

CHAPTER 2

Scientific Basis for Estuarine and Coastal Waters Quantitative Nutrient Criteria

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

2.1 INTRODUCTION

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

Purpose and Overview

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

Coastal areas, including estuaries and upwelling regions, account for only 10% of the ocean by area but at least 25% of the ocean's primary productivity and upwards of 95% of the world's estimated fishery yield (Walsh 1988). These areas are also an important organic carbon sink of atmospheric CO₂. 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.

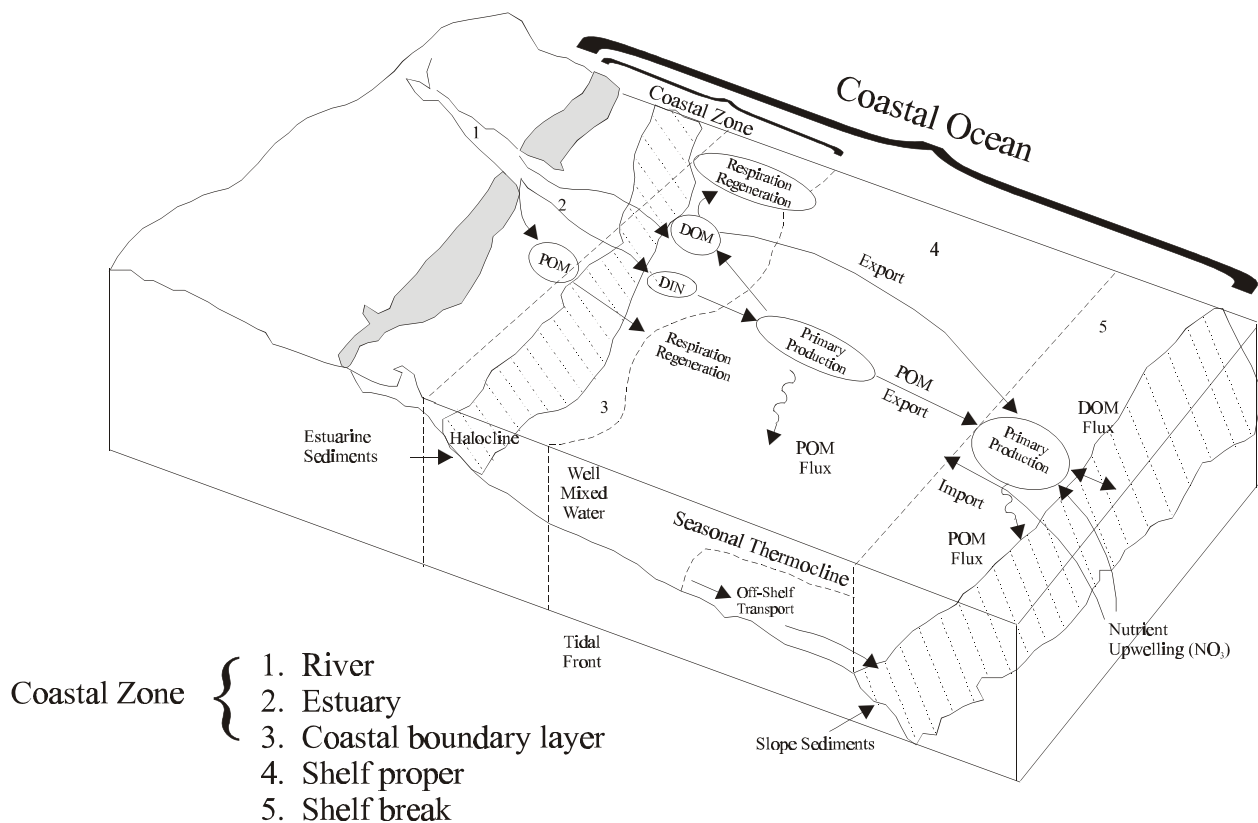


Figure 2-1. Idealized scheme defining the coastal ocean and the coastal zone, with some key biochemical fluxes linking land and sea and pelagic and benthic processes. The latter are not to scale. Source: Alongi 1998.

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 indeterminacy 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 break 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

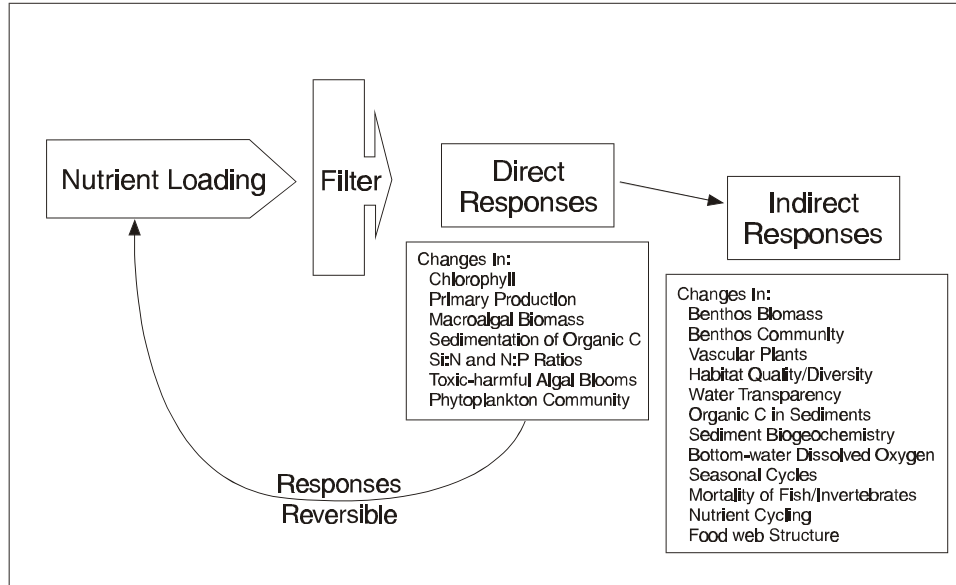


Figure 2-2. Schematic representation of the contemporary (Phase II) conceptual model of coastal eutrophication. Advances in recent decades include explicit recognition of (1) a complex suite of both direct and indirect responses to change in nutrient inputs; (2) system attributes that act as a filter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

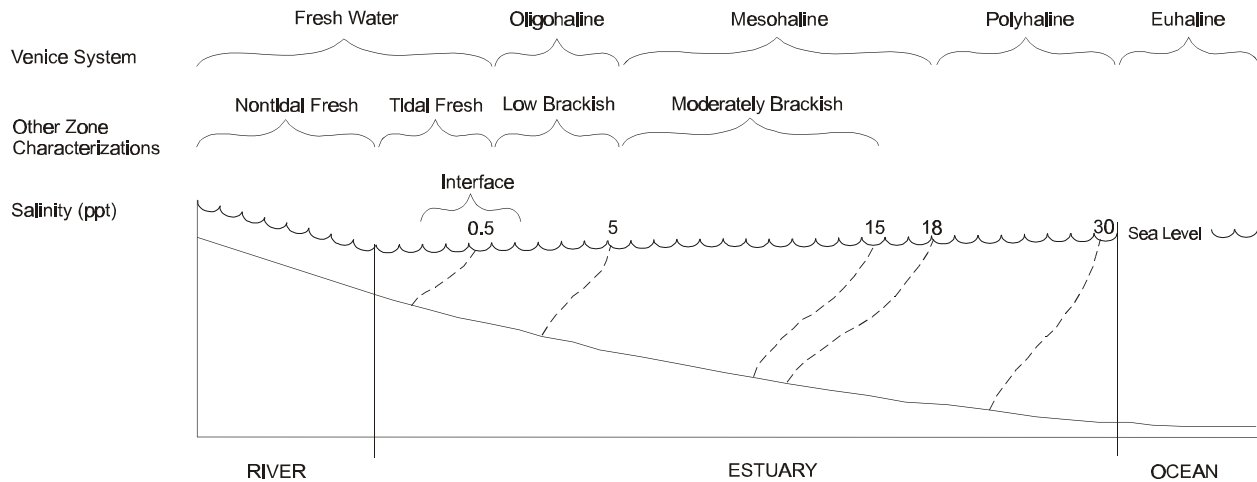


Figure 2-3. Salinity zones. The Venice System is a well-accepted method of characterizing salinity zones and covers the salinity ranges from riverine regions to the ocean. The freshwater category in the Venice System has been modified in this atlas to account for the tidal and nontidal regions found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

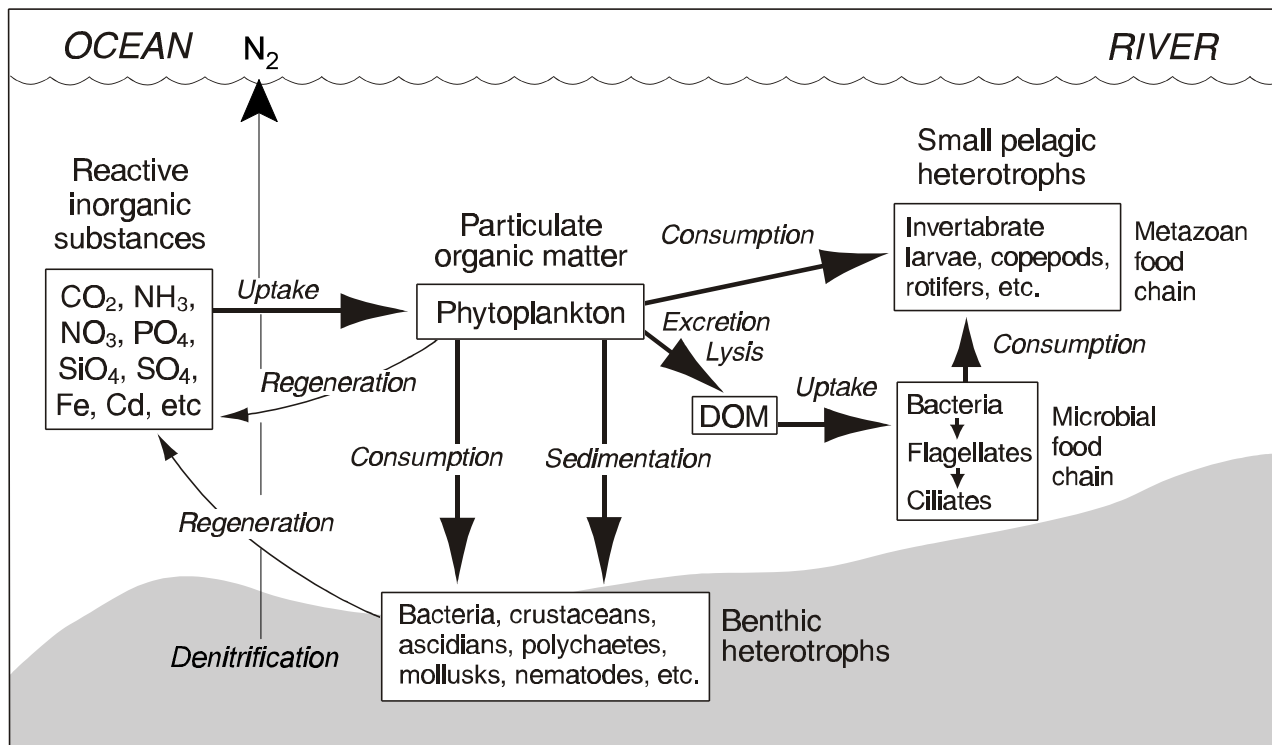


Figure 2-4. Schematic illustrating the central role of phytoplankton as agents of biogeochemical change in shallow coastal ecosystems. Phytoplankton assimilate reactive inorganic substances and incorporate these into particulate (POM) and dissolved organic matter (DOM) which support the production of 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)
<i>Eastern Boundary Current</i>			
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
<i>Western Boundary Currents</i>			
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
<i>Mesotrophic Systems</i>			
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-Uruguay	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
<i>Phototrophic Systems</i>			
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
<i>Eutrophic Systems</i>			
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 Zijlstra.

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 managers 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.

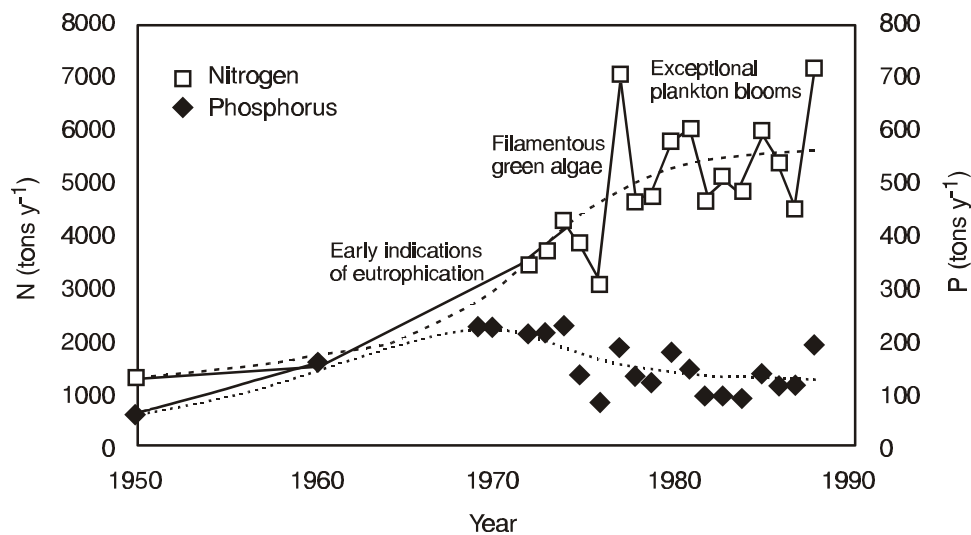


Figure 2-5. Transport of nutrients to Laholm Bay, Sweden. Periods of significant changes in the marine biota are also indicated (modified from Rosenberg et al. 1990).
Source: NRC 2000.

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 $\mu\text{g/L}$ with every 1 μM increase in DIN or approximately about 0.75 $\mu\text{g chl}/\mu\text{M DIN}$ (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 $\mu\text{g/L}$, 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 $\mu\text{M TN}$ (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,

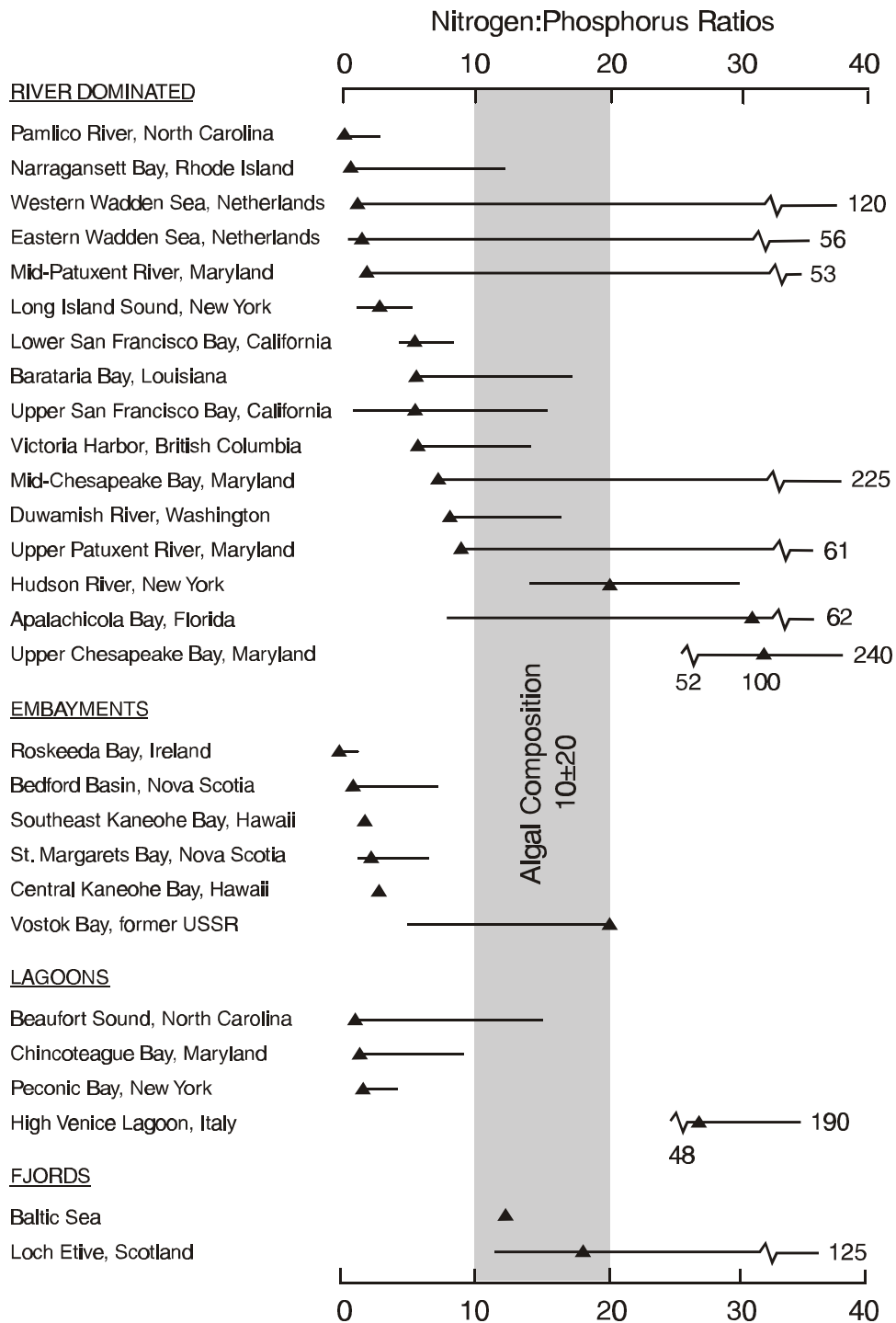


Figure 2-6. Summary of nitrogen:phosphorus ratios in 28 sample estuarine ecosystems. Horizontal bars indicate the annual ranges in nitrogen:phosphorus ratios; solid triangles represent the ratio at the time of maximum productivity. Vertical bands represent the typical range of algal composition ratios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

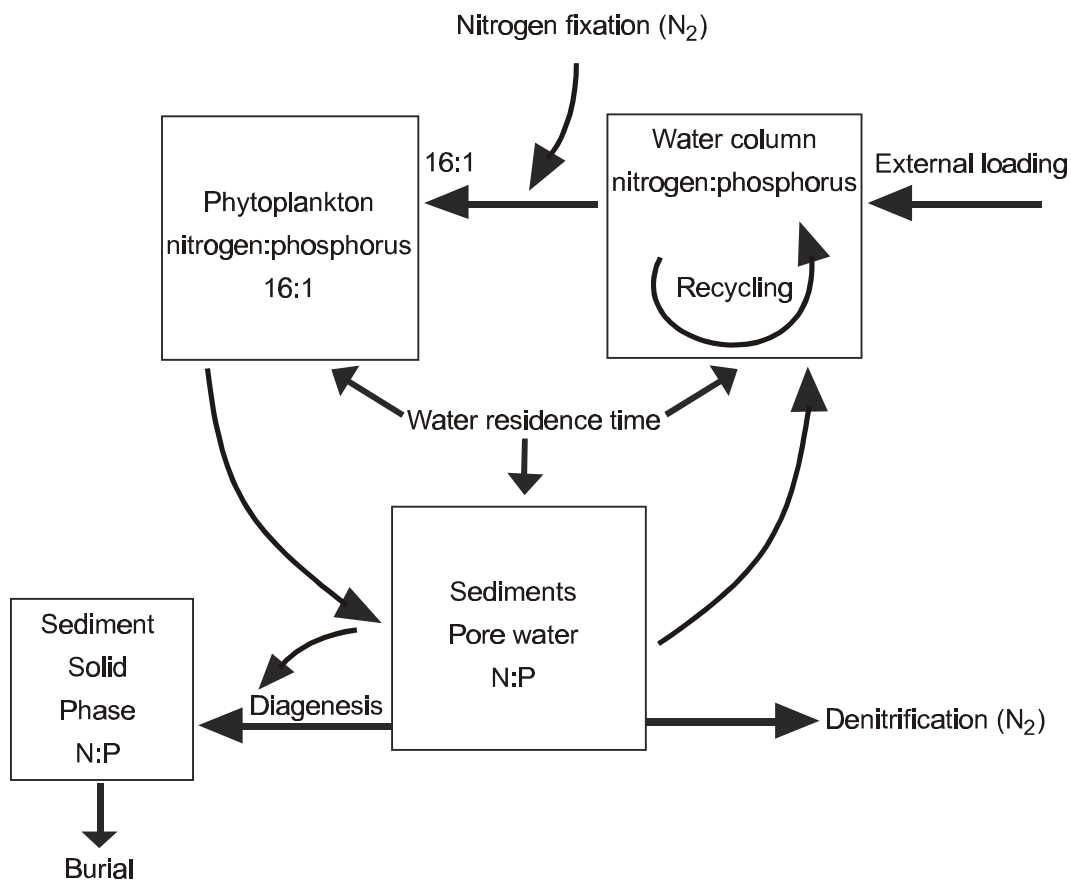


Figure 2-7. Factors that determine whether nitrogen or phosphorus is more limiting in aquatic ecosystems, where one of these macronutrients is limiting to net primary production. Phytoplankton use nitrogen and phosphorus in the approximate molar ratio of 16:1. The ratio of available nitrogen in the water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

^aH= hypoxia, A= anoxia.

^bVolumetric 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/m³, 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).

^cSee 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⁻²y⁻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.

^dExcept 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.

^eAlso 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.

^fAlso 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)

^gMainstem stratification, increasing anoxic extent; Officer et al. 1984, Boynton and Kemp 2000.

^hTop line is for dry flow, bottom line is for wet flow.

ⁱOnly 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⁻²y⁻¹).

^jOviatt 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.

^kUses longer 7-d residence time during very low flow conditions, Asselin and Spaulding 1993.

^lTN budget includes 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.

^mNorthern harbor section, Kelly 1998. Harbor station production of Kelly and Doering 1997.

ⁿArea represents greatest measured extent of hypoxic zone. Higher production is for immediate plume (Rabalais et al. 2000). TN loading is to a 20,000-km² hypoxic zone only (and thus is a maximal rate) based on Mississippi/Atchafalaya input of 130 x 10⁹ 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.

^oAssumed 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.

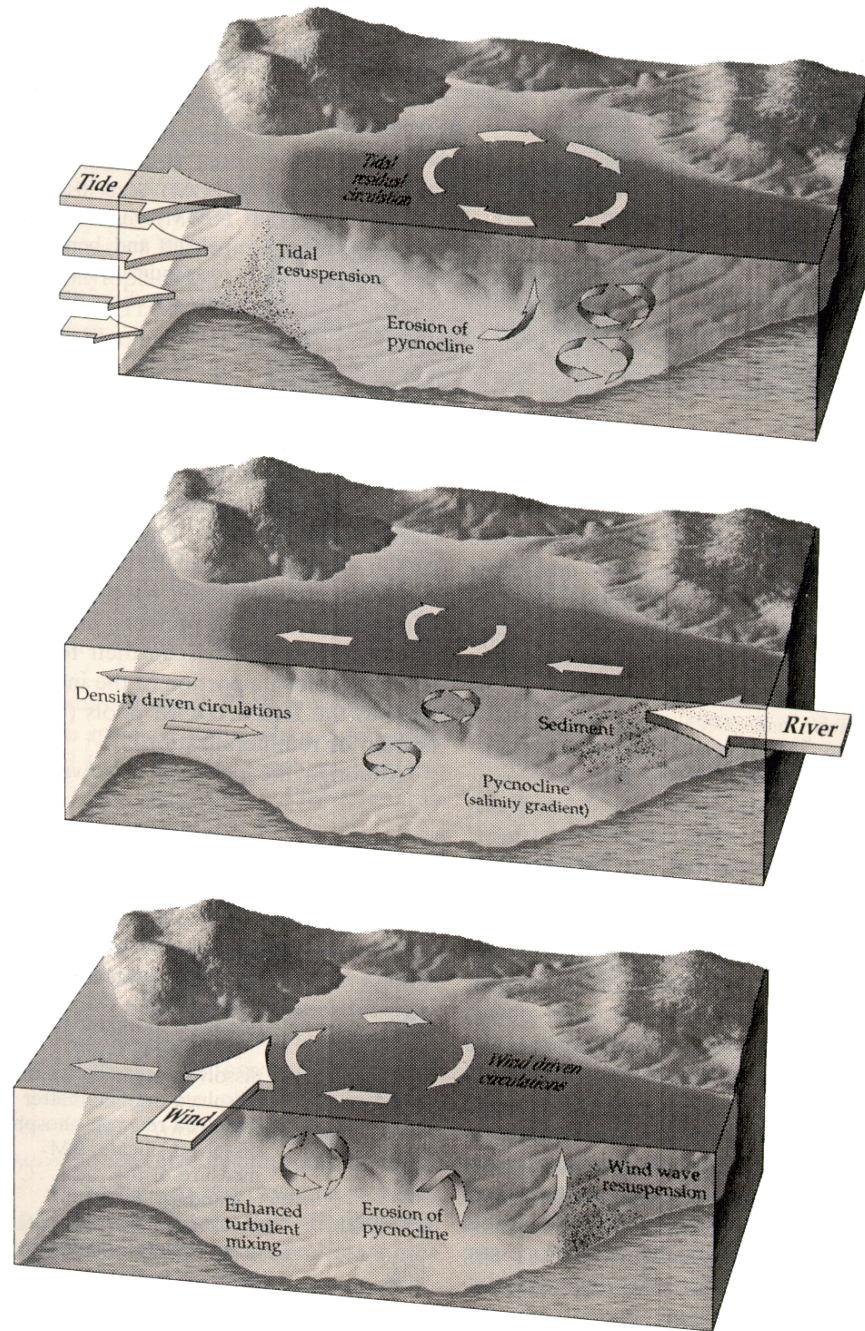


Figure 2-8. Cartoon diagrams of three physical forcings that operate at the interface between SCEs and the coastal ocean (tides), watershed (river inflow), and atmosphere (wind). Each physical forcing influences the growth rate of the resident phytoplankton population through, for example, its influence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

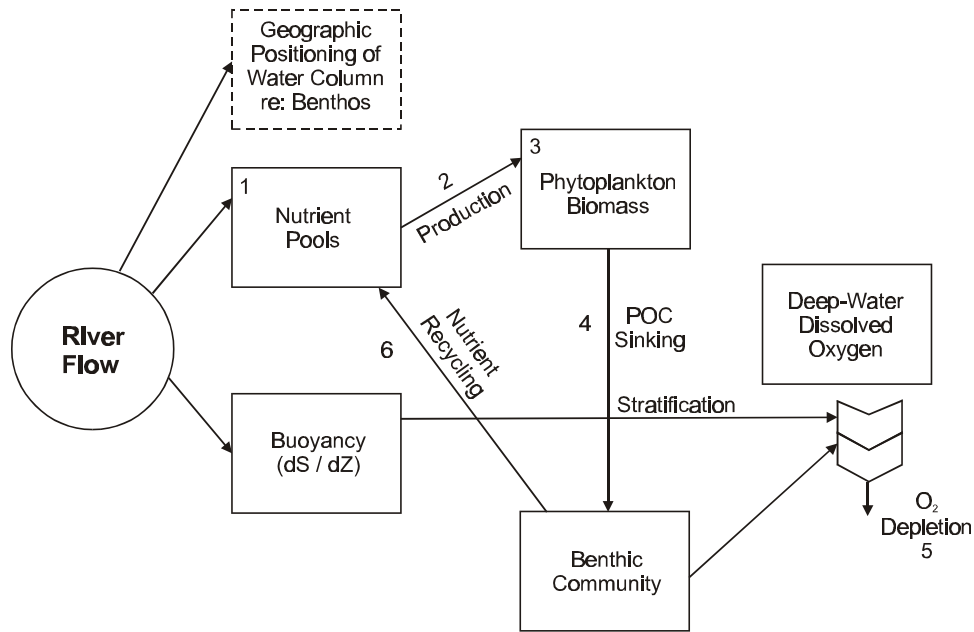


Figure 2-9. Simple schematic diagram showing the influences of river flow on ecosystem stocks and processes examined in this study. The mechanistic relationships between river flow and the stocks and processes shown in the diagram are explained in the text. Source: Boynton and Kemp 2000.

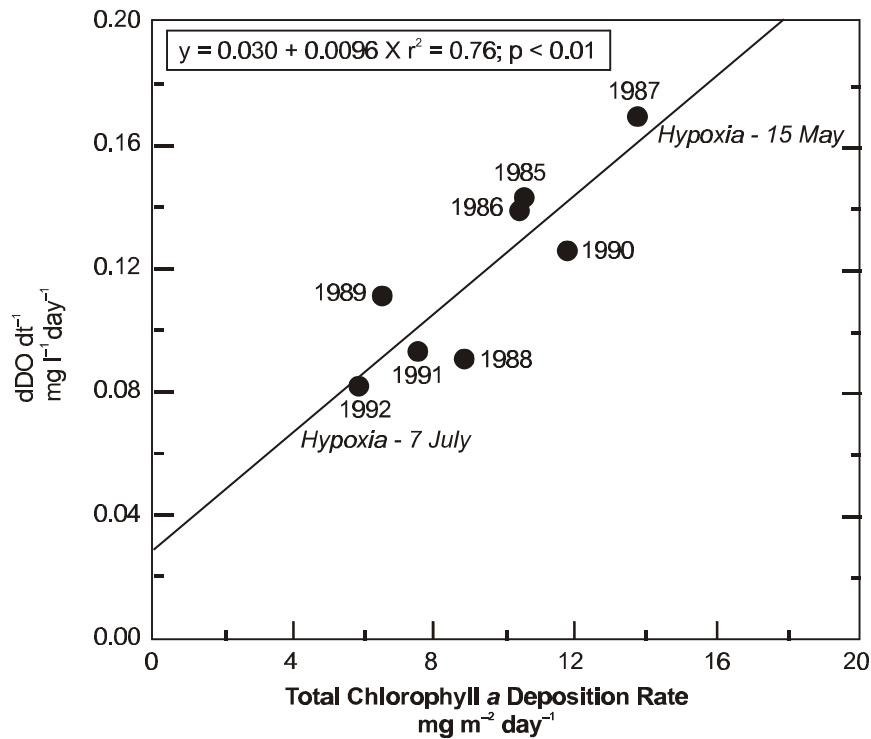


Figure 2-10. Scatter diagram showing the relationship between the rate of decline in dissolved-oxygen concentrations in deep water ($dDO dt^{-1}$) and average deposition rates of total chlorophyll *a* during the spring-bloom period. Data are from the 1985-1992 period and were collected at the R-64 site. 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 poikilotherms, 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).

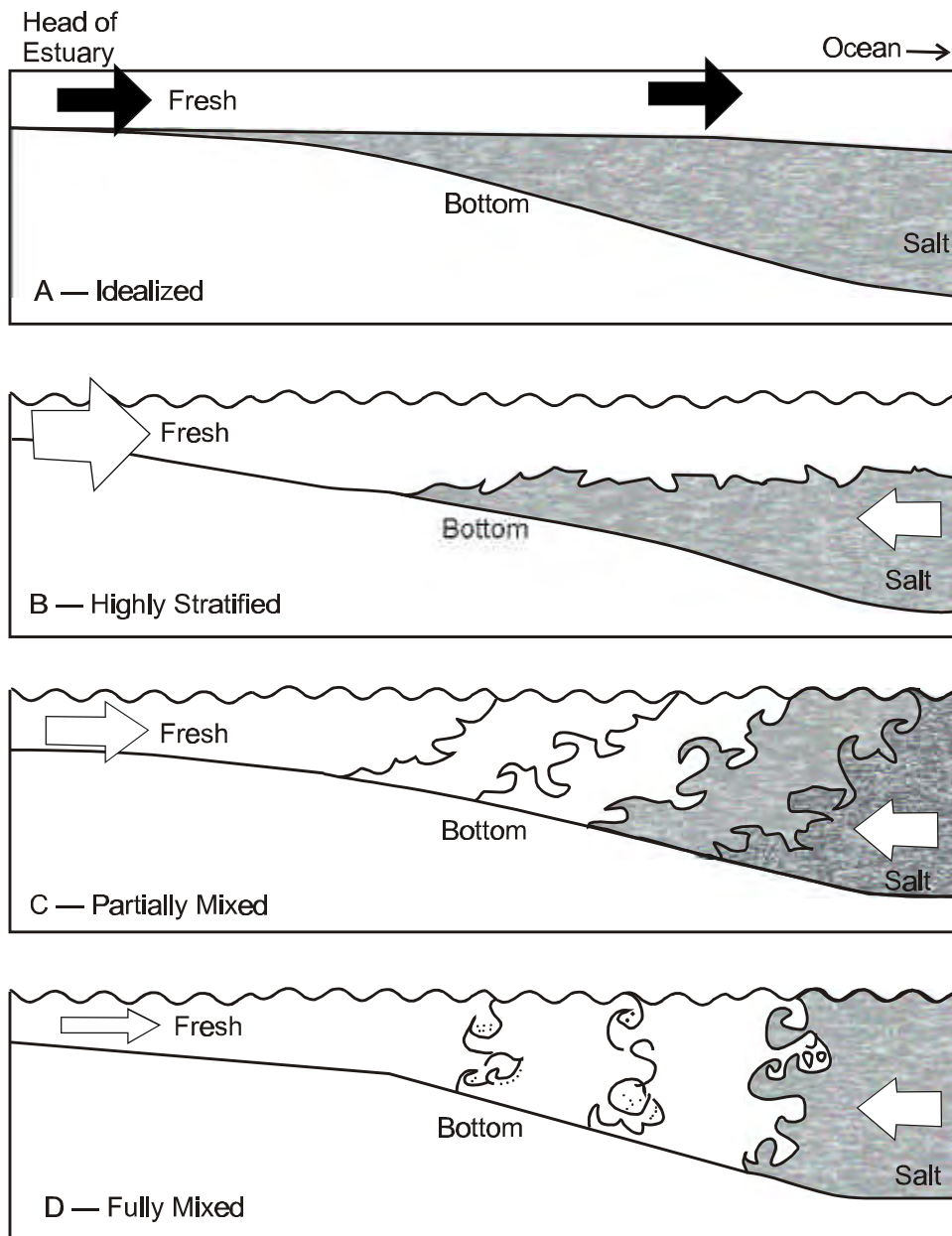


Figure 2-11a-d. Schematic diagram of coastal plain estuary types, indicating direction and degree of mixing. Arrows show direction of net mass transports of water, and the arrow size indicates the relative magnitudes of the transports. Source: Lippson et al. 1979.

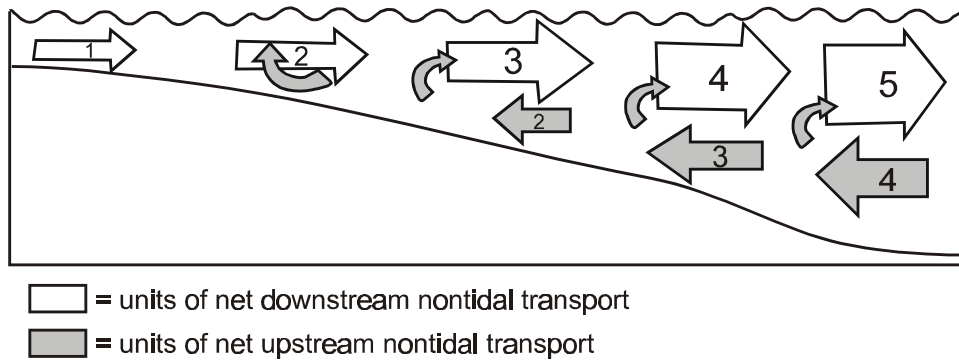


Figure 2-12. Net transports in estuaries resulting from estuarine flows and mixing. At any one point along an estuary, the difference between upstream- and downstream-directed transports is equal to the freshwater input to that point. In this example with no tributaries, the difference is equal to the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

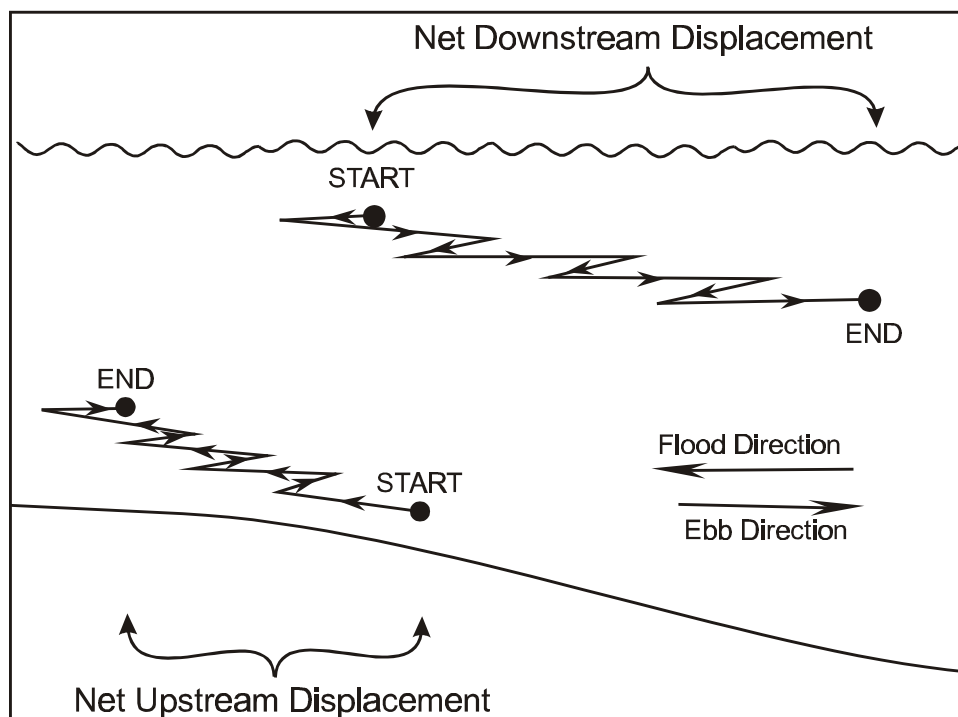


Figure 2-13. Net movement of a particle in each layer of a two-layered flow system. Source: Lippson et al. 1979.

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

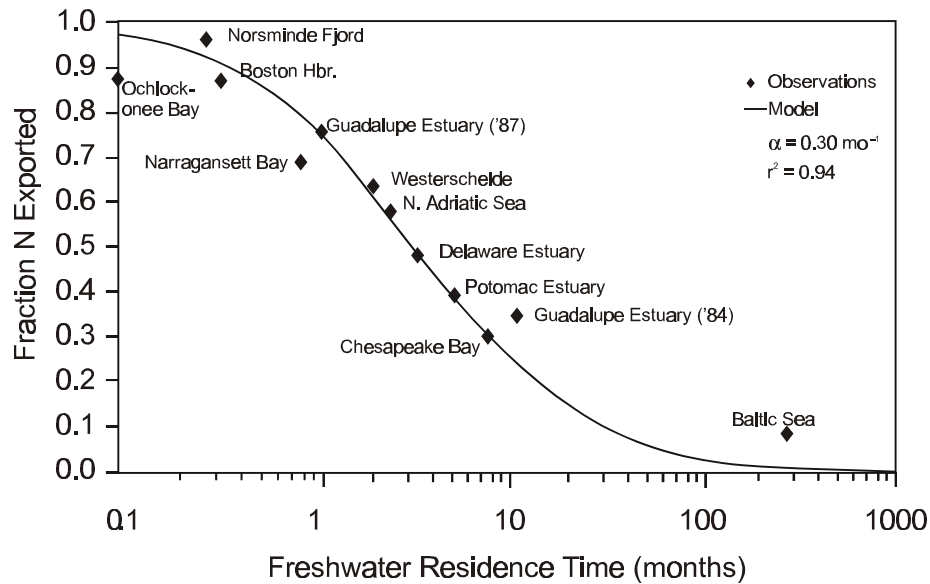


Figure 2-14. The fraction of landside nitrogen input exported from 11 North American and European estuaries versus freshwater residence time (linear time scale). Baltic Sea not shown. Source: Dettmann (in press).

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

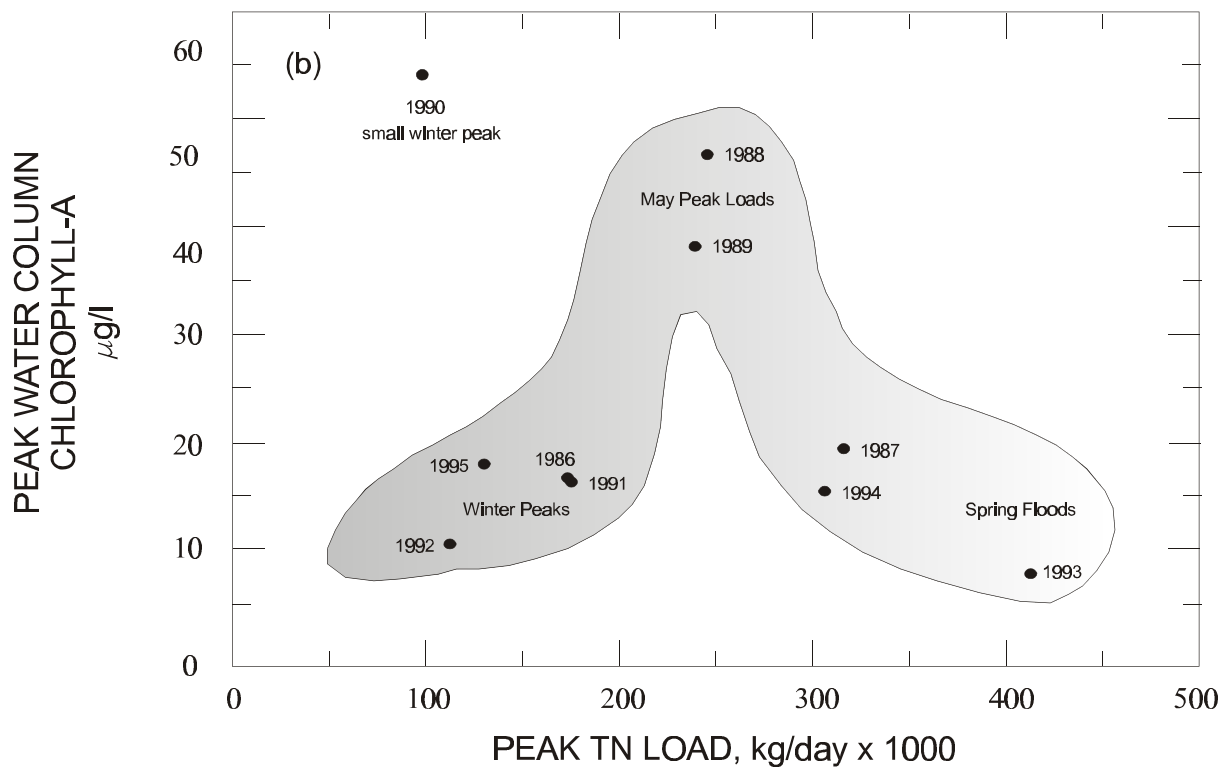
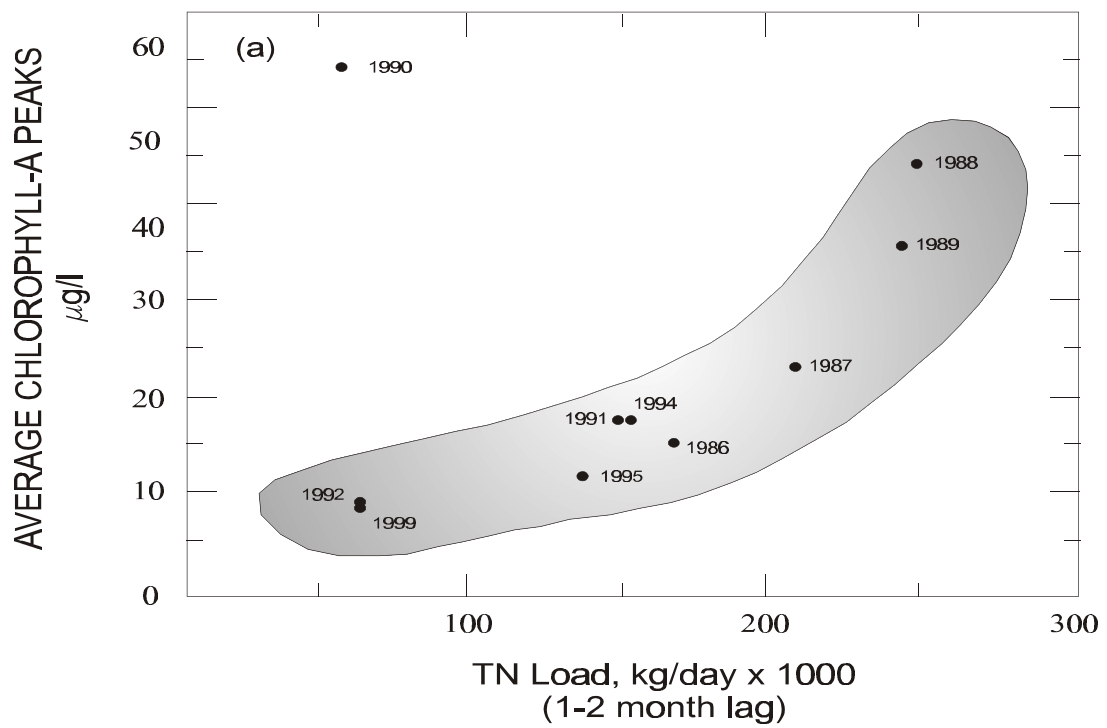


Figure 2-15. Scatter plots of water column averaged chlorophyll *a* at a mesohaline station (MLE 2.2) versus several different functions of total nitrogen (TN) loading rate measured at the fall line of the Potomac River estuary. Source: Boynton 1997.

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 H₂S apparently cause synergetic effects that make marine benthic animals more sensitive to hypoxia when H₂S 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 $\mu\text{g/L}$. 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 ($\text{g C m}^{-2} \text{ y}^{-1}$) and DIN input ($\text{moles m}^{-2} \text{ y}^{-1}$) and between fisheries yield ($\text{kg ha}^{-1} \text{ y}^{-1}$) 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

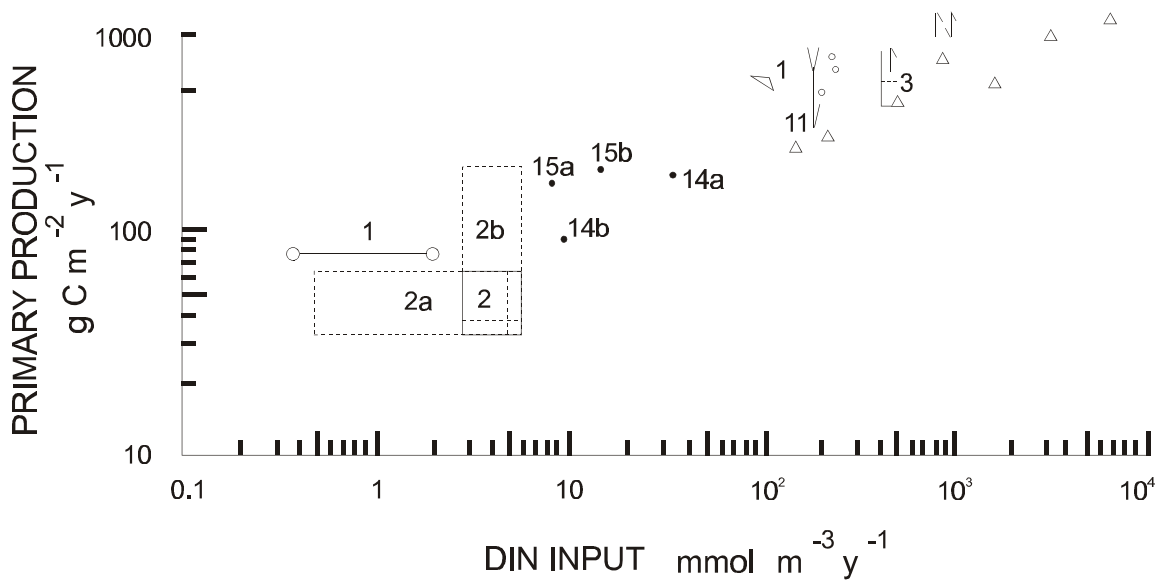


Figure 2-16a. Primary production by phytoplankton (^{14}C uptake) as a function of the estimated annual input of dissolved inorganic nitrogen per unit volume of a wide range of marine ecosystems. Source: Nixon (1992).

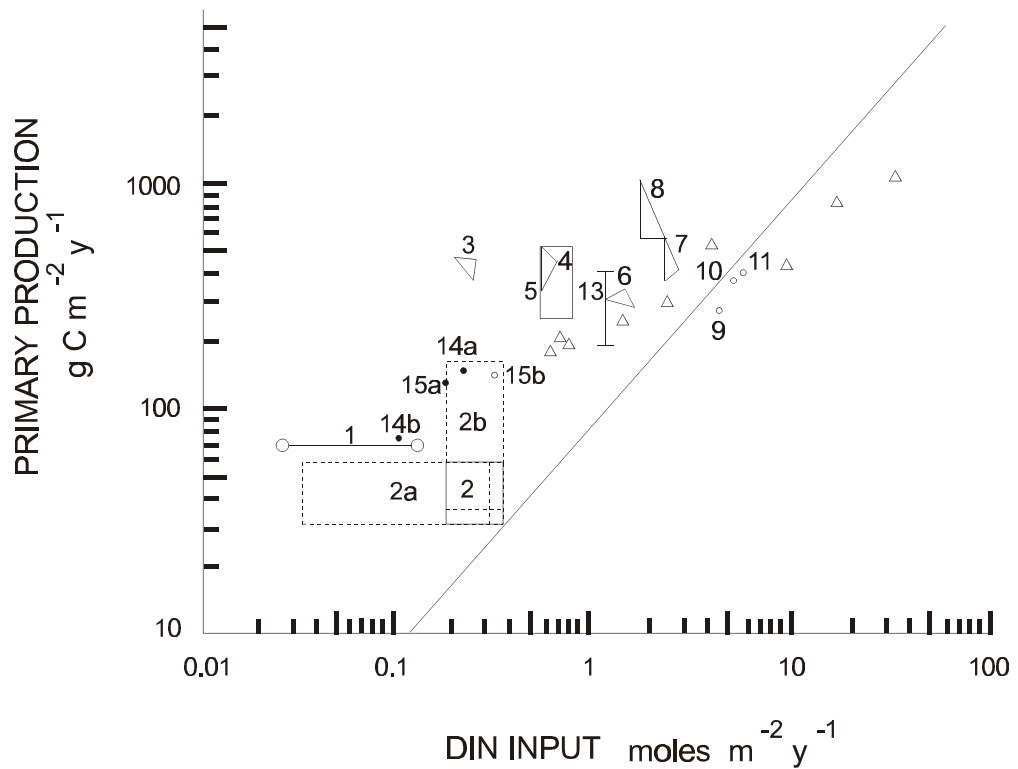


Figure 2-16b. Primary production by phytoplankton (^{14}C uptake) as a function of the annual input of dissolved inorganic nitrogen per unit area of a wide range of marine ecosystems. Source: Nixon (1992).

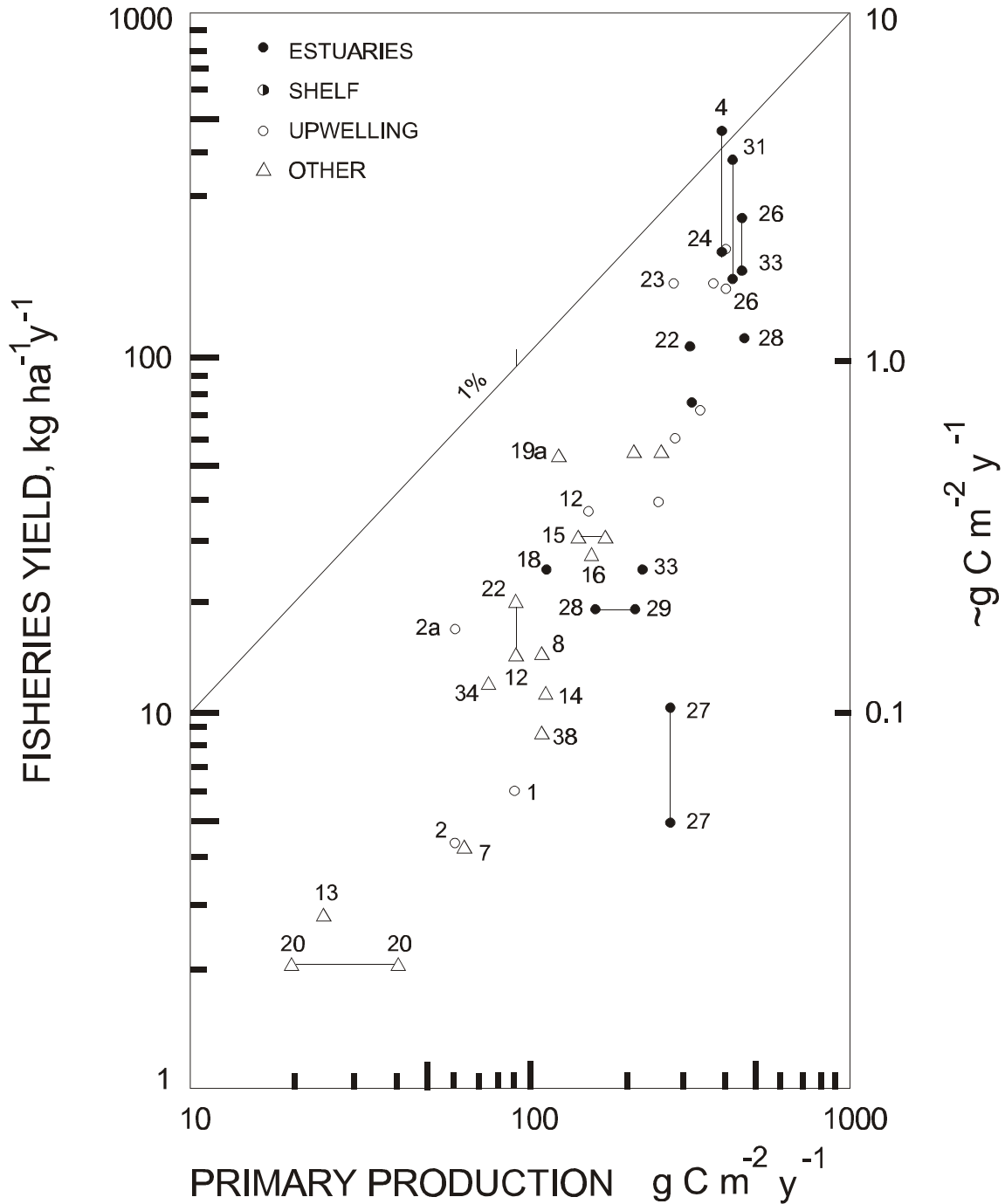


Figure 2-16c. Fisheries yield per unit area as a function of primary production in a wide range of estuarine and marine systems. Modified from Nixon (1988) to include a revised primary production estimate for the Peru Upwelling from Guillen and Calienes (1981). Systems identified and data sources in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

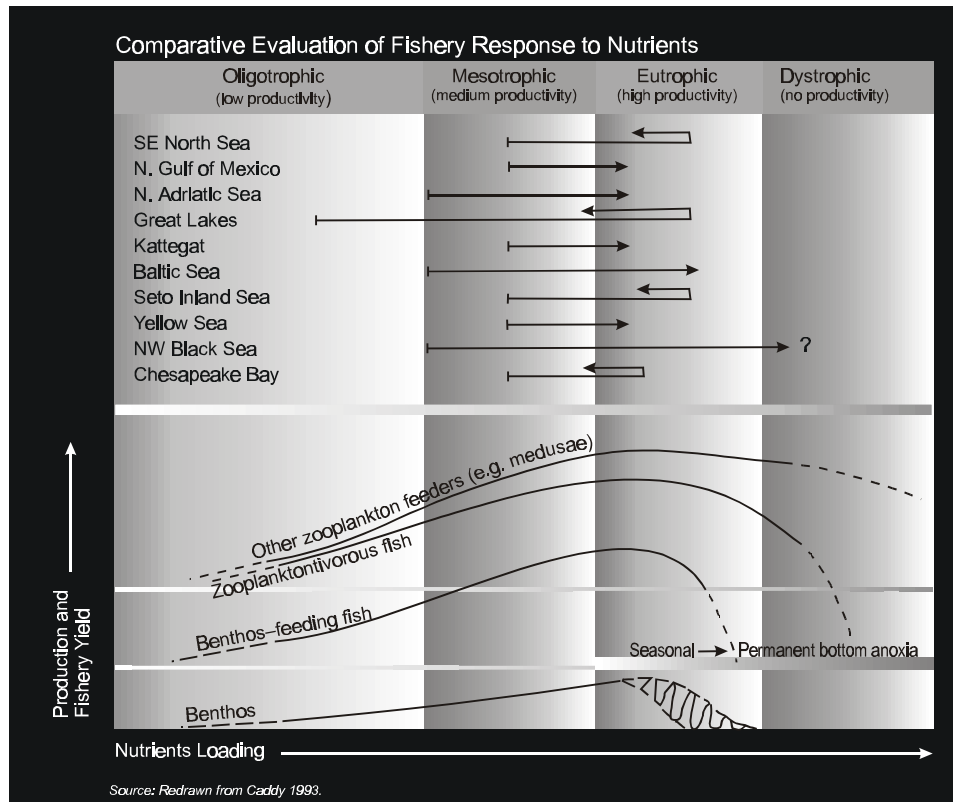


Figure 2-17. Comparative evaluation of fishery response to nutrients. Although higher nutrient concentrations initially increase the productivity of fisheries, ecological systems worldwide show negative effects as nutrient loading increases and hypoxic or anoxic conditions develop. Each generic curve 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 7

Nutrient and Algal Criteria Development

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

7.1 INTRODUCTION

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

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

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

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

7.2 ROLE OF REGIONAL TECHNICAL ASSISTANCE GROUPS

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

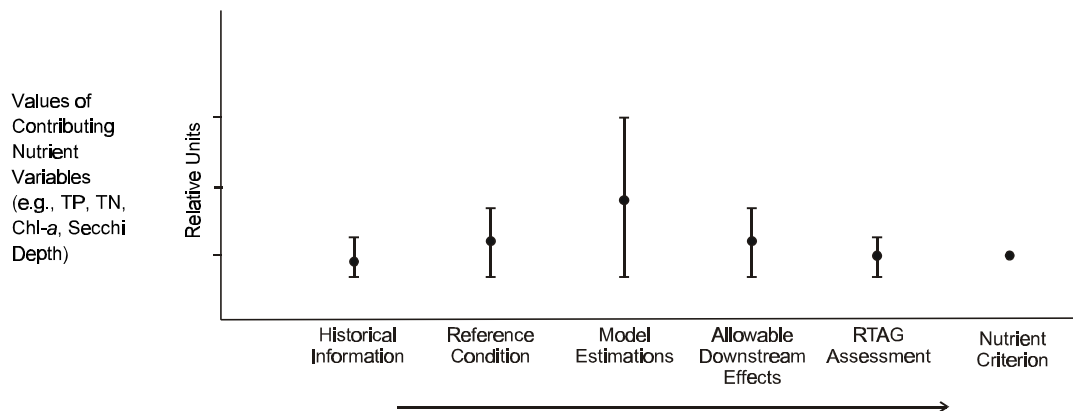


Figure 7-1. Generalized progression and relationship of the elements of a nutrient criterion.

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

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

7.3 CLASSIFICATION

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

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

7.4 DESCRIPTIVE BACKGROUND INFORMATION

Estuarine Watershed Characterization

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

Farm acreage, crop types, and fertilizer application rates also provide useful information to assess the potential historical magnitude in nonpoint source nutrient loading to estuarine systems. Atmospheric

plumes that result in nitrogen deposition should also be assessed as they may help explain an increased nitrogen load to coastal waters beyond that attributed to local land use activities. This also is a situation where reference sites, if available, will help the manager distinguish atmospheric from local anthropogenic causes of overenrichment. Conversely, increases in forest coverage and soil banking of agricultural lands may help explain potential decreases in nitrogen and phosphorus loading to estuaries and coastal waters.

Within Estuarine System Characterization

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

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

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

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

The above measurements performed as part of a general characterization may help relate nutrient enrichment effects and thresholds to existing and designated uses and identify any reference systems that

are minimally impacted by nutrient pollution. Such information may enter directly into predictive water quality models or serve as indirect collaborative information contributing to a “weight of evidence” analysis.

7.5 ELEMENTS OF NUTRIENT CRITERIA

Reference Condition

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

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

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

Historical Information

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

Models

In addition to computer modeling to help determine reference conditions, models may be used both to estimate nutrient loads and load reductions to achieve a targeted nutrient regime in the receiving estuarine and coastal waters systems. Various models are available (Chapter 9) that estimate nutrient erosion from different land uses, riverine transport of nutrients, and estuarine effects including hypoxia and chlorophyll *a* concentrations. These models are typically expensive to calibrate and verify, but in

large estuaries and coastal ecosystems their application can be cost-effective when costly nutrient control decisions are involved. A coupled nutrient transport and hydrodynamically coupled algal nutrient uptake and growth model provides the ability to address “what if scenarios” of nutrient load versus level of biological impairments (see Chesapeake Bay Case Study and web site).

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

Antidegradation Policy and Attention to Downstream Effects

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

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

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

determined from similar unimpaired systems or from a historical load and response relationship. It is then up to the States and Tribes to adopt the criteria into 303(c) water quality standards.

The RTAG

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

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

7.6 HYPOTHETICAL EXAMPLES OF NUTRIENT CRITERIA DEVELOPMENT DELIBERATIONS

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

Scenario

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

A 50 μM TN reference condition was determined by plotting TN concentrations for the mesohaline zone from 1970 to 2000. Because nutrient data were few for the estuary prior to 1970 and most of the data were collected for the mesohaline zone, a decision was made to select the 1970 summer average TN value as the reference condition with application for the mesohaline zone. Only an occasional bottom

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

7.7 EVALUATION OF PROPOSED CRITERIA

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

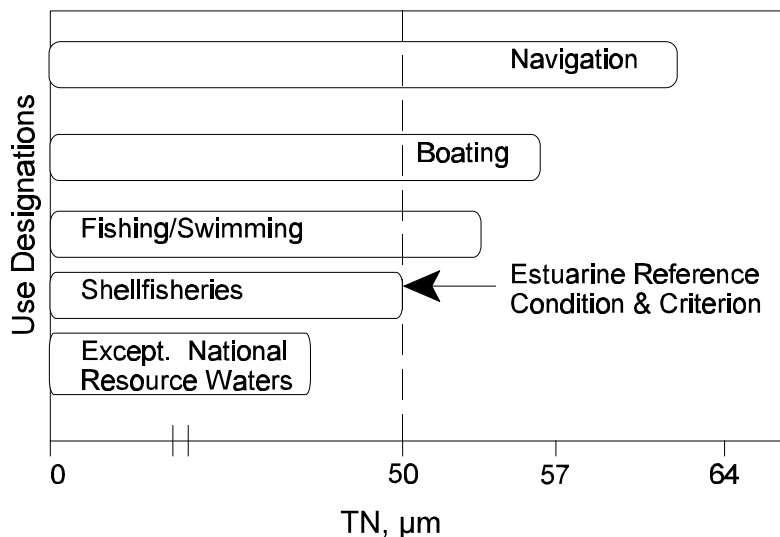


Figure 7-2. Hypothetical illustration of developing a TN criterion in an estuary.

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

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

Guidance for Interpreting and Applying Criteria

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

Do the Criteria Protect Designated Uses?

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

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

Outstanding National Resource Waters

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

Aquatic Life Uses

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

Fisheries

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

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

Recreation

Swimming/Primary Contact Recreation

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

Boating and Secondary Contact Recreation.

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

Restoration Goals

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

Sampling for Comparison to Criteria

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

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

The question arises, how many replicate samples are needed to obtain an acceptable precision of data in order to detect differences between sites and changes over time? This depends on the nature of the

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

7.8 NUTRIENT CRITERIA INTERPRETATION PROCEDURES

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

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

Decisionmaking Protocol

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

Multivariable Enrichment Index

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

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

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

* Each of the eight variables receives an EI score. The scoring procedure is: 0 = meets criterion; 2 = fails to meet criterion by 10%; 3 = fails to meet criterion by 25%; 5 = fails to meet criterion by 50% or more.

** Enrichment Index Value is the sum of the EI scores. The maximum i.e. worst score achievable is 25.

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

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

Frequency and Duration

Frequency and duration are important concerns when evaluating any water with respect to meeting criteria. This is a difficult process at this initial phase of the program because the data sources for criteria development are presently so diverse. In general, however, the method of data gathering for compliance should be as near as possible to that used to establish the criteria, especially keeping in mind

tidal phase and salinity. Once consistency is established, excursions from the criteria based on frequency and duration can be evaluated whether based on a decision rule or a multivariable index.

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

7.9 CRITERIA MODIFICATIONS

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

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

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

7.11 IMPLEMENTATION OF NUTRIENT CRITERIA INTO WATER QUALITY STANDARDS

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