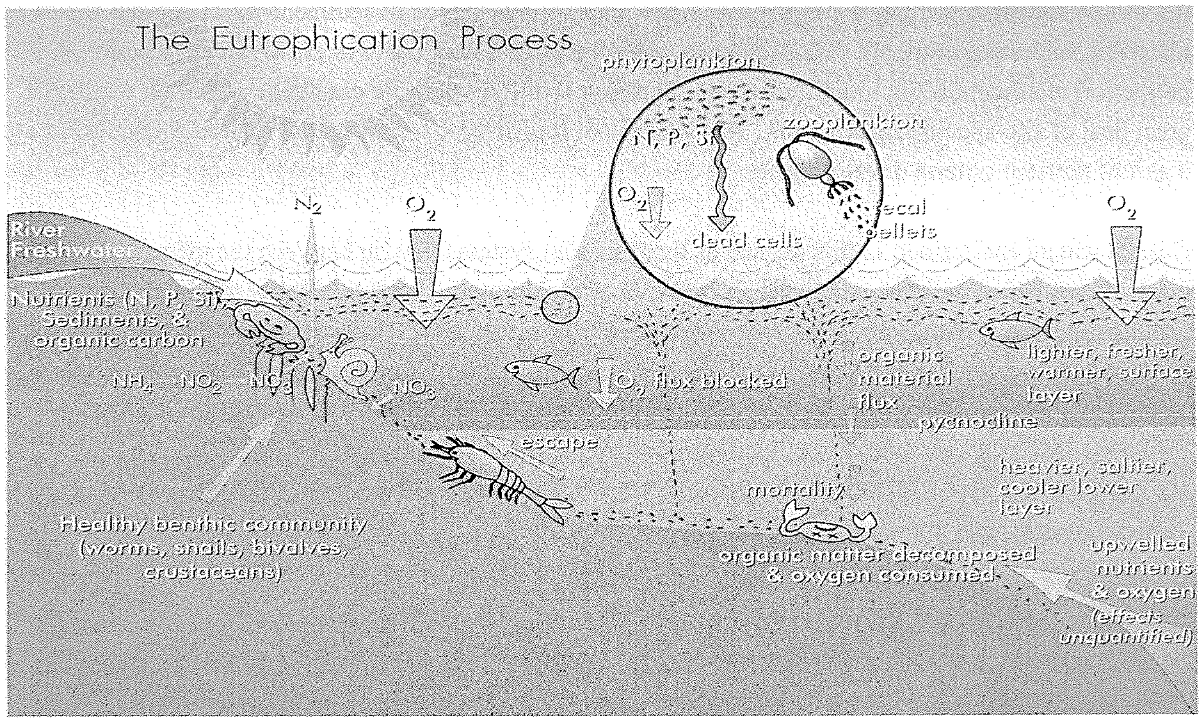

Figure 1-2. The eutrophication process. Eutrophication occurs when organic matter increases in an ecosystem. Eutrophication can lead to hypoxia when decaying organic matter on the seafloor depletes oxygen, and the replenishment of the oxygen is blocked by stratification. The flux of organic matter to 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

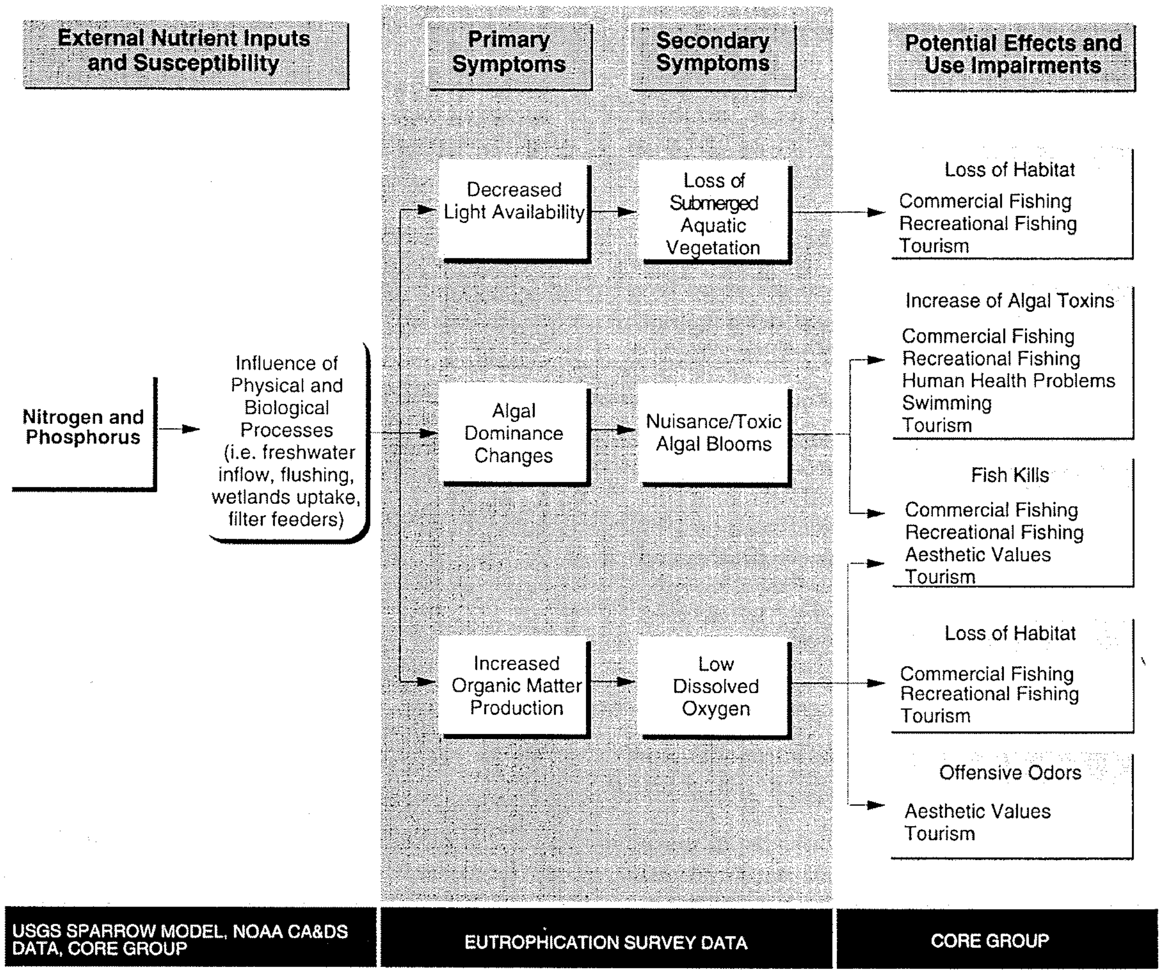

Figure 1-3. Expanded nutrient enrichment model. Source: Bricker et al. 1999.

(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,

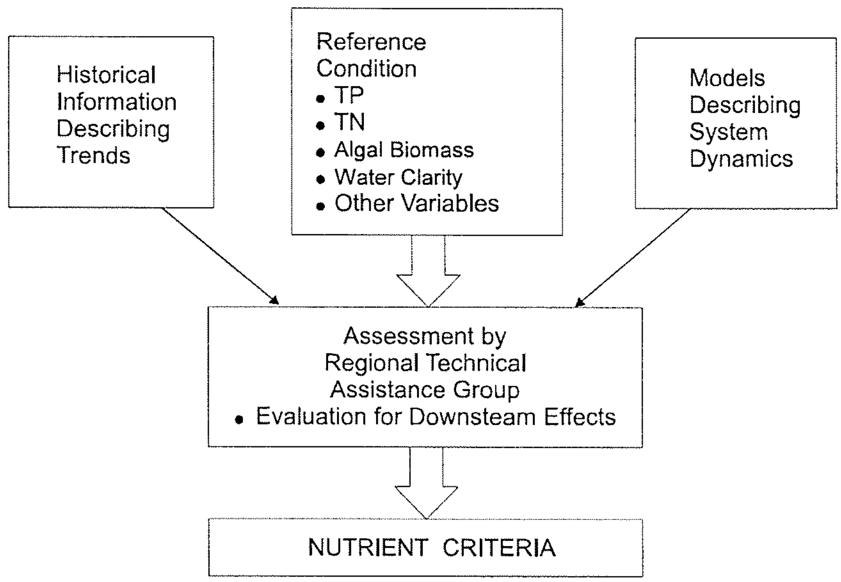

Figure 1-4. Elements of nutrient criteria development and their relationships in the process.

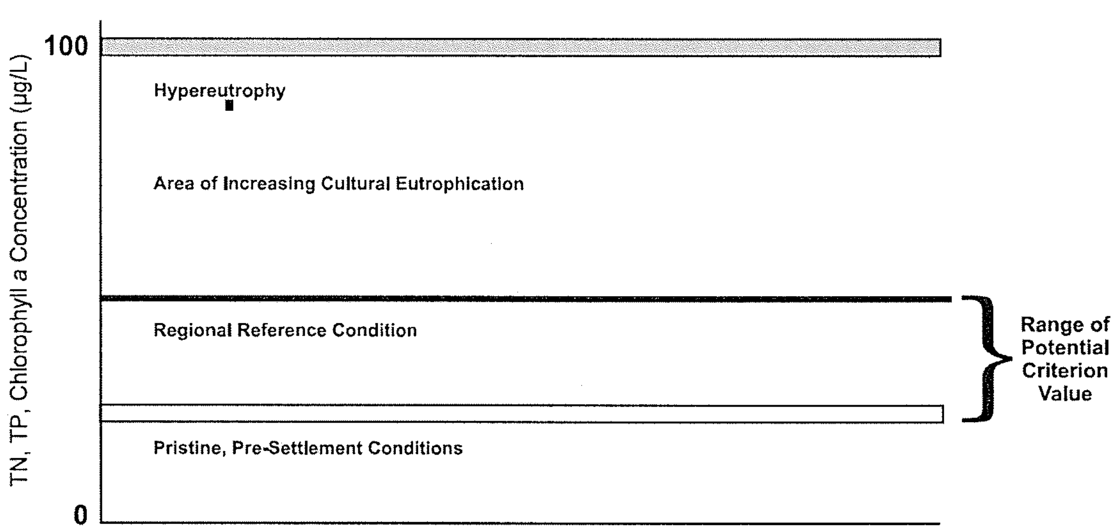

Figure 1-5. Derivation of the reference condition and the National Nutrient Criteria Program using TP, TN, and chlorophyll *a* as example variables. Clarity or Secchi depth would be on a reversed scale. Protectivity nutrient criteria should be between pristine conditions and present reference conditions,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

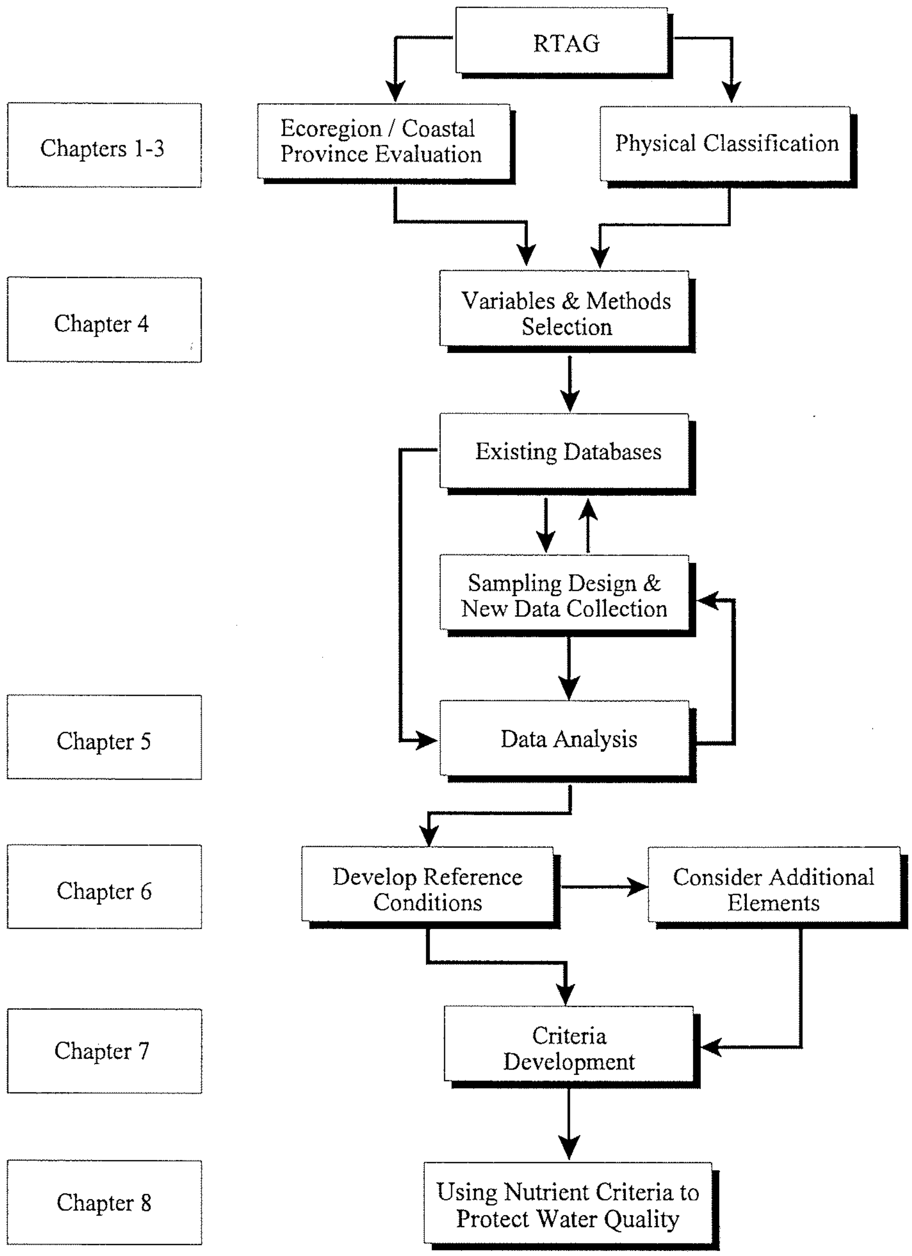

Figure 1-6. Flowchart of the nutrient criteria development process.

■ ***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

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.

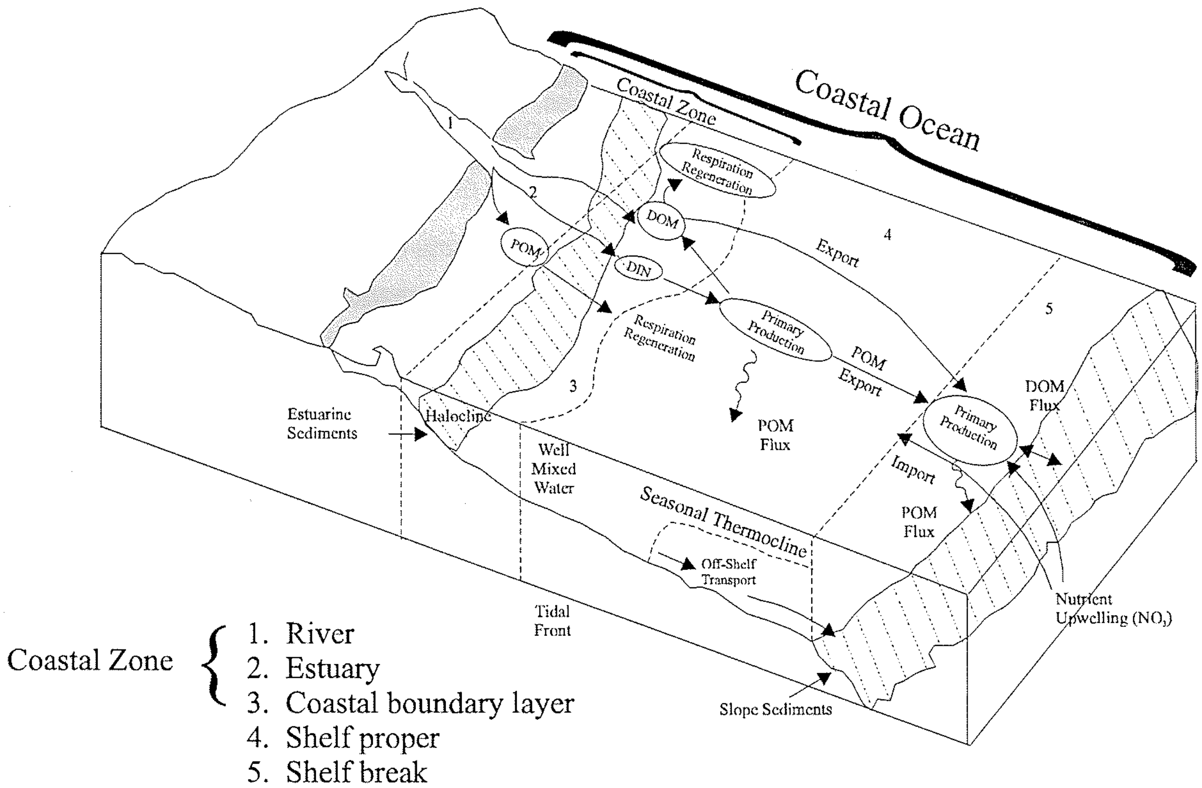

Figure 2-1. Idealized scheme defining the coastal ocean and the coastal zone, with some key biochemical fluxes linking land and sea and pelagic and benthic processes. The latter are not to scale. Source: Alongi 1998.

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

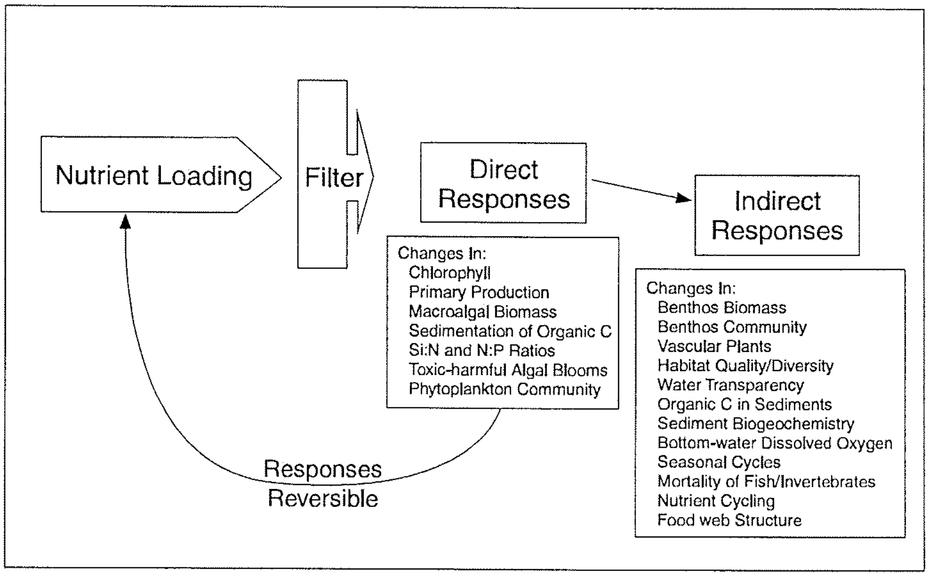

Figure 2-2. Schematic representation of the contemporary (Phase II) conceptual model of coastal eutrophication. Advances in recent decades include explicit recognition of (1) a complex suite of both direct and indirect responses to change in nutrient inputs; (2) system attributes that act as a filter 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).

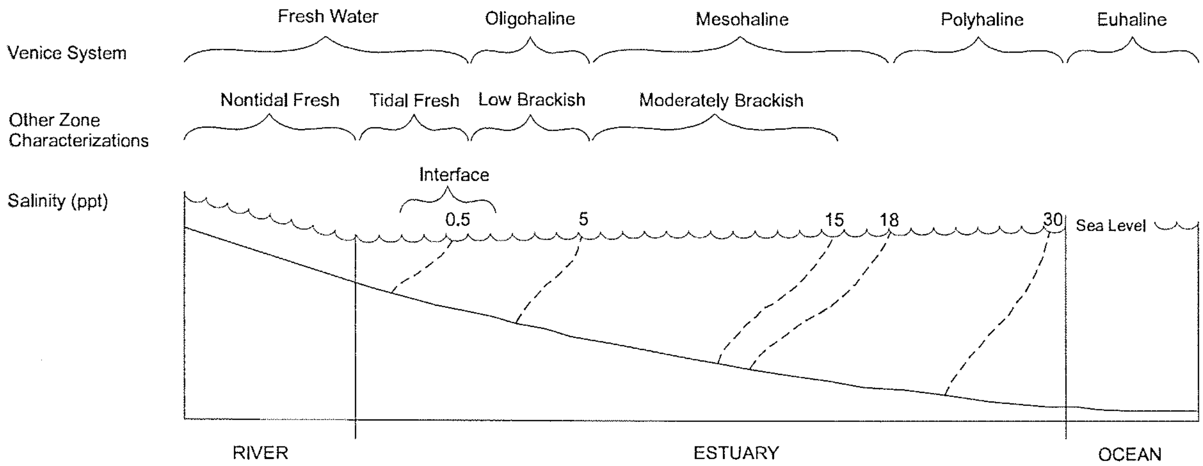

Figure 2-3. Salinity zones. The Venice System is a well-accepted method of characterizing salinity zones and covers the salinity ranges from riverine regions to the ocean. The freshwater category in the Venice System has been modified in this atlas to account for the tidal and nontidal regions found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

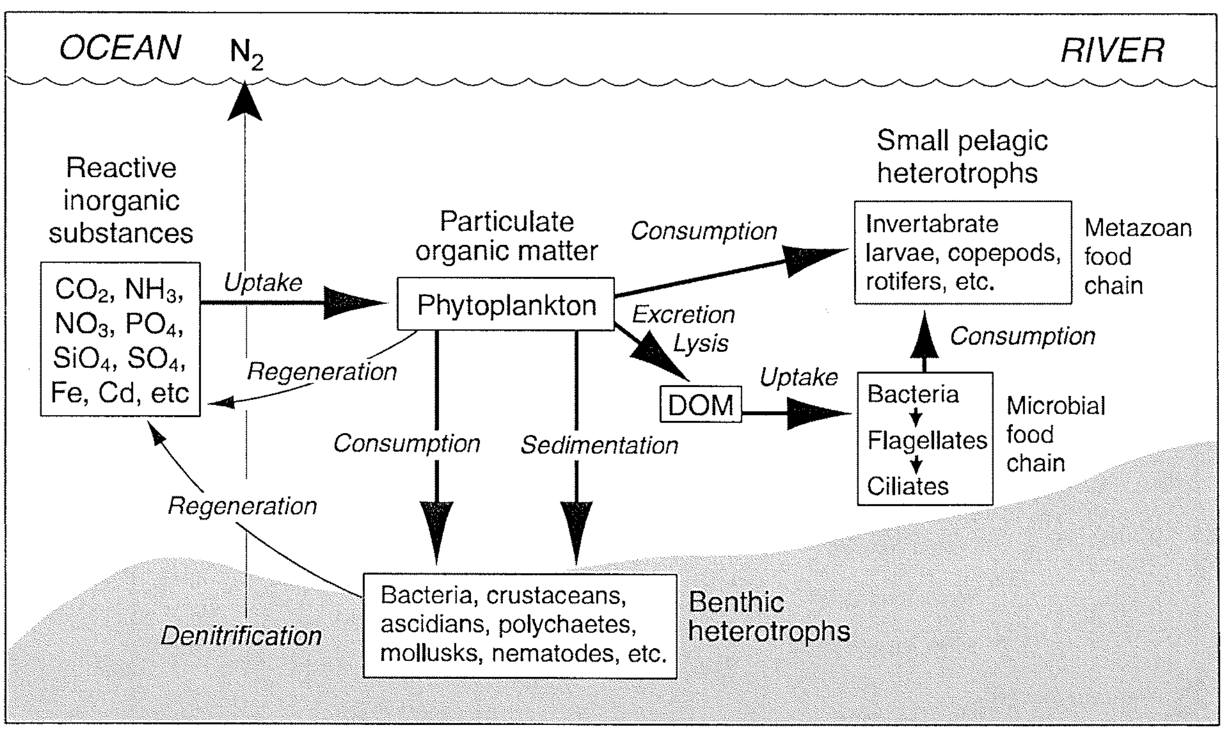

Figure 2-4. Schematic illustrating the central role of phytoplankton as agents of biogeochemical change in shallow coastal ecosystems. Phytoplankton assimilate reactive inorganic substances and incorporate these into particulate (POM) and dissolved organic matter (DOM) which support the production of 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.

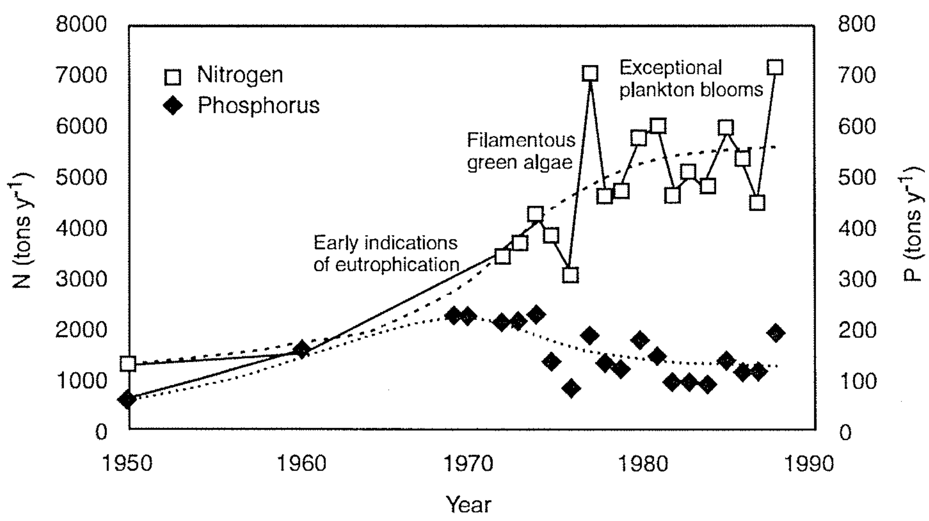

Figure 2-5. Transport of nutrients to Laholm Bay, Sweden. Periods of significant changes in the marine biota are also indicated (modified from Rosenberg et al. 1990). Source: NRC 2000.

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,

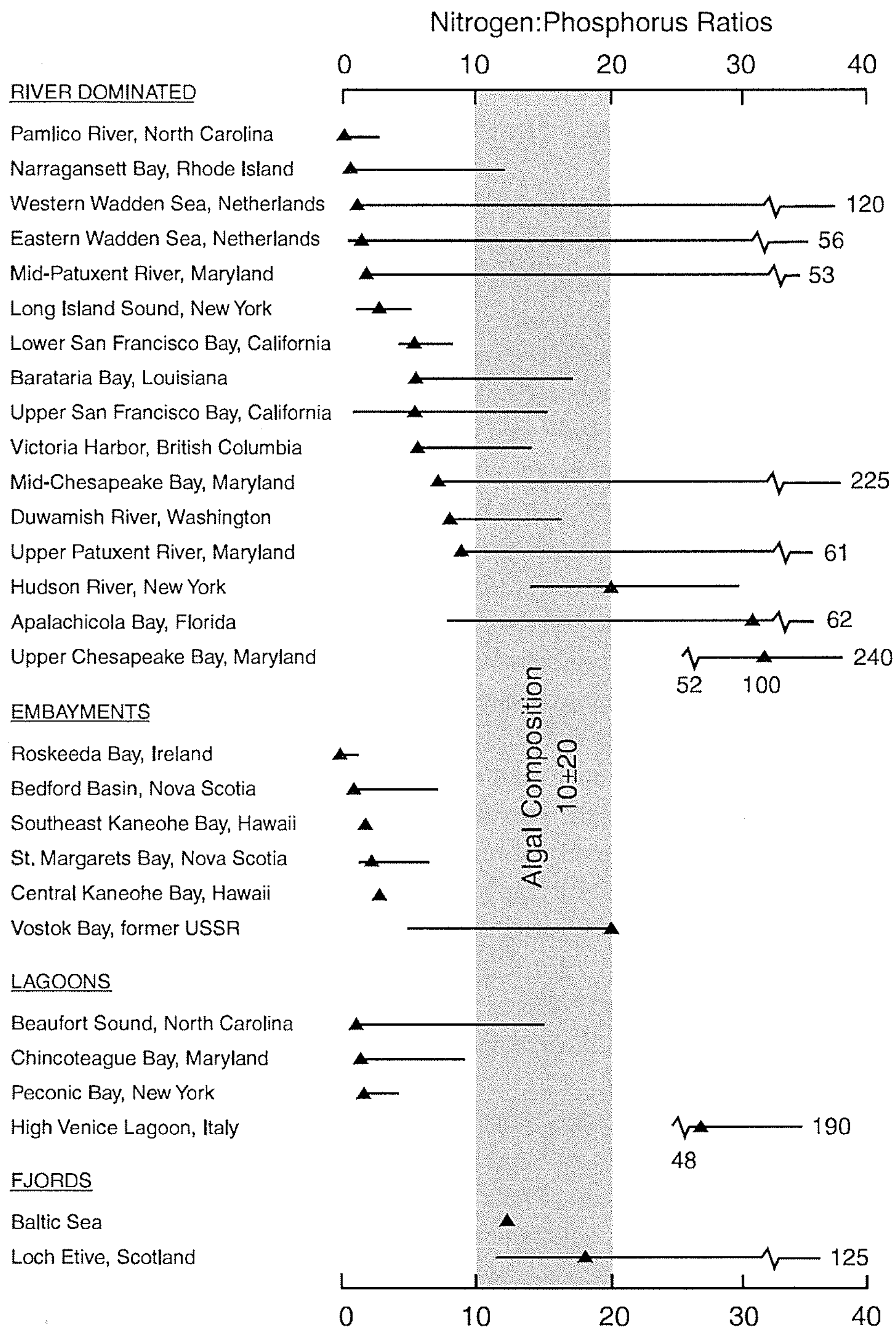

Figure 2-6. Summary of nitrogen:phosphorus ratios in 28 sample estuarine ecosystems. Horizontal bars indicate the annual ranges in nitrogen:phosphorus ratios; solid triangles represent the ratio at the time of maximum productivity. Vertical bands represent the typical range of algal composition ratios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

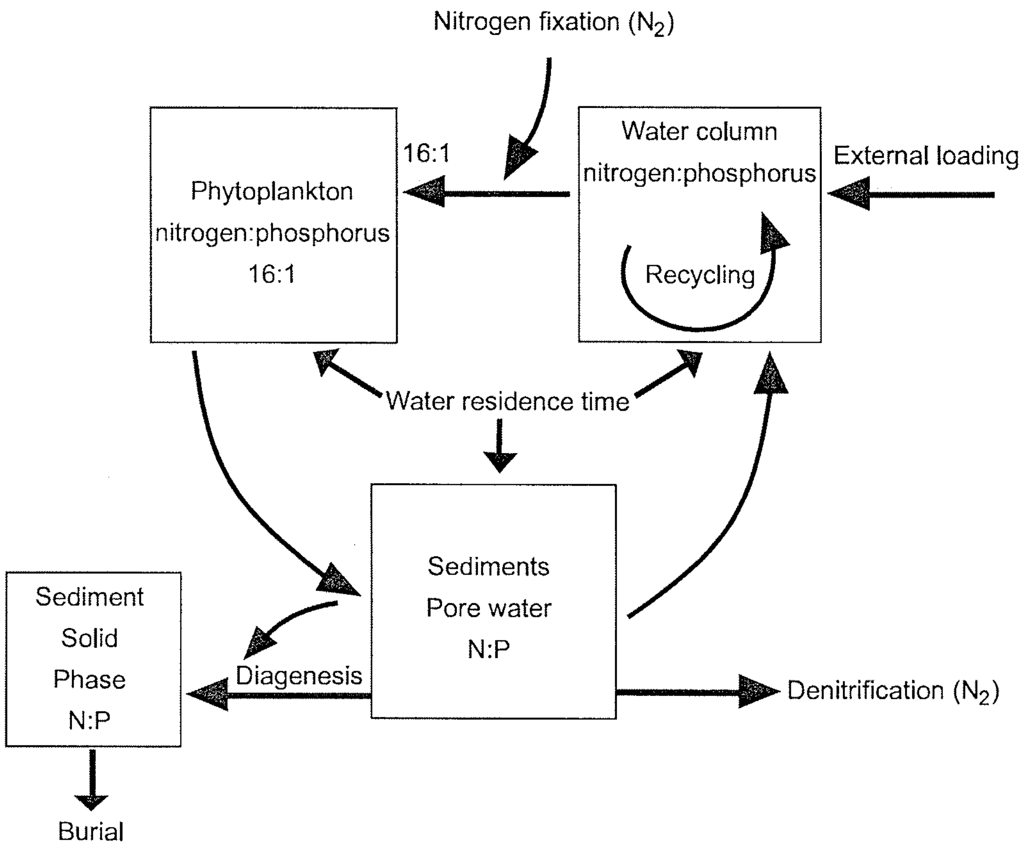

Figure 2-7. Factors that determine whether nitrogen or phosphorus is more limiting in aquatic ecosystems, where one of these macronutrients is limiting to net primary production. Phytoplankton use nitrogen and phosphorus in the approximate molar ratio of 16:1. The ratio of available nitrogen in the water 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.

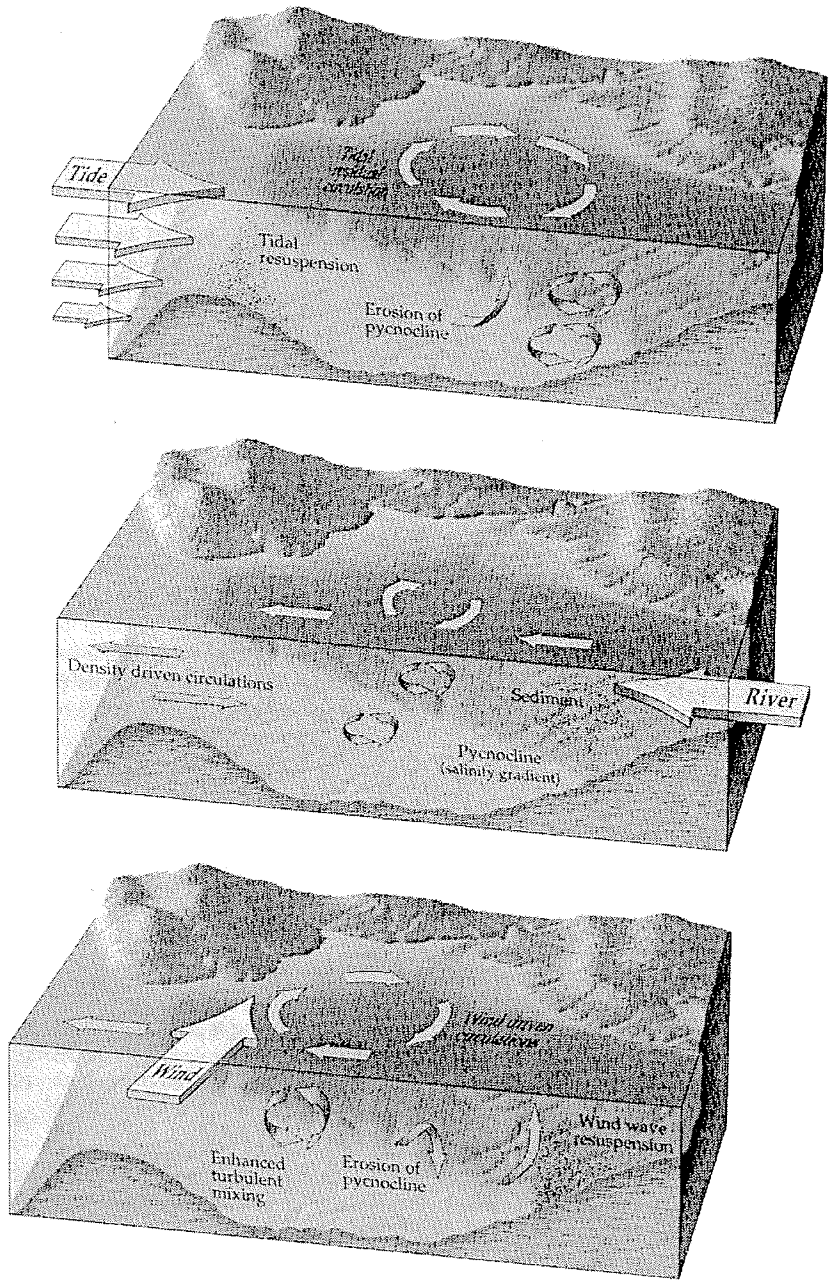

Figure 2-8. Cartoon diagrams of three physical forcings that operate at the interface between SCEs and the coastal ocean (tides), watershed (river inflow), and atmosphere (wind). Each physical forcing influences the growth rate of the resident phytoplankton population through, for example, its influence 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.

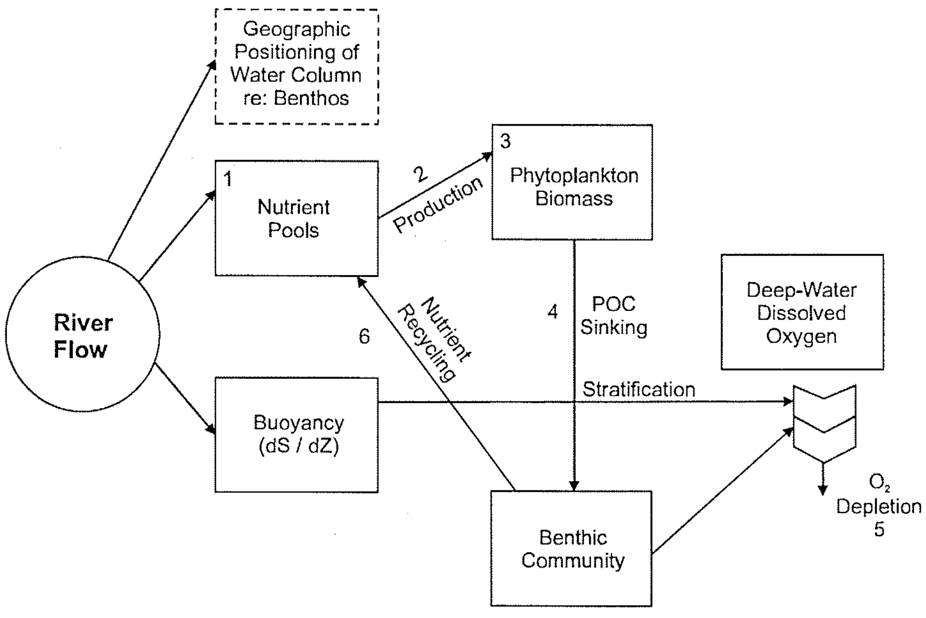

Figure 2-9. Simple schematic diagram showing the influences of river flow on ecosystem stocks and processes examined in this study. The mechanistic relationships between river flow and the stocks and processes shown in the diagram are explained in the text. Source: Boynton and Kemp 2000.

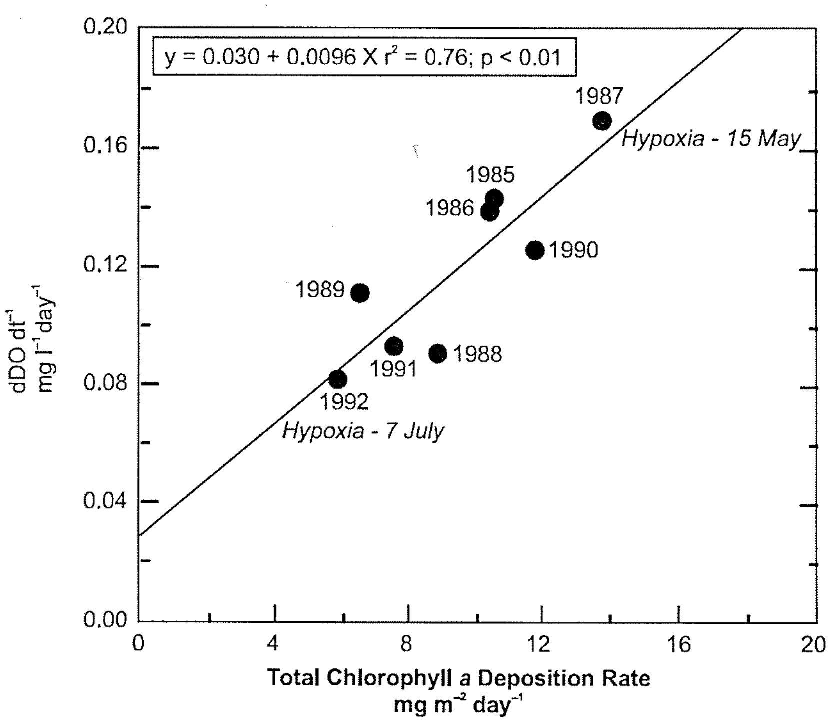

Figure 2-10. Scatter diagram showing the relationship between the rate of decline in dissolved-oxygen concentrations in deep water (dDO dt^{-1}) and average deposition rates of total chlorophyll *a* during the spring-bloom period. Data are from the 1985-1992 period and were collected at the R-64 site.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

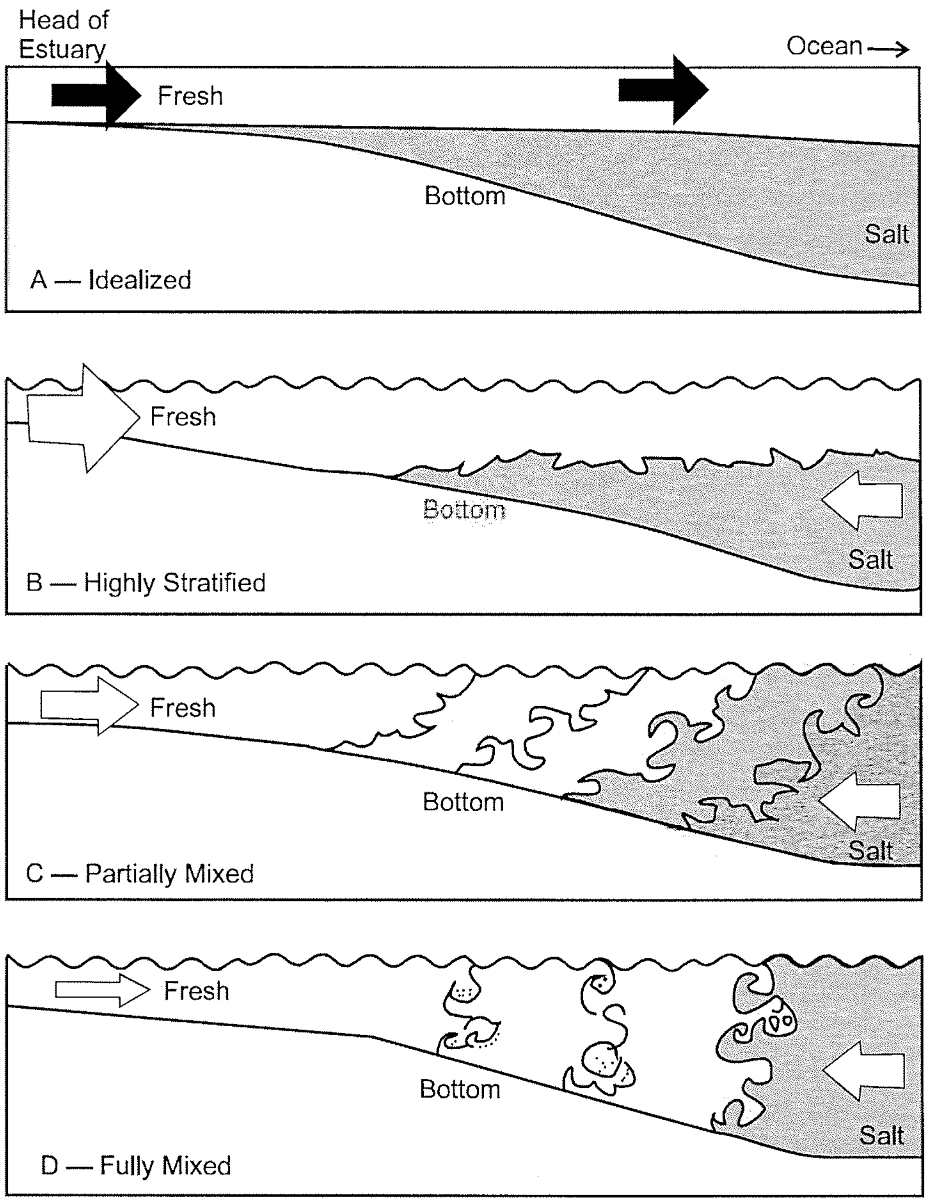

Figure 2-11a–d. Schematic diagram of coastal plain estuary types, indicating direction and degree of mixing. Arrows show direction of net mass transports of water, and the arrow size indicates the relative magnitudes of the transports. Source: Lippson et al. 1979.

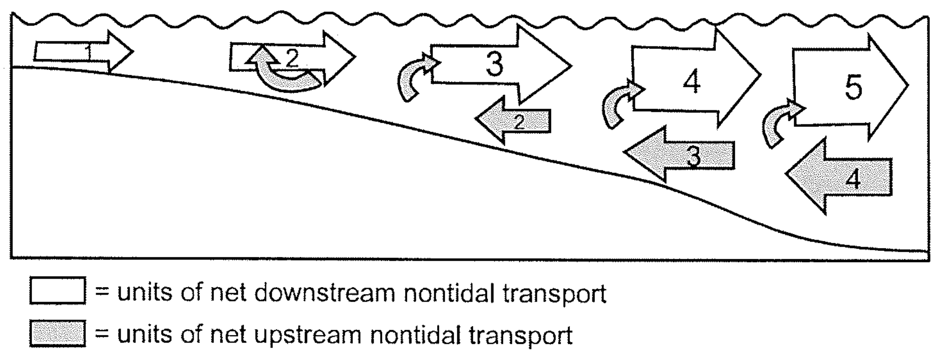

Figure 2-12. Net transports in estuaries resulting from estuarine flows and mixing. At any one point along an estuary, the difference between upstream- and downstream-directed transports is equal to the freshwater input to that point. In this example with no tributaries, the difference is equal to the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

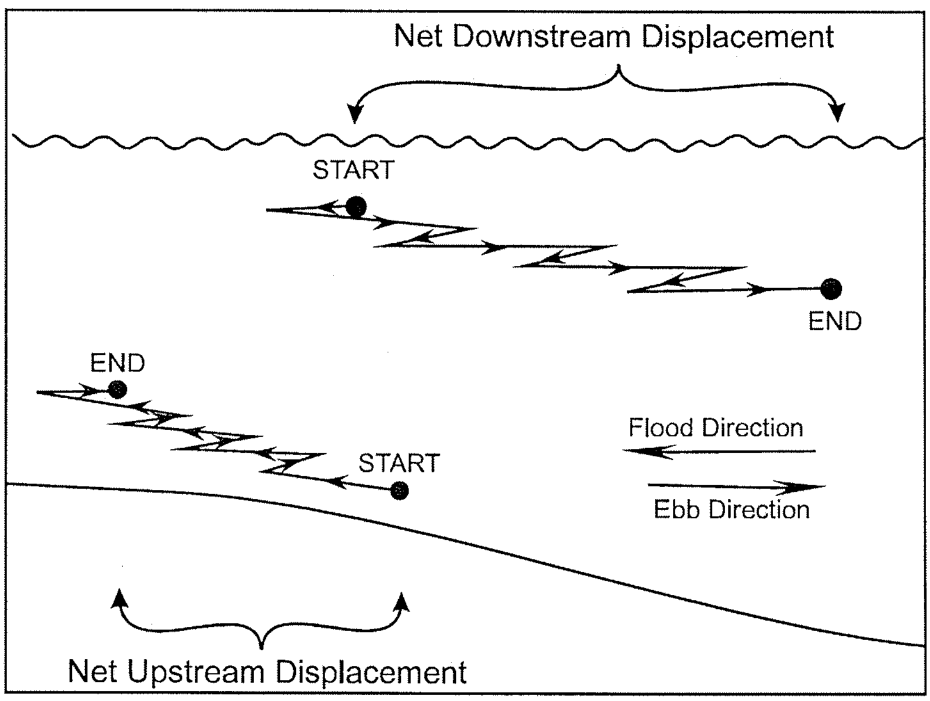

Figure 2-13. Net movement of a particle in each layer of a two-layered flow system. Source: Lippson et al. 1979.

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

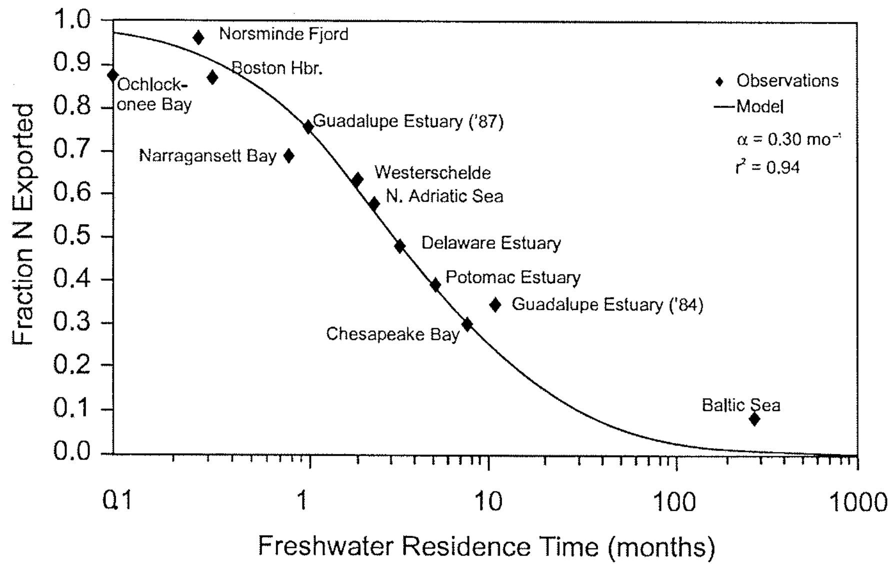

Figure 2-14. The fraction of landside nitrogen input exported from 11 North American and European estuaries versus freshwater residence time (linear time scale). Baltic Sea not shown. Source: Dettmann (in press).

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

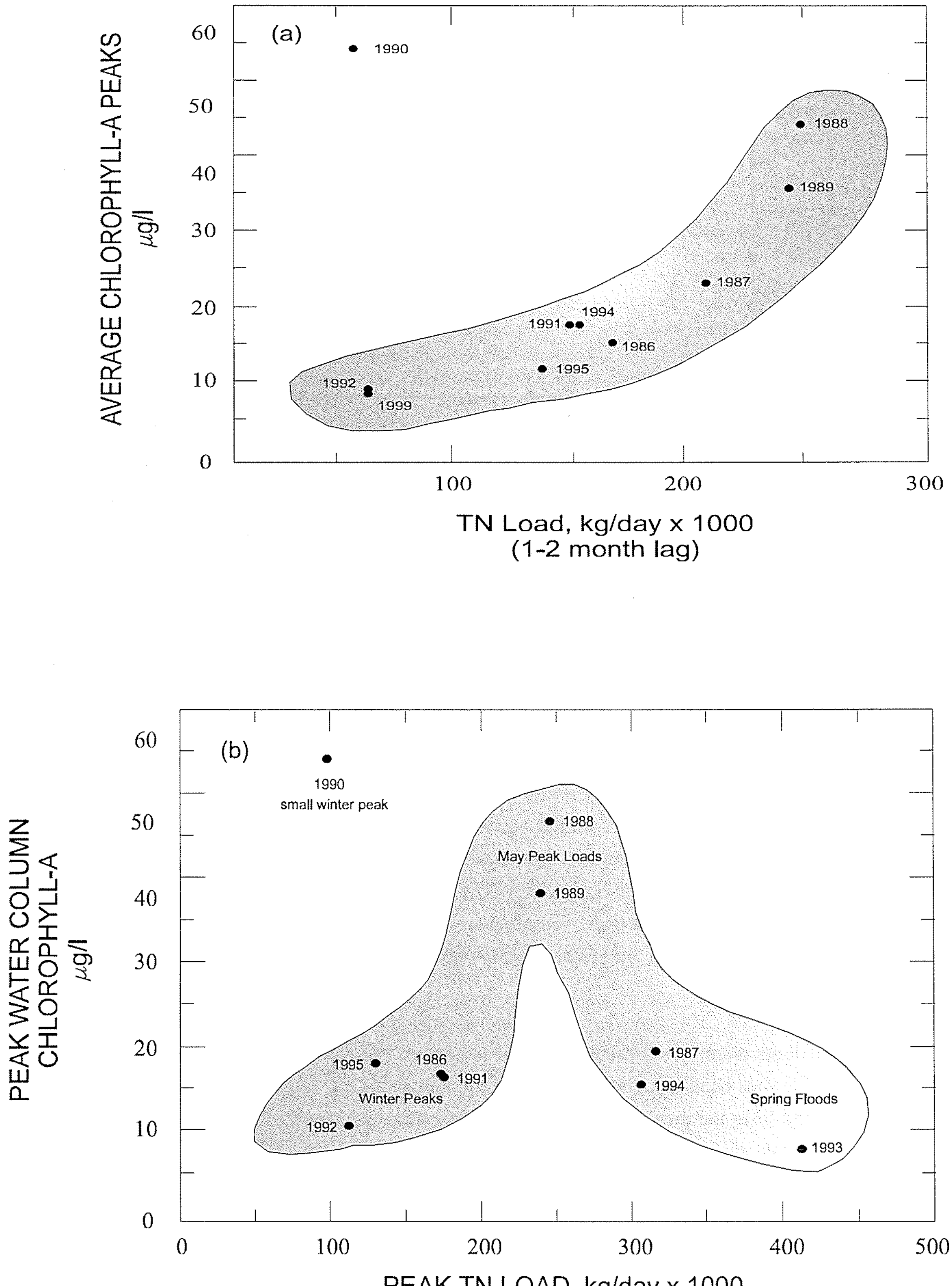

Figure 2-15. Scatter plots of water column averaged chlorophyll *a* at a mesohaline station (MLE 2.2) versus several different functions of total nitrogen (TN) loading rate measured at the fall line of the Potomac River estuary. Source: Boynton 1997.

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

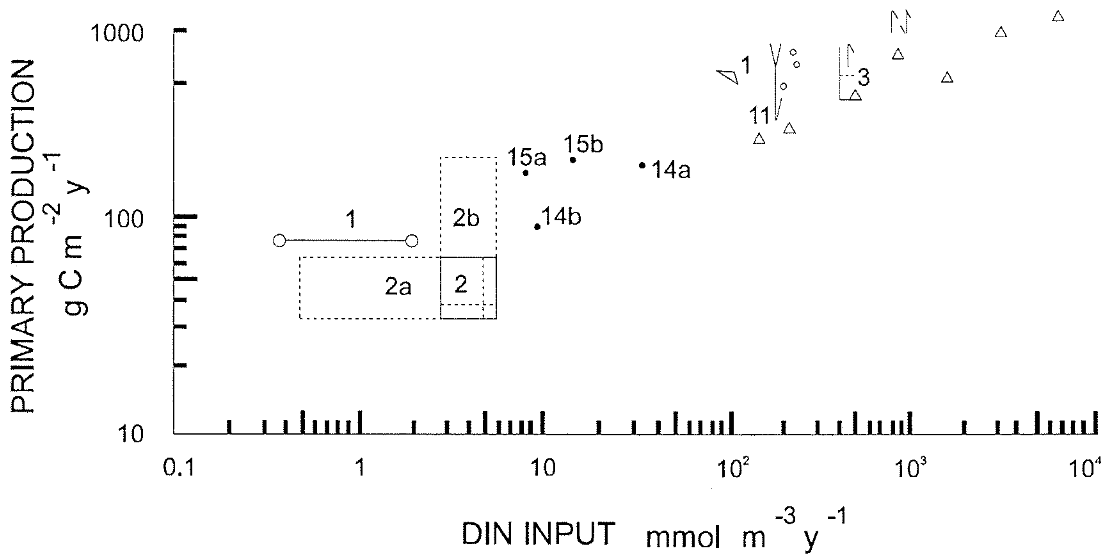

Figure 2-16a. Primary production by phytoplankton (^{14}C uptake) as a function of the estimated annual input of dissolved inorganic nitrogen per unit volume of a wide range of marine ecosystems. Source: Nixon (1992).

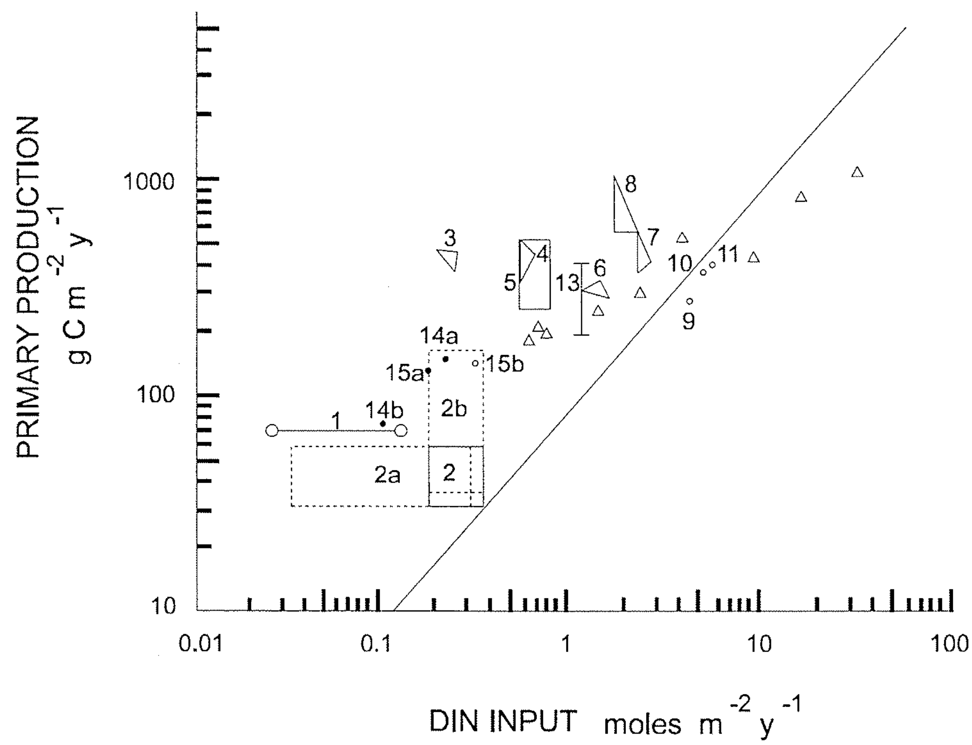

Figure 2-16b. Primary production by phytoplankton (^{14}C uptake) as a function of the annual input of dissolved inorganic nitrogen per unit area of a wide range of marine ecosystems. Source: Nixon (1992).

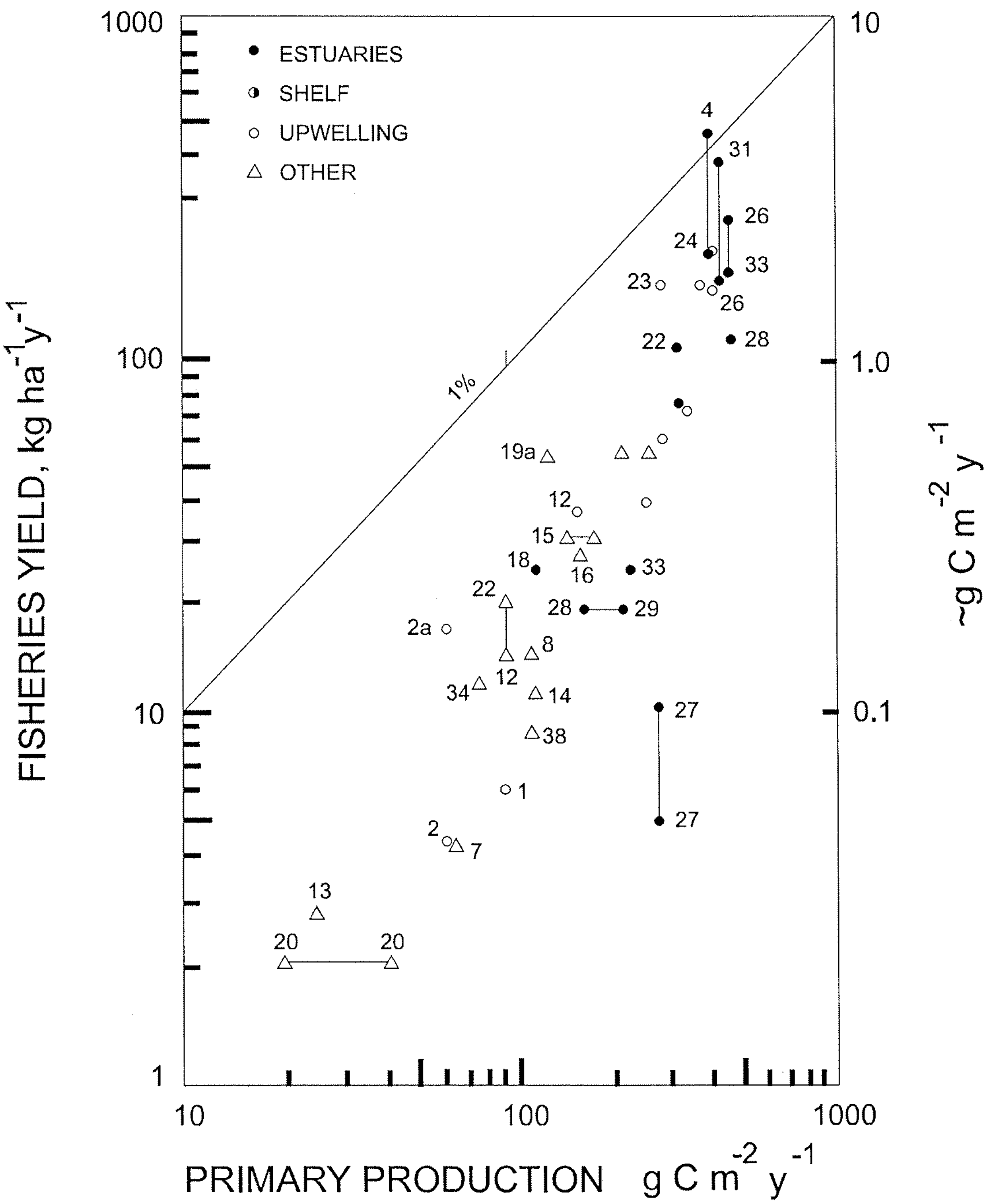

Figure 2-16c. Fisheries yield per unit area as a function of primary production in a wide range of estuarine and marine systems. Modifed from Nixon (1988) to include a revised primary production estimate for the Peru Upwelling from Guillen and Calienes (1981). Systems identified and data sources in Nixon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

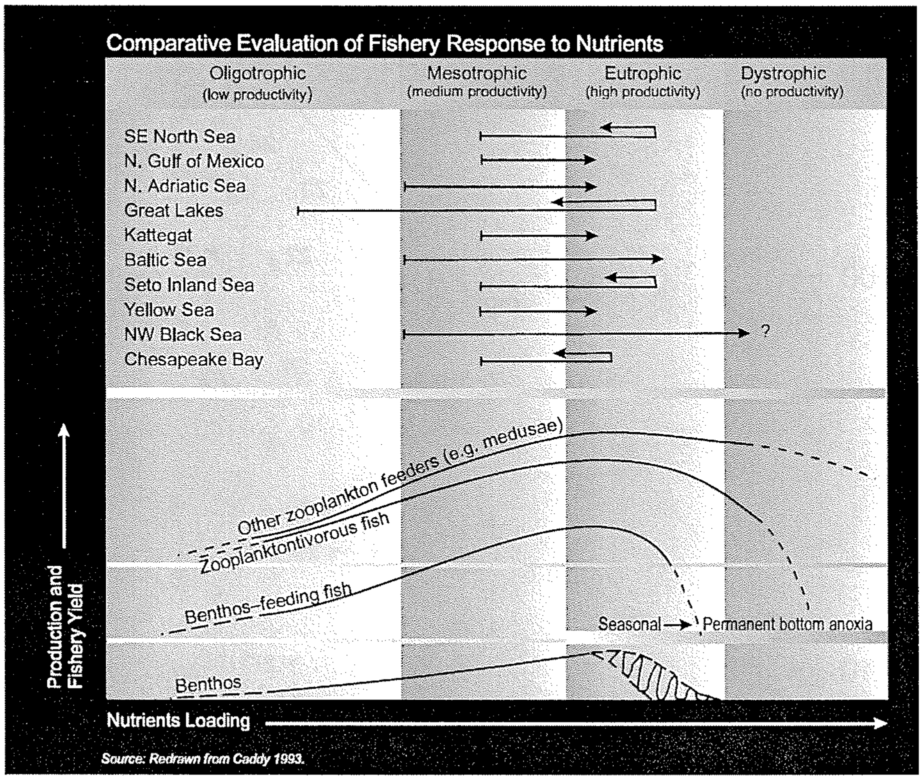

Figure 2-17. Comparative evaluation of fishery response to nutrients. Although higher nutrient concentrations initially increase the productivity of fisheries, ecological systems worldwide show negative effects as nutrient loading increases and hypoxic or anoxic conditions develop. Each generic curve 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.

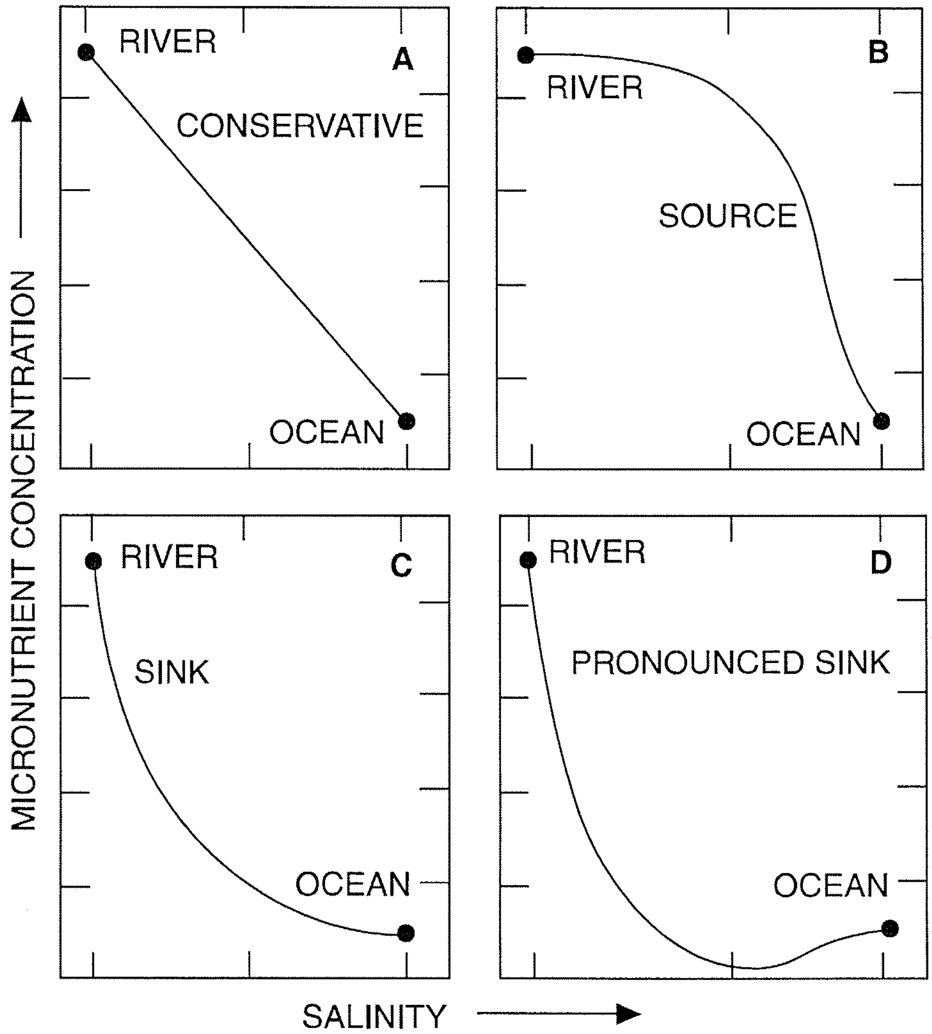

Figure 3-1. Idealized micronutrient-salinity relations showing concentration and mixing of nutrient-rich river water with nutrient-poor seawater. Source: Peterson et al. 1975. A. Expected concentration-salinity distribution of a substance behaving in a conservative manner (e.g., chloride) in an estuary. 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.