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Chapter 10. Nitrogen Effects on Coastal Marine Ecosystems

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Throughout the 20th century, a rapidly growing human population increased the global circulation of nitrogen (N). In the United States and elsewhere, human populations and activities have been disproportionately distributed towards coastlines, leading to markedly increased N inputs to coastal receiving waters. Nitrogen inputs to coastal waters come from the land, from the sea, and from the air; because of these multiple sources and the complexity of the N cycle, confident estimates of total N loading to coastal systems are not routine. Ecological problems from increasing inputs of N to coastal waters are well known and arise from stimulation of algal growth. There is, however, a great diversity in coastal systems (estuaries, small and large embayments, lagoons, open shelfwaters, and semi-enclosed coastal seas) and vulnerability to increased N loading varies greatly. The combination of uncertainties in characterization of loading and variability in response together have hindered development of predictive N loading–ecological response relationships and, in part, have engendered a case-by-case approach to defining protective limits for N loading for coastal systems.

Evidence for a causal relationship between N loading and a variety of ecological effects is strong. The general pattern for plankton responses to N inputs is non-linear, with sharpest effects at lower loading rates and progressively shoaling to a point where other factors (e.g., light, physical controls) become more limiting. Related effects of increased N loading include depressed levels of dissolved oxygen (hypoxia and anoxia) which injure or kill sensitive biological species; decline or elimination of submerged aquatic vegetation (SAV) which provides critical habitat for some fish and shellfish in shallow areas; promotion of certain algal species that are harmful because they produce toxins; and other effects on coastal food webs and fisheries. Evidence for effects comes from multiple observations in individual ecosystems over time, comparative analyses across many systems with different loading rates, empirical and simulation modeling, and from field-scale or mesocosm experiments. The progression of ecological symptoms from increasing levels of N is generally predictable, but the precise levels of loading that promote a certain effect vary across systems.

This chapter updates the 1st edition (2001) by reflecting on emerging 21st century literature, which indicates this is a frenetic research area. Increasing N inputs to

the coastal zone have been confirmed in more places; symptoms of over-enrichment continue to be reported worldwide and apparently are growing. Fundamental similarities between coastal over-enrichment and lake eutrophication are broadly recognized, although finer details are still debated. We cannot yet scale the effects of N loading to be predictive of effects, but this goal may be within our reach. Such a scaling would consider: water residence time, system bathymetry and morphology, water column stratification, regional biogeography and landscape setting, water clarity and light penetration, food-web structure/history, and would recognize multiple limiting nutrients. The emerging research perspective has matured to appreciate how the problem is interwoven with a host of environmental and societal issues that converge in the coastal zone.

1. INTRODUCTION: NITROGEN AND COASTAL MARINE SYSTEMS¹

In the late 1960s the problem of nutrient inputs to freshwater systems, and scientific debate about it, was reaching a peak. Although a critical volume on aquatic eutrophication edited by Gene Likens (1972) did not exclude papers on marine and estuarine systems, the principal focus was clear (Likens 1972, p. vi):

“I hope that the information provided in this volume may be useful to decision-makers and thus contribute to a slowing of the eutrophication rate in our *lakes and streams*.”

Even before the freshwater decision-makers became fully focused on setting limits on phosphorus (P) loading by a seminal experiment of Schindler (1974); Ryther and Dunstan (1971) suggested that N played the more critical role in coastal marine systems. We now know that there are low-salinity portions of some estuaries, as well as some special individual marine cases, which are P-sensitive (e.g., Howarth, 1988; Krom et al., 1991; Doering et al., 1995; Malone et al., 1996; Boynton, 2000). Similarly, there are various freshwater systems that are N-sensitive (e.g., Elser et al., 1990). Moreover, it is clear that there are possible ecological consequences of relative availability of silicate (Si) and N in coastal marine systems (e.g., Officer and Ryther, 1980; Ryther and Officer, 1981; Doering et al., 1989; Conley, 2000; Rabalais et al., 2000). But the overwhelming evidence, including

¹This chapter is only slightly revised from the 1st edition (2001). An “Afterword (2007)” (Section 6) has been included to discuss recent research trends and perspectives on the problem in coastal systems.

The term “coastal marine” or “coastal systems” is used in this chapter as shorthand for estuaries, shallow embayments, and lagoons, as well as more open near-shore and shelfwater ecosystems along oceanic-terrestrial margins, as distinct from similar coastal systems on large inland seas like the Great Lakes.

observational evidence and controlled mesocosm experiments, confirms a central role of N in establishing biological responses to nutrient loading in coastal waters (Howarth, 1988; Doering et al., 1995; Oviatt et al., 1995; NRC, 2000).

Issues of shifted element ratios and their moderation of N effects are fascinating and substantial (cf. Nixon et al., 1980; Nixon, 1981; Doering et al., 1989; Rabalais et al., 2000; Seitzinger, 2000). But it is challenging enough to review effects relating to different levels of N, which is the principal focus of this chapter.

Nitrogen circulation through the global environment has been growing dramatically (Vitousek et al., 1997). Delivery via rivers to receiving systems has increased greatly in the last half of the 20th century (e.g., Howarth et al., 1996; CENR, 2000; and other chapters in this volume). Unlike P, there is a significant atmospheric pathway for N. Local- to longer-range atmospheric transport and deposition to many regions, including coastal waters, has risen as a consequence of fossil fuel burning, agricultural practices, and other human activities. Riverine and atmospheric enrichments, combined with burgeoning coastal zone populations of humans and their associated land use changes and wastewater releases, make coastal systems a frontline receiving system for much of the enormous global N enrichment experiment now in progress. What is happening in coastal ecosystems and what do we expect to happen if N loads continue to rise?

In the same volume cited above (Likens, 1972), Clifford Mortimer gave some opening comments (p. viii), which provide a fabric for this chapter:

“Although the analogy can not be pressed too far, it will be noted that the research aimed toward cure of cancer-like eutrophication, a problem of unwanted proliferation of cells—devotes considerably more effort and resources to the study of cell processes than to description of symptoms. Until now, the reverse has been the case in eutrophication research.”

It is not that recognition of coastal marine eutrophication had its genesis in 1971; indeed it began at least at the latter part of the 19th century (cf. historical perspectives of Nixon et al., 1986; Nixon, 1992, 1998). But the paper of Ryther and Dunstan (1971), with work started much earlier by Ryther (1954), was an epiphany which did spur an emphasis on N research that has continued and increased for the last three decades (Nixon, 1995). At the beginning of the 21st century, we now have very little problem identifying Mortimer’s “symptoms” of N overenrichment, due to innumerable related conferences, symposia, and associated volumes, articles, books, and virtually uncountable numbers of grey-literature reports (Lauff, 1967; NAS, 1969; Nielson and Cronin, 1981; Boynton et al., 1982; Carpenter and Capone, 1983; Chesapeake Bay Program, 1983; Kennedy, 1984; Valiela, 1984; Rosenberg, 1985; Nixon et al., 1986; Kullenberg, 1986a; Howarth, 1988; Nixon, 1988; GESAMP, 1990; Elliott and Ducrottoy, 1991; Vollenweider et al., 1992; NRC, 1993; Nixon, 1995; Bricker and Stevenson, 1996; Nixon et al., 1996; Anderson and Garrison, 1997; NRC, 2000; Hobbie, 2000). At core, symptoms are similar to those

noted early on by John Ryther in the oyster beds and waters adjacent to the N-polluting duck farms of Long Island.

This chapter emphasizes that we now know quite a bit about coastal eutrophication² and Mortimer's "proliferation of [autotrophic] cells." We have struggled to provide good loading estimates and nutrient budgets. We have observed, probably in tens of thousands of places and in more than millions of samples, the concentrations of nitrogenous nutrients in coastal waters. There has been strong progress made in connecting inputs, concentrations, and effects *in concert* in selected places. But we cannot yet predict the stimulation of some of the most undesired cells (e.g., toxic dinoflagellates) or the precise point at which adverse secondary consequences of cell proliferation will occur in any given system. And we do not have a *generalized and quantitative* description of adverse effects of N loading, in part because there are a wide variety of coastal systems.

Also complicating the picture of N as a pollutant is that to some level and to some beholders, the effects of N loading are desirable. This is the agricultural paradigm so effectively written about by Scott Nixon – the notion that fertilization enhances productivity and leads to higher yields of desired species (see Ketchum, 1969; Sutcliffe, 1972, 1973; Sutcliffe et al., 1977, 1983; Nixon et al., 1986; Nixon, 1988, 1992, 1995). Nature repeatedly has shown that it can produce other than desired results, but we do not precisely know the *positive* limits of fertilization. Notwithstanding, the basic ingredients of the recipe for "enriching the sea to death" (Nixon, 1998) are known. Observations over the past few decades indicate that many individual system's limits have been passed to realize an oxygen-depleted, mortality-inducing recipe. Examination of such cases, among other lines of evidence, should help resolve a fundamental question: when and where does that unfortunate death recipe result? More subtle effects than fish kills occur, so there are related fundamental questions: when and where does N stimulate undesired species changes or wholesale food-web shifts?

The state of knowledge is such that it does not yet allow us to answer the above questions to satisfaction for many systems. This is unfortunate, for such answers are critical to an ability to set N limits that would be protective. There is, however, a huge, and growing, world literature; this review draws heavily from it, even as it reflects my own experience and an admittedly US/north temperate bias.

²Nixon (1995) offered a definition of coastal eutrophication as "an increase in the rate of supply of organic matter to an ecosystem." It was offered, in part, because the term has had considerable ambiguity in usage and to emphasize that it is a process, not a state. In context, "eutrophication" does not necessarily equate to "undesired effects." In fact, Nixon suggested the definition to be "value neutral." Accepting it, one should talk of the "consequences" of eutrophication as part of the possible set of responses to, or effects of, N enrichment.

2. SYMPTOMS OF NITROGEN ENRICHMENT

A recent summary (NRC, 2000) lists commonly known ecological responses to N enrichment. These include:

- Increased plant biomass and primary productivity
- Increased oxygen demand and hypoxia or anoxia
- Shifts in benthic community structure caused by anoxia and hypoxia
- Changes in plankton community structure caused directly by nutrient enrichment
- Stimulation of harmful algal blooms (HABs)
- Degradation of seagrass and algal beds, formation of macroalgal mats
- Coral reef destruction

Also listed as a concern, not with N *per se*, but with one vector for it, human sewage, is a potential increase in disease and pathogen species. With the exception of coral reefs, and the substitution of a term like “nuisance” for “harmful” and “macrophytes” for “seagrass,” these effects are the classic symptoms of lake eutrophication (Wetzel, 1983).

I have grouped effects for review and discussion in this chapter into five prime categories of response to N loading. These have served generally as focal points and endpoints for research:

1. Chlorophyll
2. Phytoplankton primary production
3. Dissolved oxygen (DO)
4. Benthic producers (SAV, macroalgae) in shallow waters
5. HABs, as part of change in phytoplankton species composition

After a short discussion on N loading (Section 3), I examine each of these five fundamental effects of concern, using examples of where they have been noted and/or have been increasing (Section 4). The probable role of N is suggested, and I try to capture the different kinds of evidence that can link it to the problem. Evidence includes what I refer to as “epidemiological” associations, a spatio-temporal co-occurrence, either local or regionalized. Other evidence includes: time trends of N and effects observed at individual or multiple sites; empirical patterns that emerge from comparing conditions across sites, which begins to assess the generality of the coupling between input and response; and finally, experimentally observed linkages (primarily in microcosm or mesocosm³ experiments), which help confirm and in some cases quantify the nature of the relationship. The strength and kind of evidence

³Mesocosms are considered to be contained systems (tanks, ponds) larger than bottle or laboratory-size (i.e., “micro”-cosms), which capture some or many of the environmental features and realism of a natural system (usually outside exposed to natural

linking N and each problem varies, but to the degree possible I indicate some situations where a quantitative linkage has been established. In summary (Section 5), I speculate on the quantitative sequence of events with increasing loading.

The five effect categories are compiled in a very simple conceptual model to frame how symptoms relate to N loading (Figure 1). I have not attempted to include all the ecological components, flows, confounding factors, etc. in a spaghetti-like picture of interactions that captures more of the true complexity of “sophisticated” constructs or model formulations. Briefly, water column chlorophyll, phytoplankton primary production, and other algal increases are viewed as a direct, nutrient uptake response. Algal increases, representing increased levels of organic matter, secondarily promote low DO through increased decomposition and respiration. Increased algae shade SAV in shallow water to produce a secondary effect of seagrass decline through light reduction. Competitions among the algal community may ultimately promote toxic or nuisance blooms of harmful algae. A concert of secondary effects acts further on food webs/fisheries, but even the direct and first-level indirect effects of N loading (Figure 1) have been difficult to quantify broadly. Sections 5 and 6 discuss some ramifications of these effects, which have consequence to esthetics, human health, valued estuarine and marine populations, food webs, diversity, and ecosystem sustainability (CENR, 2000).

Any consideration of coastal systems and their potential responses (Figure 1) must also recognize some special, complicating aspects. These systems are generally very open to flow of water and materials, including organisms, from both “upstream” and “downstream” sources (due to tides and circulation changes, as well as biological transport or active migration). Most coastal systems have many subareas and pockets of different habitats, so *spatial and temporal variability is a confounding problem in their fundamental ecological characterization and in definition of their response to inputs*. Coastal systems also represent a set of fairly bewildering diversity in size, shape, and other physical, chemical, and biological characteristics. Monbet (1992) suggests that responses “vary from estuary to estuary, from segment to segment within a given estuary, and from time to time within any segment of an estuary.” Perception of estuaries each as unique is echoed through the literature. The notion of “yes, but that doesn’t hold for my system,” is a common one and is bolstered by recognition that “the extreme variation in response to any level of loading clearly demonstrates the importance of other factors that determine differences between estuaries” (NRC, 2000). Continued intensive field studies and site-specific

lighting). Systems are usually replicated and manipulated for controlled experiments. Example systems, cited in this chapter in relation to nutrient enrichment experiments, include the MERL (Marine Ecosystem Research Laboratory) systems (2.63 m² area, 5 m deep, with coupled pelagic and soft-bottom communities; cf. Nixon et al., 1984, 1986) and several shallow pond/lagoon/tank systems used for macrophyte or seagrass studies (e.g., Twilley et al., 1985; Short, 1987; Short et al., 1995; Taylor et al., 1995a, b).

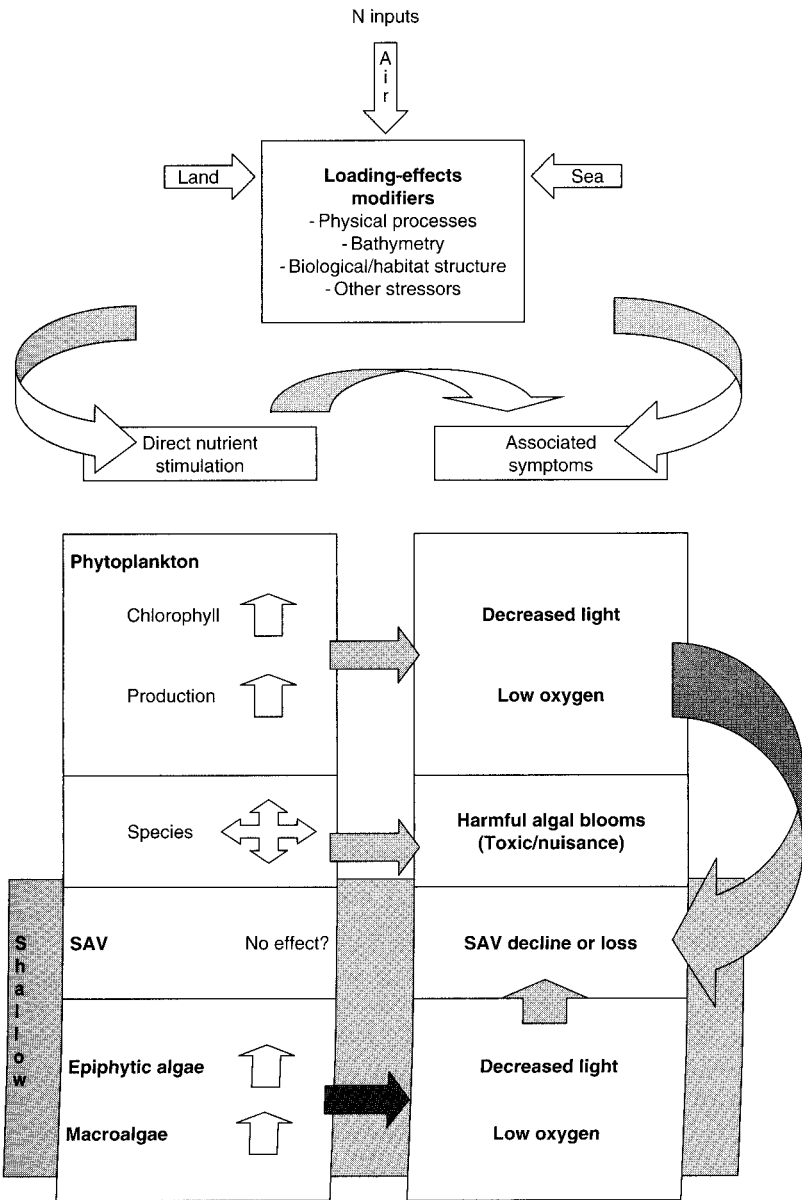


Figure 1. Conceptual model of N loading and effects discussed in this chapter.

modeling of select systems will undoubtedly reveal much more we should know in terms of N and ecological responses. There also has been a strong recognition that we cannot study each of the many thousands of systems intensively. We need to be able to group similar systems in terms of their vulnerability to N enrichment. The notion of ecological classification is thus in the vanguard of the attempt to aggregate across the diversity of systems and to develop more generally a quantitative relationship of coastal responses to enrichment (cf. Jay et al., 2000; NRC, 2000).

3. NUTRIENT LOADING TO COASTAL SYSTEMS

3.1. Multiple Sources, Uncertainties, and High Nutrient Loading

To quantify effects and develop N loading–biological response relationships, one needs to start with nutrient inputs. Quantifying inputs to coastal systems has not been a small task, for several reasons. The possible sources of N are many. Obvious point sources were initially and easily tracked, but usable methods and models to assess non-point N surface flows, as well as atmospheric deposition, have taken considerable effort to develop and apply. Offshore exchanges and groundwater inputs are still not easily or routinely assessed. Moreover, all sources have changed markedly in a brief span of history. In addition, there are many forms to analyze, which contribute to “total” N (TN = ammonia, nitrate, nitrite, dissolved organics, particulate matter, organic, and inorganic). Now standard analytical methods were not all standard until after 1970 and raging debates have ensued as to whether only dissolved inorganic N (DIN) loading, or also organic N forms, are stimulatory nutrient sources. So, estimating TN (like total P for lakes) has not always been the goal of those assessing loading or responses; many examples cited here use DIN. In addition to these factors, many coastal systems are large, so spatial and temporal assessment of sources is not a small matter.

Even with incomplete N budgets and sources not as well characterized as for some other systems, we have known for some time that coastal systems receive high nutrient loading. Figure 2 shows estuarine systems for which land-derived inputs were summarized in the mid-1980s. Coastal systems often integrate flows and inputs from large watersheds, so from their position in the landscape, we could expect many of them to receive relatively high nutrient inputs compared with other systems. There is a significant range in loading for coastal systems, but it is not as wide as observed in lakes. Eutrophic/hypereutrophic lakes reach the same high levels of loading as many estuaries, but oligotrophic lakes are far less enriched (up to over two orders of magnitude lower). Less-enriched systems, such as some lakes and forests, tend to receive relatively high N loads (and thus have a higher N/P input ratio), because a majority of their input is from atmospheric sources (Kelly and Levin, 1986). Many estuaries receive inputs from terrestrial sources at rates well above those applied to intensively fertilized agricultural fields.

In spite of the difficulties of source assessment, we now believe we have good input budgets for TN (and DIN) for a few coastal systems. The most complete loading estimates have a smattering of measurement, modeling, averaging across years,

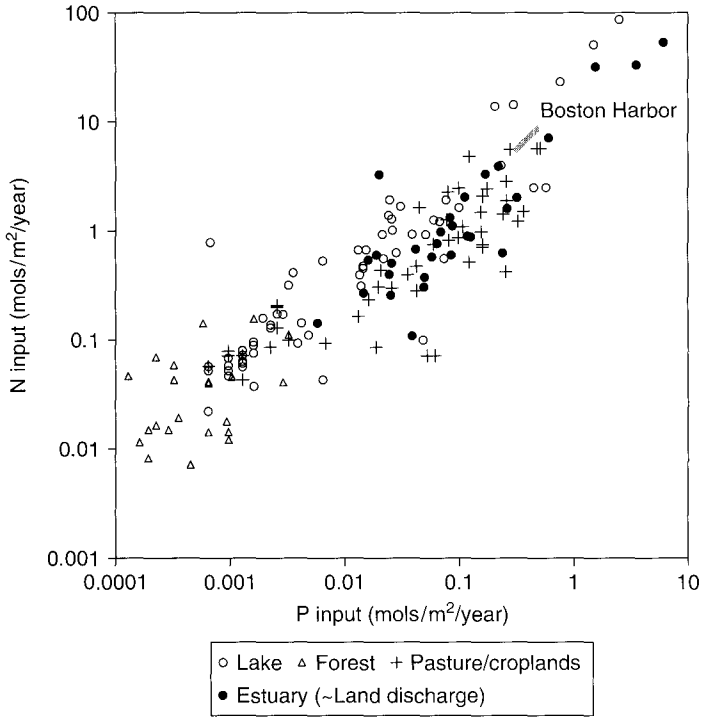
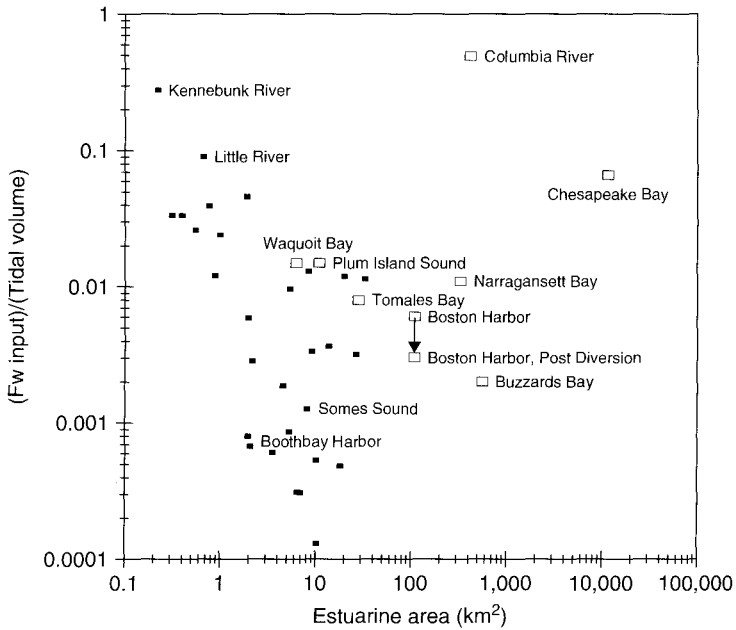


Figure 2. Nitrogen and phosphorus input to a variety of terrestrial and aquatic ecosystems. Modified from Kelly and Levin (1986). Estuarine systems show DIN or TN input from land and in some cases, atmosphere. Boston Harbor is from land-derived sources only (Kelly, 1997a), showing an example range for a system considering only DIN (lower end of bar) or TN (upper end of bar).

and some measure of best professional judgment (e.g., Nixon et al., 1995, 1996). Compared with values in the summary of Figure 2, total inputs are probably higher for most coastal systems, in part due to inclusion of several sources that were not well known or quantified in the mid-1980s. For example, atmospheric inputs are substantial to some systems (principally larger, more open-water ones), and also have been increasing (e.g., Paerl and Whitall, 1999). Groundwater inputs have also been quantified and are significant in certain systems (e.g., Valiela et al., 1997a).

Most recently, in the course of developing complete coastal nutrient budgets, it has become broadly recognized that loading from the seaward, as well as the landward, edge can be very substantial (Garside et al., 1976; Nixon, 1997; Kelly, 1998; Sigleo et al., 2005). Boston Harbor, Narragansett Bay, and other northeastern US systems are an appropriate region to focus on ocean inputs because of large tidal ranges and, in comparison, relatively low freshwater inputs (Figure 3a). For example, Boston

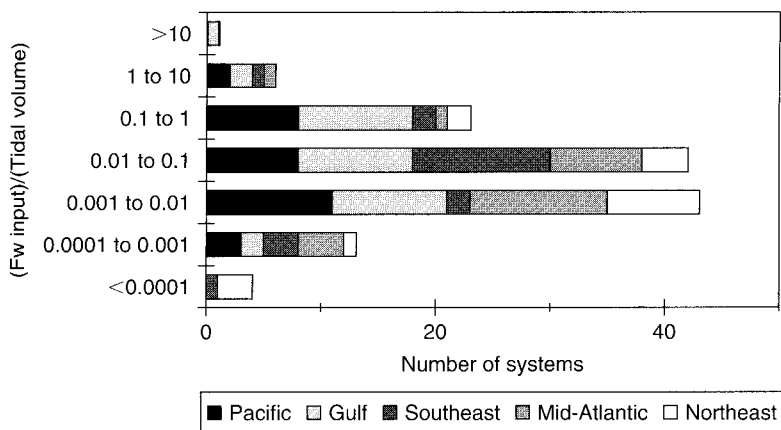


(a)

Figure 3a. Freshwater (Fw) volume: tidal volume ratio in small and large coastal ecosystems. Data are from a summary of Maine (closed squares) and other (all open squares) larger northeast systems (Narragansett Bay, Boston Harbor, Buzzards Bay) by Kelly (1997b), along with intensive coastal LMER sites around the United States described by Jay et al., 1997. Systems of different size range from river- to ocean-dominated.

Harbor has a freshwater to tidal volume ratio <0.01 . At this ratio, the concentration of N in freshwater must be ~ 100 times that in the tidal floodwater to provide equivalent loading; even with Boston's large effluent discharge to the Harbor (now being diverted offshore) this turns out not to be the case. Inclusion of ocean loading to the budget based only on land and atmosphere sources raises Boston Harbor's N input estimate by $\sim 100\text{--}200\%$ (DIN and TN, respectively; Kelly, 1998). Many systems do not have the direct wastewater load of Boston and many have Fw/tidal volume ratios far less, indicating greater potential for ocean-domination of loading. Not all of the tidal volume input actually mixes with the water within an embayment, and this must be accounted for to assess ocean loading. The ratio nonetheless is a first-order indicator of relative source strengths. Figure 3a suggests why several systems have "River" as part of their name and that Chesapeake Bay is much more freshwater-driven than many northeastern sites.

The northeast and its macrotidal conditions appears to be skewed to Fw/tidal volume ratios less than 1, whereas other US geographic regions have a distribution that includes ratios >1 , or even >10 (Figure 3b). There is considerable overlap in the frequency distribution for each region; clearly the potential for both river- and ocean-dominated flows exists in all regions. The significance of ocean loading as a nutrient source will vary with the offshore N concentration, which may show a general decrease with latitude. Increased atmospheric N deposition (e.g., Prospero et al., 1996; Paerl and Whitall, 1999) directly to adjacent near-coastal waters could increase the role of the oceanside source of N to estuaries and embayments. These uncertainties reinforce the notion that we are still learning to quantify sources of nutrients to many coastal systems, and that source characterization is a big factor that has limited the development of loading-effects relationships.



(b)

Figure 3b. Frequency diagram, by US geographic regions, of the ratio of freshwater volume to tidal volume input. From data compiled by NOAA estuarine susceptibility/eutrophication survey (see Bricker et al., 1999), using tide gauges near mouths of estuaries. Pacific N = 32, Gulf N = 35, Southeast N = 20, Mid-Atlantic N = 27, Northeast N = 18.

3.2. 20th Century Trend of Increasing Nitrogen Concentrations and Loading to Coastal Systems

Human populations along coastlines have been dramatically increasing with global population rise, as have associated anthropogenic pressures on coastal systems. Recent increases in loading to coastal systems are rather spectacular in some cases, but they are also just part of a general global increase in N circulation throughout the atmosphere and terrestrial as well as aquatic ecosystems

(e.g., Nixon, 1995; Howarth et al., 1996; Prospero et al., 1996; Vitousek et al., 1997; other chapters in this volume).

For a number of coastal systems, it has been possible to measure or reconstruct trends of increase in N loading or *in situ* water column N concentrations over several decades or even from pre-European settlement in the United States. A few examples, all with markedly increasing N trends, include: the open and coastal Baltic Sea since the 1950s and 1960s (Elmgren, 1989; Cederwall and Elmgren, 1990; Rosenberg et al., 1990), Narragansett Bay/Albemarle-Pamlico Sound since the 1800s (Nixon, 1995), the Ythan estuary in Scotland from 1960s to 1990s (Balls et al., 1995), the Mississippi River plume from the 1960s to 1990s (Rabalais et al., 2000), and Chesapeake Bay from 1950 onward (Boynton, 2000). Conley (2000) compared several of these published trends in the United States and Europe; he summarized that N loads have increased by a factor of 1.5 to 4.5 over the 20th century and are presently as much as 60 times more than what might be judged as "pristine" condition loading.

3.3. Modifiers of Nitrogen Loading that have Consequence for Expression of Effects

Like other aquatic systems, coastal systems experience multiple stressors. When we look for nutrient-related effects there can be confounding problems from suspended solids, toxic contaminants, and habitat loss. But even in cases where we think we know all the N inputs and other stressors, we do not necessarily know much. There are a number of features within coastal systems that modify how and when nutrients reach biological receptors and in essence create the N "exposure." Principal among these modifiers is flushing and the residence time of water within the system; this feature is an emphasis of this review. Other physical features, like stratification, are also significant. Additionally, work of Seitzinger (2000) and Nixon et al. (1996) show that sediment microbial denitrification converts DIN to N₂ gas, removing >25% of N loading to longer residence time systems. Also, larger biological organisms modify the distribution of N forms, spatially or seasonally, or graze upon plankton and affect the way primary producers respond to nutrients. As will be discussed, many features complicate relationships between N loading and effects, in part by affecting the concentration experienced at a given loading rate in different systems.

4. LOADING-RESPONSE RELATIONSHIPS

4.1. Chlorophyll Response to Nitrogen Loading and Concentrations

Nutrient inputs (especially N) generally stimulate plankton biomass in coastal systems, and this is a first response in the sequence of related effects (Figure 1). The response is regularly measured in terms of chlorophyll *a*. Considerable evidence for stimulation exists at all levels of ecological organization and complexity (cf. Hecky and Kilham, 1988). Studies note enhancement by N additions in axenic cultures, community (bottle) assays whole-system enclosure/mesocosm experiments, and

natural systems. The latter is inferred from historical trends, taking advantage of natural system "experiments" (e.g., sewage increases or diversions), and comparative trend analyses for many coastal systems (e.g., Boynton et al., 1982; Nixon, 1983; Nixon et al., 1986; Howarth, 1988; Nixon, 1992).

In examining empirical patterns to develop quantitative relationships across whole systems (experimental or natural), an interesting challenge is the characterization of chlorophyll concentration, which has very high space and time variability. Should we be looking at peak (individual sample) concentrations, mid-summer ranges, or annual depth-integrated/spatially averaged means? The most successful efforts to relate N and chlorophyll have been constructed using annual means and spatial averaging across a range of sites when possible. Where data are too infrequent or poorly spaced (in time or across the estuary) the estimate of a systems value may be miscast and create variability for pattern analyses. When we look across time or across systems we must constantly ask: are data summaries comparable and reliable, and how wide are the bounds of the estimate?

Increasing chlorophyll concentrations over years to decadal or greater time scales have been observed at very many sites around the world in the last half-century. In many cases a rise in benthic microalgae or phytoplankton chlorophyll has been correlated with N concentration increases. To generally summarize from many studies: when viewed across sites along enrichment gradients within or between ecosystems, a basic pattern often emerges between planktonic chlorophyll and water column DIN concentrations. Over a range of annual average DIN from <1 to $>20 \mu\text{M}$, chlorophyll tends to rise less than $1 \mu\text{g/L}$ of chlorophyll with every $1 \mu\text{M}$ increase in DIN; about $0.7\text{--}0.8 \mu\text{g Chl}/\mu\text{M DIN}$ is a very rough rule. There is variability and a tendency for the chlorophyll rise to be below the rough rule at increasingly higher DIN levels, such as may be found *within* a given coastal system near sewage treatment or other strong point source of nutrients. Observations such as this have been used to suggest light limitation at very high nutrient levels (e.g., Malone, 1982; Monbet, 1992).

Many of these generalizations can be seen in Figure 4, which also adds a dimension of classification to the trends. The parameter range is broad enough that it has to be viewed on a log scale. The increasing general trends and variability noted above are nonetheless apparent, but for two fairly distinct classes of systems – those which have very large tidal ranges (macrotidal) and those which have smaller (microtidal). Microtidal systems appear to be more sensitive to N enrichment, judging by a higher chlorophyll level observed at any given N level. Tidal energy may produce effects upon the light received by plankton by increasing vertical mixing. Destratification, sediment resuspension, and flushing may all reduce the chlorophyll *a* response per unit N.

A number of studies, Monbet's included, have tried to relate chlorophyll with *N loading*, not just *in situ* N concentrations, with differing degrees of success. No doubt this is due to underlying variability in the nature of different systems (such as suggested by Figure 4), as well as uncertainties in both loading and response measurements. Efforts generally have confirmed a strong correlation to N loading

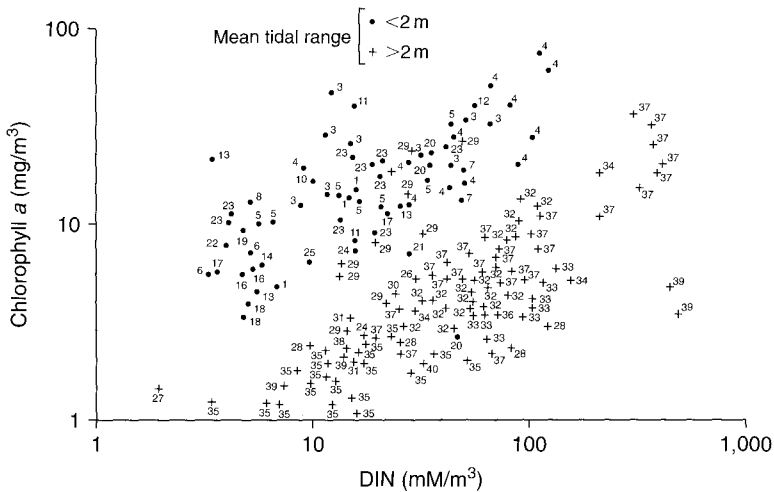


Figure 4. Mean annual concentrations of DIN and chlorophyll at multiple sites within different estuaries. Redrawn from Monbet (1992). Units are equal to μM (DIN) and $\mu\text{g/L}$ (chlorophyll). See original reference for coastal systems (represented by a series of similarly numbered stations) and data sources.

(versus P), and provide evidence of a general relationship, but not necessarily a satisfyingly predictive one. Some issues of comparability and reliability inherent with empirical trends observed from cross-system correlations have been overcome by whole-system mesocosm experiments. For example, Marine Ecosystem Research Laboratory (MERL) enrichment gradient experiments (e.g., Nixon et al., 1986; Keller, 1988a, b) show an unequivocal tie between N loading and chlorophyll standing stock. Data show both a general increase in mean annual chlorophyll and an increase in the overall range of instantaneous measurement variability within increasing nutrients. Nixon et al. (1986) showed a strong relationship between annual DIN inputs, annual average *in situ* DIN concentrations, and chlorophyll over the following ranges for DIN (~ 5 – $300 \mu\text{M}$) and chlorophyll (~ 3 – $75 \mu\text{g/L}$). Marine and coastal systems, for which there are comparable loading and chlorophyll data, follow the general MERL trend, with exceptions noted by Nixon (1992) that are observable in Figure 5. The data overall (log-log scale) suggest a hyperbolic relationship familiar from bottle assays. Chlorophyll, although it continues to increase with additional nutrients, does not keep pace 1:1 with increasing nutrient loads or concentrations.

Nixon, Oviatt, and their co-workers' efforts to conduct experiments and compare results with natural systems have provided strong quantitative evidence of the relationship between N and chlorophyll. Scatter in the available data, however, suggest a single empirical relationship may not apply as a strong site-predictive model

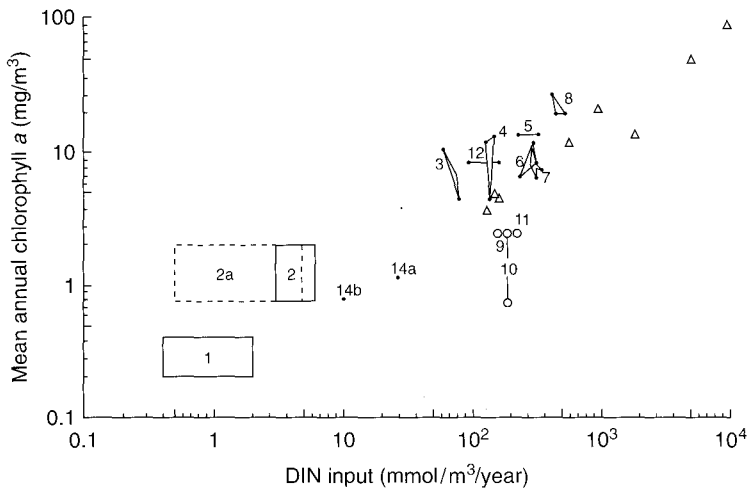


Figure 5. Nitrogen loading and chlorophyll response in a mesocosm study compared to a range of coastal and marine systems. Redrawn from Nixon, 1992. Triangles are data from a MERL mesocosm enrichment experiment. Numbered points or polygons are natural field systems. The rectangles (1, 2) are open sea systems (Sargasso Sea, North Central Pacific). Systems 9, 10, 11 are continental shelf and upwelling areas. The remainder includes estuaries or bays (e.g., Kaneohe Bay [HA], 14a/b [before/after sewage removal]; subestuaries of Chesapeake Bay, 3, 4, 6, 7, 8). See original reference for systems and data sources.

unless we improve in normalizing for other critical variables that will also influence the response. For example, all else being equal, shorter water residence time systems (including those more energetically flushed by tides) will tend to have lower DIN concentrations for a given N loading (Kelly, 1997a, b), so that the problem of chlorophyll “sensitivity” to loading (versus *concentration*) in different systems is further complicated. Boynton and Kemp (2000) have tried a “primitive” scaling of nutrient loading (correcting areal input for hydraulic fill time and depth, as has been successful for lakes [Vollenweider, 1976]). Interestingly, their significant linear regression using mean chlorophyll and “scaled nutrient loading” for various Chesapeake Bay sites (and a few others) begins to suggest that physically different systems can be better aligned along similar response trends if properly normalized.

The range of chlorophyll concentrations (e.g., Figures 4 and 5) is largely what would be considered mesotrophic (mean $4.7 \mu\text{g/L}$, range 3–11) to eutrophic (mean $14.3 \mu\text{g/L}$, range 3–78), with some hypereutrophic (range 100–150 $\mu\text{g/L}$). This judgment applies the standard lake classification (see Vollenweider, 1976; Wetzel, 1983; NRC, 1993). Few coastal marine examples would be oligotrophic (by the lake standard: mean $1.7 \mu\text{g/L}$, range 0.3–4.5); Kaneohe Bay (tropical, microtidal) or offshore

areas of larger, open-water temperate macrotidal systems (e.g., Buzzards Bay [MA], Bay of Brest [France] might qualify). More fully marine systems, like mid-ocean gyres (Sargasso and North Pacific) as well as some continental shelves some distance from land also have annual mean chlorophyll $<2 \mu\text{g/L}$ and also qualify as oligotrophic. The relative lack of low-N/low-chlorophyll coastal marine systems, at least among those that have been more extensively studied, has implications for trying to observe effects from N loading, and this is next put into perspective through examination of productivity-nutrient loading trends.

4.2. Productivity Response to Nitrogen Loading

It is important to derive a relationship between primary production and N loading, because *in situ* productivity in large part sets the system's organic supply and establishes a potential for metabolic effects (i.e., low DO, Figure 1); organic supply has been cast as a prime basis for establishing "eutrophication" classes (Nixon, 1995). The ability to quantify the productivity-loading relationship suffers from the same difficulties faced in relating chlorophyll and N. Year-to-year variability in production in coastal systems can be considerable even with fairly constant loading rates, a situation that limits site-specific predictability. Variability occurs because many factors besides nutrient loading can moderate production processes and response to enrichment (e.g., cloudiness, climate, water stratification and circulation; *in situ* physico-chemical properties; as well as grazing rates and biological structure). Nonetheless, a relationship should be at least broadly evident because phytoplankton production correlates well with chlorophyll biomass. For example, Keller (1988a) provides an empirical regression between annual productivity (P_y , g C/m²/year, using the ¹⁴C technique) and mean annual chlorophyll biomass (B , mg Chl *a*/m³) for data from about nine natural systems and a MERL experiment:

$$\begin{aligned} P_y &= 95.4 (\pm 20.2) + 13.0 (\pm 1.0) B, \text{ with } n = 20, \\ r^2 &= 0.91, \text{ standard errors in parentheses.} \end{aligned} \quad (1)$$

Using MERL studies and extensive data for Narragansett Bay from 1978 to 1983, P_y was correlated to a composite parameter (Keller 1998b; following Cole and Cloern, 1987) that recognizes not only the influence of B , but incorporates the influence of the depth of the photic zone (Z_p) and incident light (I_o). The resulting relationship was:

$$\begin{aligned} P_y &= 25 (\pm 10) + 0.3 (\pm 0.02) B Z_p I_o, \text{ with } n = 32, \\ r^2 &= 0.92, \text{ standard errors in parentheses.} \end{aligned} \quad (2)$$

For the last century, we have known there is a connection between nutrient inputs and plankton productivity for marine systems (cf. Johnstone, 1908 or several historical considerations of productivity [Nixon et al., 1986; Nixon, 1992]). Even

so, the relationship between N input and productivity has only comparatively recently been quantified, and this can best be seen in a succession of progressive efforts reported by Nixon (1983, 1992) and Nixon et al. (1986, 1996, 1997). When restricted to those relatively few field systems – mostly for open shelf and open or semi-enclosed seas (i.e., the Baltic) – for which there is high confidence in estimates of *total* DIN inputs (including ocean loading) and ¹⁴C-based production, and combined with experimental MERL mesocosm data, Scott Nixon’s analyses show a trend that would suggest a strong predictive ability (Figure 6).

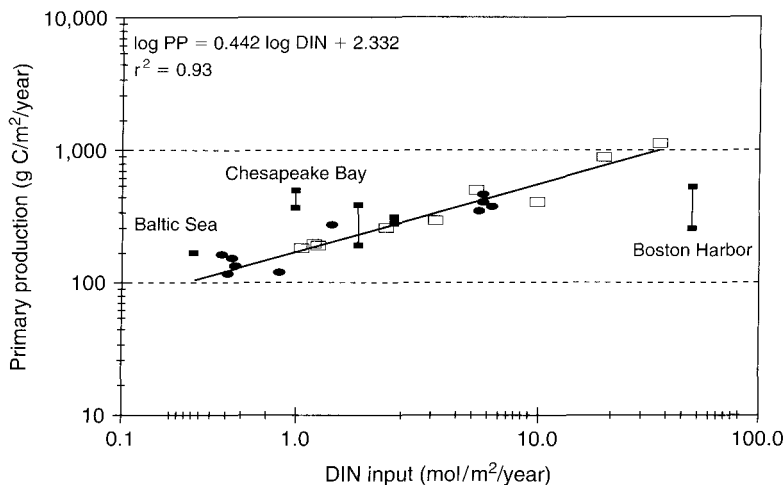


Figure 6. DIN inputs and phytoplankton primary productivity. Modified from Nixon, 1997. Open squares are the first year of a MERL experiment (Nixon et al., 1986; Nixon, 1992). Closed circles are various open marine systems, coastal continental shelf, or estuary systems. The regression shown from Nixon (1997) does not include data (all closed squares) which I have added to the original plot: outer Boston Harbor, Chesapeake Bay, Delaware Bay, Potomac River, and Baltic Sea. These additional areas have suitable data, including DIN loading estimates with offshore exchanges considered (Boynton et al., 1995; Nixon et al., 1996; Kelly, 1998; Nixon, 1992). Chesapeake uses 0.7 TN to estimate DIN, from Boynton et al., 1995 (following Nixon et al., 1996). Data for Boston Harbor are for the northern outer harbor where production measurements were made (Kelly, 1998). Plots like this attempt to derive a pattern of response to N as the primary stimulant; there are obviously other nutrients and inputs occurring in both experimental and field situations.

Nixon’s resolution of the producers’ response, because it relies on some of the most complete input budgets and productivity data, confirms a strong coupling, shows a strikingly tight trend, and has been already much cited. It draws from relatively few

systems, mostly more open coastal or marine, and the upper end is primarily driven by MERL results. Not all estuarine and coastal systems will strictly abide by it, a point illustrated in Figure 6, where I have added a few other systems that I believe have comparable and suitable data. The first “anomalous” example included on the plot is for outer Boston Harbor, a highly enriched shallow coastal embayment – one of the few such systems where all inputs, including offshore exchange have been estimated. Production in Boston Harbor appears distinctly low compared with the prediction. Kelly and Doering (1997) suggested this might be due to light limitation or a short water residence time. If the plankton doubling times are not always shorter than the water residence time, then the plankton population will be regulated by “washout” of cells to the offshore. The physics in such a case does not allow higher production because the population level to support it simply cannot accumulate. Boston Harbor stations have low chlorophyll for their nutrient levels, which is consistent with a lower than expected cell buildup and *in situ* production rate, but data otherwise seemed to follow the basic rules of chlorophyll-production relationships that apply to most other coastal waters (Cole and Cloern, 1987; Keller, 1988a, b; Kelly and Doering, 1997). We should indeed look at other factors to explain a situation like Boston Harbor, rather than suggest the anomaly invalidates a general prediction of increasing production with increasing input. On the other hand, we also know that there are upper limits on production (e.g., Bannister, 1974) and the increasing trend will not go without bounds, as self-shading by bloom conditions will become a factor (Wetzel, 1983). The MERL mesocosms operate with strong vertical mixing and a favorable light environment for plankton, so the upper treatments may be more productive than can be achieved in many field settings. However, treatments showing $>900 \text{ g C/m}^2/\text{year}$ (Figure 6) are higher than those used in the model to estimate them and seem inconsistently high compared with another measure of metabolism (Nixon, 1992), so there is reason to view them with caution. There are some natural systems with apparent loading rates that exceed the upper end of the MERL experiment (Jaworski, 1981; Monbet, 1992), but I am unaware of production estimates for them, except the Boston Harbor example.

A second “anomaly” is Chesapeake Bay. It has been noted (e.g., Boynton et al., 1982; Nixon, 1992) that this bay’s chlorophyll and productivity ranges are relatively high for the input of N, as shown on the plot. The same appears to be true for the Baltic Sea, although it is a little less pronounced.

The main trend and deviations of Figure 6 are usefully put in another perspective (Figure 7). Following an earlier paper (Kelly and Levin, 1986), I have overlain production data for freshwater systems (using P loading) with coastal and marine systems (using N loading) by rectifying the axis to a Redfield ratio (N:P = 16:1, by atoms). From the previous summary, I excluded lakes where production data included macrophytes or other producers in addition to plankton. For lakes, instead of actual N input, the x-axis is simply representing the P input times 16, which is the necessary N equivalent for the average marine or freshwater plankton tissue (e.g., Schindler, 1974; Hecky and Kilham, 1988). This approach is preferable to using actual N inputs to freshwaters,

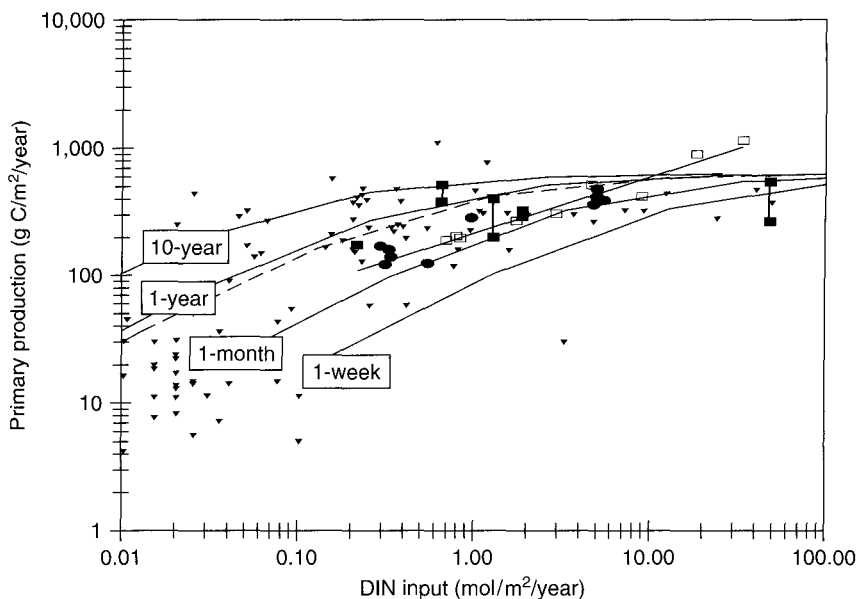


Figure 7. Production-loading response in aquatic ecosystems. Lakes (closed triangles) are overplotted on the estuarine/marine data of Figure 6, by converting P inputs to the equivalents needed by phytoplankton (Kelly and Levin, 1986). Using a classic empirical model for lakes (see text), which recognizes the influence of water residence time and depth, the plot shows lines of predicted production for conditions, as a function of nutrient input on an areal basis. The family of solid curves shows different water residence times for a 5-m water depth. The unmarked dotted curve below the 1-year, 5-m projection represents a 20-year, 50-m condition, such as the Baltic Sea.

since lakes are usually responsive to P and can make up for N deficiencies by N fixation, which is often not measured as a loading term (see Howarth et al., 1988).

Figure 7 also shows predictions of pelagic primary production (*PP*) based on the empirical lake model of Vollenweider (1979 – see Wetzel, 1983). This is of the form:

$$PP(g\ C/m^2/year) = 6.985 (X^{0.76}) / (0.29 + 0.11X^{0.76}) \tag{3}$$

where

$$X = [P_i] / (1 + \sqrt{T_w}), \tag{4}$$

based further on P_i =average P inflow concentration and T_w =average residence time. X is the "expected" or predicted P concentration for the water body (see Wetzel, 1983).

Using this empirical model, aquatic production has been forecast in Figure 7 for several water residence times (1 week to 10 years) and a standard depth of 5 m, typical of many coastal systems. A projection is also shown for a longer residence time (20 years) and greater depth (50 m) appropriate for some lakes and the Baltic Sea. The PP prediction is actually based on X which estimates in-lake concentration, not loading. From it, one can back-calculate to units of areal loading and plot results consistent with the standard expression that has been used in marine studies (Figure 6). The formulation, within the bounds of parameters chosen, encloses most of the lake data (from which it is generally derived); use of longer residence times would include virtually all of it.

There are a number of messages to be gleaned from this exercise. First is the notion pointed out above with respect to Figures 2, 4, and 5 – that coastal systems generally receive high nutrients and are not like oligotrophic lakes. The more rapid rise in production that is apparent from the least-loaded freshwater systems cannot be assessed for coastal systems because no such poorly loaded ones are studied on this scale. Even a prehistoric input estimate to Narragansett Bay, about 0.27–0.33 mols N/m²/year (Nixon, 1997), is an order of magnitude higher than equivalent loading to oligotrophic lakes.

Second, it has often been noted that the range of production in marine coastal systems is not very large, especially in comparison with loading (e.g., Nixon and Pilson, 1983; Nixon et al., 1986; Oviatt et al., 1986; Nixon, 1992). The regression of Figure 6 shows the relation is non-linear and there is only a factor of 4.4 increase for each order of magnitude increase in loading. From the perspective of Figure 7, this range of production and degree of stimulation is consistent with the fact that the coastal marine systems are biased to the upper half of the general aquatic nutrient saturation curve, a non-linearity that would be more evident if the plot were not on a log-log scale. Both lake and marine studies recognize a self-shading effect begins to limit phytoplankton at very high nutrient levels. An appreciation for the difficulty of sorting out a production increase "signal," amidst the "noise" of system differences (and potential signal modifiers) is gained, and the role of the MERL mesocosm experiment in defining an unambiguous response is recognized.

Third, lakes and coastal systems may be described by the same simple rules – loading, depth, and residence time – once critical limiting nutrients are accounted for. Hecky and Kilham (1988) point out fundamental similarities in physiology between freshwater and marine algae, and perhaps it should not be surprising that the two conditions could basically follow the same model. But to my knowledge, this has never been fully recognized and Figure 7 is the first suggestion this may be so. The freshwater model curve pretty well predicts Nixon's trend for conditions of a 1-month water residence time and a 5-m water column, which is basically the configuration of the MERL nutrient gradient experiment (Figures 6 and 7). The model

does not, however, duplicate the trend for the upper two MERL points, although it is a near-perfect hit for the very enriched outer Boston Harbor (2-day residence time, 5-m depth). The Baltic Sea, with a 20-year, 50-m condition is predicted. By the model, the Chesapeake Bay (~8-month residence time, 6-m depth) should indeed have higher production than the MERL trend, and the match for the Chesapeake is close; it would improve if TN input (rather than DIN, ~0.7 TN) had been used.

There is undoubtedly room to review more coastal data and improve model formulations, but the principal lesson from Figure 7 involves water residence time. A linchpin of the lake model concept is that internal system physics modifies loading to produce different concentrations of nutrients maintained within the receiving water, to which biology responds. Evidence confirms a relationship between residence-time corrected inputs and *in situ* concentrations for open coastal systems and the MERL experiment (e.g., Kelly, 1997a). Estuarine scientists have been slow to incorporate this concept, developed long ago for lakes (e.g., Dillon, 1975). Part of the problem is that residence time in estuaries is not dictated just by freshwater throughput as it is in lakes, and it has been difficult to come to grips with this. Tidal inflow and mixing are significant and in very many cases the “freshwater residence time” is much longer than the true estuarine water residence time. Some physical oceanographers (Geyer et al., 2000) provide a perspective:

One of the most important quantities relating physics to the ecology of estuaries is residence time. A widely cited example is the work of Vollenweider (1976), who demonstrated that in lakes it is not just the nutrient loading, but rather the product of nutrient loading and residence time that determines the impact of phytoplankton production. Unfortunately, estuarine physicists have been rather unenthusiastic about attempting to quantify residence time, due in part to how easily misinterpreted a single number would be in characterizing the complex exchange processes that influence an estuary.

While “more effort should be placed in developing more accurate and sophisticated approaches to estimating residence time” (Geyer et al., 2000), there are already simple box-modeling techniques to derive estuarine water residence times useful for exercises like Figure 7 (e.g., Officer, 1980; Pilson, 1985; Doering et al., 1990; Asselin and Spaulding, 1993; Smith, 1993; Kelly, 1998; Hagy et al., 2000). Such studies, along with results in some cases of complex hydrodynamic mixing models, are the basis for residence-time corrections used by Nixon et al. (1996) in assessing estuary retention of nutrients, in characterizing the anomalies of Figure 6, and later in this chapter.

4.3. Dissolved Oxygen Response to Nitrogen

The most obvious concern of an adverse ecological effect with N enrichment is development of low DO (hypoxia) or even anoxia (no oxygen) in the water column of coastal marine systems. The fundamental conceptual model for the effect of

N on DO is simple. Nitrogen stimulates primary production (i.e., it causes “eutrophication”). At some point of stimulation, the associated respiration rate of accrued autotrophic biomass begins to exceed the capacity of the water body to replenish itself by re-aeration and equilibration with the atmosphere, and DO concentrations can fall to hypoxic or anoxic levels. A water column concentration of DO >0 but <2 mg/L is the common definition of hypoxia. Most often noted in stable bottom waters of vertically stratified systems (and thus affecting sessile benthic organisms), hypoxic/anoxic levels can also occur throughout the water column, even in vertically well-mixed conditions. It is, of course, true that DO concentrations often go to zero within several millimeters of the surface of soft-sedimentary deposits. Benthic infauna, which live in these sediments and which cannot easily move to avoid conditions, can tolerate low DO (even hypoxia) in the overlying water column. For example, Rosenberg (1980) suggested ~ 2.8 mg/L as a limit noted for coastal benthic communities, and later Rosenberg et al. (1991) lowered this limit to an overlying water exposure of 1.4 mg/L for several days to weeks, using shallow shelf organisms tested within their natural sediment environment. Many US States have long used 5 or 6 mg/L as a standard, recognizing that the lower thresholds for biological effects are higher in sensitive species and sensitive life stages (e.g., NRC, 2000), including species which live within the water column, where DO concentrations are measured. Bricker et al. (1999) recognized this in the National Oceanic and Atmospheric Administration (NOAA) survey, and thus characterized hypoxia as >0 and ≤ 2 mg/L, with >2 –5 mg/L characterized as “biologically stressful,” in an effort to note different levels of potential DO problems. These characterizations are offered as a point of reference; it is not the goal of this review to develop estuarine/marine DO criteria, which is an ongoing effort within the USEPA. I focus on the occurrence of hypoxia/anoxia (≤ 2 mg/L) as a very serious condition documented in coastal systems, and explore how it may generally relate to N loading.

Hypoxia and a “dead zone” in the northern Gulf of Mexico have received recent attention in the both scientific and public sectors (e.g., Rabalais et al., 1991, 2000; CENR, 2000; NRC, 2000). But a DO problem has been found in many coastal systems worldwide (e.g., GESAMP, 1990; Nixon, 1998) and major one-time or chronic low DO events have been detected since the 1970s. Examples include the New York Bight, Chesapeake Bay, Potomac River, Baltic Sea, Scheldt River estuary, western Long Island Sound, the Venice lagoon, northern Adriatic Sea, several Alabama estuaries, Pamlico River, Providence River, and Hudson River areas (Falkowski et al., 1980; Officer et al., 1984; Oviatt et al., 1984; Larsson et al., 1985; Kullenberg, 1986b; Justic et al., 1987; Turner et al., 1987; Parker and O’Reilly, 1991; Stanley and Nixon, 1992; Nixon et al., 1996; NRC, 2000).

A recent *Science* news article (Malakoff, 1998) suggested the Gulf of Mexico hypoxia was one of more than 50 coastal regions worldwide experiencing severe oxygen decline. A citation for these 50 systems was not given, but worldwide there are a substantial number of coastal systems presently affected, or vulnerable to low oxygen in the near future. In the United States alone, NOAA’s National Estuarine

Eutrophication Assessment survey (Bricker et al. 1999) categorized 42 of 121 estuaries (~35%) with sufficient information as having “moderate or high” depression of DO concentrations. The NOAA survey relied on conditions described by regional experts with extensive first-hand knowledge of each estuary. A USEPA Environmental Monitoring and Assessment Program (EMAP) statistical study (Summers, 2001) has reported results of an unbiased random sample ($n = 1,133$ stations) of 1,516 Atlantic (south of Cape Cod) and Gulf Coast estuaries. The study included a total of 74,744 km² (42 large estuaries [>250 km²], 1,464 small estuaries [2–250 km²], and tidal portions of 10 large tidal rivers). Stations were sampled between 1990 and 1997 in late summer, when DO problems tend to be most pronounced. The spatial distribution of stations with measured hypoxia was centered among northern Gulf of Mexico estuaries and Chesapeake Bay subestuaries, with a sprinkling in the Florida and New York/southern NE regions. The EMAP study estimated that 4% of the represented area had hypoxic conditions and another 16% had DO concentrations between 2 and 5 mg/L. Thus, an estimated ~3000 km² was hypoxic, and a total of ~15,000 km² had DO within a threshold range for biological responses. In comparison, the Gulf of Mexico hypoxic zone may cover up to an additional ~20,000 km² of the Louisiana continental shelf adjacent to the Mississippi and Atchafalaya River deltas (Rabalais et al., 2000).

The inherent vulnerability of systems to low DO events must vary, independent of the N delivery, because factors such as climate, river flow, tides, physical oceanography, individual bathymetry, and geomorphology have influence through constraints on flushing, stratification, and temperature (as a regulator of metabolic processes). Such processes are usually mathematically formalized in sophisticated, coupled hydrodynamic-water quality models or even in simpler DO models (e.g., Officer et al., 1984). Models may not yet fully capture some finer-scale physical processes (Kelly and Doering, 1999), or include all significant biological structure, such as benthic grazers, which potentially affect DO via food web and metabolic influences (e.g., Cloern, 1982; Doering et al., 1986, 1989; Simenstad et al., 2000). In principle though, sophisticated mass-balance or process-type models can link nutrient loading to DO response. Importantly, model formulations explicitly recognize that DO levels will vary with factors other than nutrient delivery or organic matter supply, and they can be useful sensitivity tools for that reason. Models are available and parameterized for a handful of coastal systems, but there are scores of coastal ecosystems for which there exists no calibrated or validated predictive model. Recognizing this lack, it may still be possible to examine time and space trends for a variety of coastal systems to make broader statements about a relationship between N levels and DO depression.

Malakoff (1998) suggested there has been a tripling of reports of dead zones in the last 30 years. At a broad scale, such reports of hypoxia/anoxia in coastal waters map principally within the northern hemisphere (US Atlantic and Gulf coasts, western and northern Europe, areas of the Mediterranean, Japan) around industrialized regions with high human populations and downstream of their associated N exports

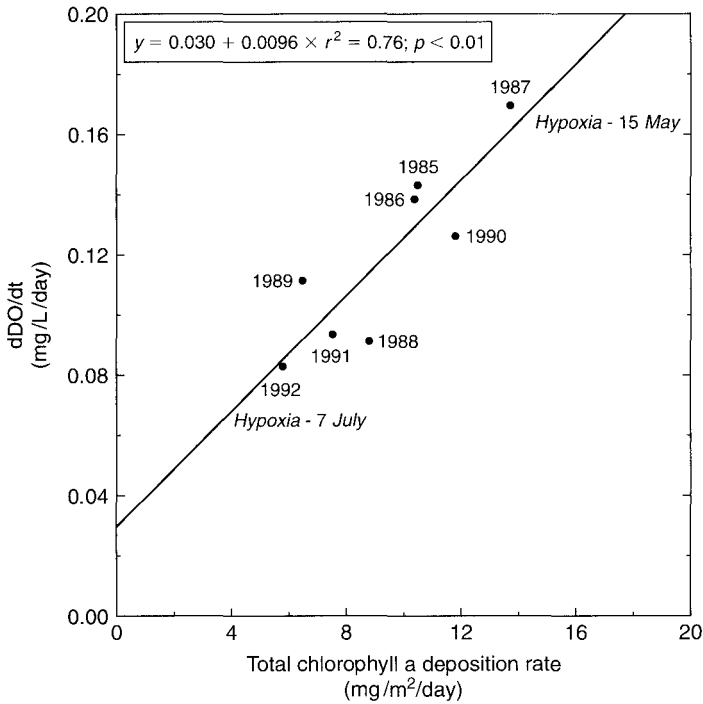
(Malakoff, 1998, Nixon, 1998). However, it is always a quandary to determine if an increased incidence in part arises from looking more, and the connection to a specific cause can be tenuous as we try to interpret, in essence, “epidemiological” data on the basis of an observed symptom. In the United States, for NOAA’s (Bricker et al., 1999) study, of the 44 (of 121) systems expressing what were termed highly “eutrophic” conditions, only about 22 were included for having a “high” or “moderately high” expression of low DO symptoms. Also, some systems display “eutrophic” symptoms unrelated to variations in nutrient loading (e.g., Bricker et al., 1999).

Fortunately, there is more than epidemiological evidence. It is not common to have the intensity of monitoring information to detect signals among the noise of natural variability, but there are cases of an increased scale or intensity of low DO documented within the last half-century. Some cases show DO strongly correlated to nutrient deliveries (Officer et al., 1984; Justic et al., 1987; Parker and O’Reilly, 1991; Boynton and Kemp, 2000).

One study, Rabalais et al. (2000), has described the Mississippi plume dynamics in the northern Gulf of Mexico. By patching together and comparing several time series, it is shown that surplus oxygen concentrations in surface water (indicator of net production from river-originated nutrients) peaks about one month after the Mississippi River flow peaks. A resultant DO minimum in bottom-water follows the surface-water peak by about another month, and is thus associated with decay of recently produced organic matter settling from surface production by diatoms. Most instances of hypoxia are coincident with high water column stability and stratification produced by strong surface-to-bottom density differences. A lighter, freshwater surface plume overlying a denser, saline layer creates such stratification. Importantly, the rate of N loading and the level of diatomaceous remains (as Si) in underlying sediments appear to increase in lock-step through the century, thus indicating how an increase in diatom blooms and resultant hypoxia has arisen in the latter half of the 20th century.

Boynton and Kemp (2000) examined a lengthy time series (1985–1992) at a mesohaline site in Chesapeake Bay (Figure 8a). They were able to correlate seasonal DO decline in subpycnocline deep water to spring bloom deposition of organic matter. Chlorophyll, primary production, and organic deposition were all strongly correlated with river flow, which is a primary determinant of nutrient input to this region. Thus, in part by proxy, higher N input and lower DO are related by a series of expected connections that lead to a secondary consequence from initial plankton stimulation. In spite of strong correlations, Boynton and Kemp (2000) note that other factors may be involved, such as annual variability in temperature and stratification.

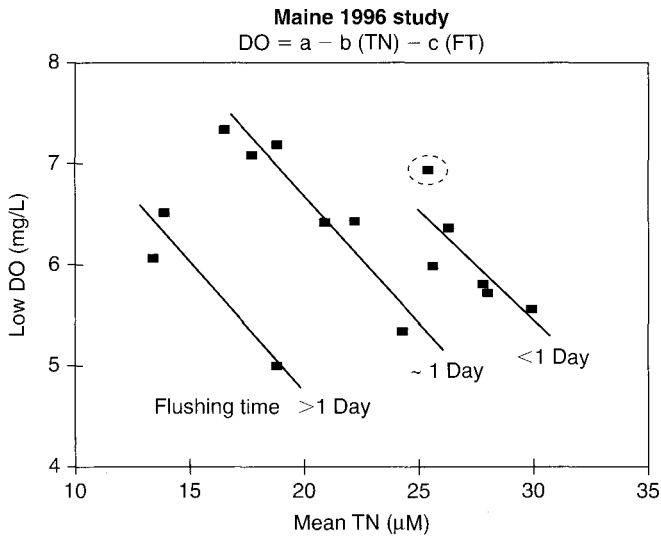
Some trends over space or across systems have been also noted. As the work of Boynton and Kemp (2000) and others (Jay et al., 2000; NRC, 2000) tend to highlight, it requires some level of “scaling” or “classification” to apply cross-system analyses most appropriately in a search for more generalized rules. These efforts have not yet been generally extended to examination of N-induced DO effects (see also Section 6.).



(a)

Figure 8a. Empirical relationship between spring organic matter input and summer seasonal DO decline rates in bottom-water at a site in the mainstem of Chesapeake Bay. Redrawn from Boynton and Kemp (2000). The date at which hypoxia (as DO <1mg/L) was first encountered in years with highest and lowest organic matter is indicated.

In contrast, Figure 8b shows a group of estuaries and embayments in Maine, where DO minima were measured along with TN concentrations and a number of other features of each system (morphology, watershed, salinity, temperature, freshwater inputs, tidal range and flushing, stratification). Hypoxia is not generally an issue in this region and the variability among sites within and across systems is small. A multivariate step-wise regression analysis nonetheless suggested a strong relationship between TN and observed DO minima, if a system's flushing time (calculated from tides rather than freshwater flow in these ocean-dominated systems) was considered as a classification factor. Interestingly, the resultant multiple regression comes close to predicting the DO concentration that occurs with the flushing time and TN concentrations of nearby Boston Harbor (Kelly, 1997a, 1998). The



(b)

Figure 8b. Relationship observed among N concentrations, minimum DO concentrations, and flushing time for 15 small, short-residence time, tidally flushed estuaries and embayments. Adapted from Kelly, 1997c, in which stepwise multiple regression analyses selected TN and flushing as first-order explanatory variables accounting for >60% of the variability. One point (circled) did not fit the trend. Flushing time is based on replacement of estuarine volume by tidal volume input every 12.42h, and as such assumes complete mixing. Freshwater replacement time is much slower; these systems generally have low Fw/tidal volume ratios (see Figure 3a).

Maine systems are all cold-water and relatively pristine, with very short residence times because of flushing dominated by tidal actions. General applicability of these results is thus untested, although the indication of physical control of DO response to N is intriguing.

There is compelling *experimental* evidence on the relationship between N loading and low DO. A MERL nutrient gradient experiment produced oxygen problems at its upper N loading levels, >9,000 mmol TN/m²/year (Oviatt et al., 1986). Low DO was also concomitant with primary production that reached at least 400 g C/m²/year but the two most enriched MERL treatments, which had more severe, chronically-low DO, averaged production rates above 750 g C/m²/year for the 2 years it was measured (Oviatt et al., 1986 and see Table 1). In contrast to many natural systems where DO problems are known, the MERL studies were conducted using

Table 1. DO, nutrient loading, and other characteristics for selected coastal areas and Marine Ecosystem Research Laboratory (MERL) mesocosm enrichment experiment.

System	Area	Depth Average (m)	Annual TN Loading (mmol/m ²)	Residence Time (month)	¹ DO Status	Vertical Mixing Status	² Normalized TN Loading (µM)	Primary Production (g C/m ² /year)
³ <i>Experimental</i>								
MERL-control	2.63 (m ²)	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	1,150 (901)
⁴ <i>Field</i>								
⁵ Baltic Sea	374,600 (km ²)	55	217	250	H/A	stratified	81	~149-170
Scheldt	277	11.2	13,400	3	H/A	??	295	?
^{6,7} Chesapeake Bay	11,542	6	938	7.6	A	stratified	98	~380-520 (361-858)
⁶ Potomac River	1,210	5.9	2,095	5	H/A	stratified	146	~290-325
⁸ Guadalupe estuary	551	1.4	548	10	?	??	322	?
Ochlocknee Bay	551	1.4	2,058	1	?	??	121	?
Delaware Bay	24	1	5,995	0.1	OK	49	?	~200-400
⁹ Narragansett Bay	1,989	9.7	1,900	4	OK	stratified	64	270-290
	328	8.3	1,960	0.9	OK	weakly stratified	17	

(Continued)

Table 1. (Continued)

System	Area	Depth Average (m)	Annual TN Loading (mmol/m ²)	Residence Time (month)	¹ DO Status	Vertical Mixing Status	² Normalized TN Loading (μM)	Primary Production (g C/m ² /year)
¹⁰ Providence River	24.13	3.7	13,600	0.083	H	stratified	25	?
^{10,11} Providence River	24.13	3.7	13,600	0.233	H	stratified	70	?
¹² Boston Harbor	103	5.5	21,600	0.266	~H	weakly stratified	86	?
¹³ N. Boston Harbor (Outer)	13	10	107,692	0.03	OK	mixed	27	263–546
¹⁴ N. Gulf of Mexico	20,000	30	6,500	¹⁵ 6	H/A	stratified	107	~290–320

Notes:

¹H = hypoxia, A = anoxia.

²Volumetric 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).

³See Nixon et al., 1984; Oviatt et al., 1986; Nixon, 1992; Nixon et al., 1996. DIN was used to enrich treatment conditions (e.g. IX...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, 1988a, which did not include measurements of primary production above 600 g C/m²/year (Nixon, 1992). These values are used in Figures 6 and 7. Parenthetical production values for year 2 are from Keller, 1988b. Hypoxic and anoxic events were periodic, not chronic.

⁴Except 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.

- ⁵Also 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.
- ⁶Also see Boynton et al., 1995; Boynton and Kemp, 2000; historical Chesapeake production range (parenthetical) is from Boynton et al., 1982.
- ⁷Mainstem stratification, increasing anoxic extent; Officer et al., 1984; Boynton and Kemp, 2000.
- ⁸Top line is for dry flow, bottom line is for wet flow.
- ⁹Only strongly stratified by freshwater at head of Bay in Providence River area, see notes 10, 11. Production range is from Nixon, 1997 (does not include historical pre-settlement estimate of 120–130 g C/m/year).
- ¹⁰Oviatt et al., 1984; Doering et al., 1990; Asselin and Spaulding, 1993; TN loading from seaward and landward inputs, average residence time (2.5 days), low DO in 13–15 m channel.
- ¹¹Uses longer 7-day residence time during very low flow conditions, Asselin and Spaulding, 1993.
- ¹²TN 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.
- ¹³Northern harbor section, Kelly, 1998. Harbor station production of Kelly and Doering, 1997.
- ¹⁴Area 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 130×10^9 moles/year (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 t/year.
- ¹⁵Assumed a 6-month residence time (\sim seasonal turnover) for illustration only; if longer, then normalized concentration would increase accordingly.

(intermittently) well-mixed mesocosms with a 5-m deep water column and underlying active and functional benthic community (e.g., Kelly et al., 1985; Nixon et al., 1986).

I have brought together several bits of data to explore patterns across experiments and field studies (Table 1). Included are natural or experimental systems having the most complete N loading (TN) and budgets available in the literature, each with indirect or direct estimates of ocean N loading. Not all of these systems have DO problems, but many do. Summarized systems have great diversity – in latitude (Baltic to Gulf of Mexico), size (for natural systems, 10–100,000s of km²), depth (1–55 m), estuarine residence time (days to years), and vertical stratification (well-mixed to strongly stratified). Areal TN loading rates have a wide range (217–107,692 mmol N/m²/year). Table comparisons indicate DO problems in some low- to medium-loaded systems (Baltic, Chesapeake), but not necessarily in all those with higher loading (e.g., Delaware Bay or Narragansett Bay), which illustrates some of the difficulty of defining directly an N loading–DO relationship.

In spite of all their differences, systems with DO problems may share a similar residence-time corrected loading, or “expected” concentration. The “expected” concentration of Table 1 is a simple correction of areal N loading for residence-time and depth, a parameter that correlates well with observed mean *in situ* N concentrations in some coastal systems (Kelly, 1997a, b). This is a similar scaling concept analogous to that used in Figure 7 and explored by others (e.g., Valiela and Costa, 1988; Kelly, 1998; Boynton and Kemp, 2000). A rough hypoxic threshold value, scanning the data of Table 1, might be an “expected” TN concentration on the order of 80 μM. There are several important issues of scale. First, Providence River has a DO problem compared with its parent Narragansett Bay system and also a higher “expected” (as well as measured) concentration (Table 1). In contrast, the outer Boston Harbor region itself does not have a DO problem and its value is lower than its whole parent system (Table 1). Both these sub-area observations support a threshold concept. Second, freshwater residence time in the Providence River strongly affects the expected value (Table 1) and it may be significant to occasional development of hypoxia/anoxia. This is a phenomenon similar to that described recently by Howarth and co-workers (in NRC, 2000) for low flow conditions in the Hudson River estuary; in that case not only was residence time affected by low flow but other elements of the hypoxic recipe, stratification and primary production, both increased. Third, note that the “illustration” value calculated for the northern Gulf of Mexico uses various assumptions that should be challenged. Input occurs to an area larger than the immediate hypoxic zone, so loading must also be lower; I do not know of an estimate for residence time in this open shelf situation and just assumed a seasonal turnover. Lastly, a suggested value near 80 for stratified natural systems is lower than indicated for the well-mixed conditions of the MERL experiment, where low DO was produced at values >130.

If the very speculative concept were valid, it would operate mechanistically through an influence of residence time on production (as per Figure 7). It is clear

that hypoxia occurs at lower primary production levels in stratified natural systems than it took in well-mixed MERL conditions. Production ranges for each system are large. If hypoxia occurs towards the higher end of most systems' range (Table 1), we could tentatively place most hypoxia-associated conditions with production $\geq 300 \text{ g C/m}^2/\text{year}$. The Baltic Sea would be a distinct exception; its very long residence time possibly allows greater long-term accrual of organic material, so legacies of past production help promote low DO. This is unproven, as is a distinct threshold of production to produce hypoxia. The simple point is that DO problems obviously occur at different levels of (area-based) N loading, so it seems logical to explore flushing and residence time as scaling factors that moderate effects such as lowered DO.

Besides production, the strength and spatial details of stratification, among others, are factors influencing DO (e.g., Turner et al., 1987, Kelly and Doering, 1999). A pattern like Figure 8b might arise in part through flushing effects on stratification. Also, temperature, turbidity or periods of cloudiness, or even shallowness itself may also be key factors. The growth of macroalgae and associated hypoxia in shallow water may occur at levels of N lower than the loading necessary to produce hypoxia in deeper areas (see Section 4.4). In contrast, grazing by benthic filter feeders may moderate enrichment effects of chlorophyll or productivity (e.g., Cloern, 1982). Ultimately, we have to recognize that N loading and productivity create only a *potential* for lowered DO; to develop quantitative relationships we need continued work to classify systems by attributes which make a DO problem more likely.

4.4. Benthic Primary Producer Response (SAV, Macroalgae) to Nitrogen in Shallow Systems

There are a number of excellent site summaries and reviews of submerged (often called submersed) aquatic vegetation (SAV) and macroalgae in coastal systems – freshwater, estuarine, and marine. SAV is a broad term that includes seagrasses (marine angiosperms) as well as freshwater macrophytes which are found in fresher regions of estuaries (e.g., Dennison et al., 1993). Studies describe many facets of SAV: the ecological importance of rooted macrophytes and seagrasses in coastal water; temporal patterns of seagrass decline, including possible relationships to nutrient loading, water quality, or other historical factors; potential for recovery from anthropogenic nutrient/sediment loads. Still other studies describe the stimulation of nuisance blooms of macroalgae by nutrients in shallow coastal systems, including coral reefs. The reader is referred to a number of examples (Thayer et al., 1975; Zieman, 1982; Stevenson, 1988; Sand-Jensen and Borum, 1991; Dennison et al., 1993; Stevenson et al., 1993; Duarte, 1995; Lapointe, 1997; Valiela et al., 1997b; Fourqurean and Robblee, 1999; NRC, 2000).

SAV is ecologically significant. It is important to waterfowl, it affects water quality by buffering turbidity in estuaries, contributes very high primary productivity and feeds a significant food chain through (mostly) detrital pathways, and offers

habitat or nursery for larvae, juveniles, or adult fish and shellfish. As an example, Heck et al. (1995) suggests that eelgrass habitat can support macroinvertebrate production (prey items for fish) that is disproportionately large compared with unvegetated areas (intertidal and subtidal muds). SAV can dominate overall secondary productivity of shallow estuaries even when its areal coverage is as low as 10%, and its contribution to the consumer food web can be more significant than implied by its level of contribution to primary production. Simply put, concern for SAV decline or loss focuses on the loss of all the stated functions above, especially for the food web (fish and shellfish) supported by its presence.

The evidence for SAV response to nutrients goes beyond epidemiological and anecdotal site trends, and there are a number of detailed examples of global SAV decline in the last half-century, in Europe, North America, and Australia (e.g., Orth and Moore, 1983; Costa, 1988; Valiela et al., 1992; Dennison et al., 1993; Fourqurean and Robblee, 1999). Throughout history there have been other causes of seagrass declines, but many during the last half of the 20th century have been linked to nutrients, specifically N. A variety of controlled experiments, including in mesocosms, have confirmed a link to N and the qualitative sequence of events with increasing loading (cf. Kemp et al., 1983; Twilley et al., 1985; Short et al., 1995; Taylor et al., 1995a, b). Based on various site trends, comparative analyses, and experimental evidence, Duarte (1995) determined that there was "an adequate empirical basis to formulate qualitative predictions on the direction of change in submerged vegetation upon nutrient enrichment," but there was a lesser basis to predict recovery with lessening of nutrient loading.

It has been noted that changes in SAV are not gradual, but have thresholds and appear as step changes with a sudden shift in vegetation, implying both direct and indirect effects are at play. A principal mechanism for nutrient effects on SAV is uniformly recognized as a secondary consequence of enrichment of other primary producers. Hansson (1988) confirmed that under very low nutrient conditions in lakes, benthic algae can access nutrients from sediments and have a competitive advantage over planktonic algae, whose advantage grows with nutrients in the water column, due to their superior access to light. Similar concepts apply where principal benthic producers are rooted macrophytes or seagrasses (Figure 9a). Direct nutrient stimulation of plankton; periphyton on sediments; epiphytes on the vegetation, or other algal, emergent; or floating overgrowth all can induce light limitation of the seagrass or macrophyte, rooted to the bottom and thus subject to shading by unattached forms. Studies also suggest that algal stimulation can affect root metabolism and indirectly affect SAV, and there is some variability in the paradigm that may be induced by the effects of grazers on different producer forms. But the simple progression of Figure 9a, long described as a freshwater eutrophication paradigm (Wetzel, 1983), appears applicable to estuarine areas (Stevenson, 1988; Sand-Jensen and Borum, 1991) and has been a principal conceptual foundation of studies examining SAV decline.

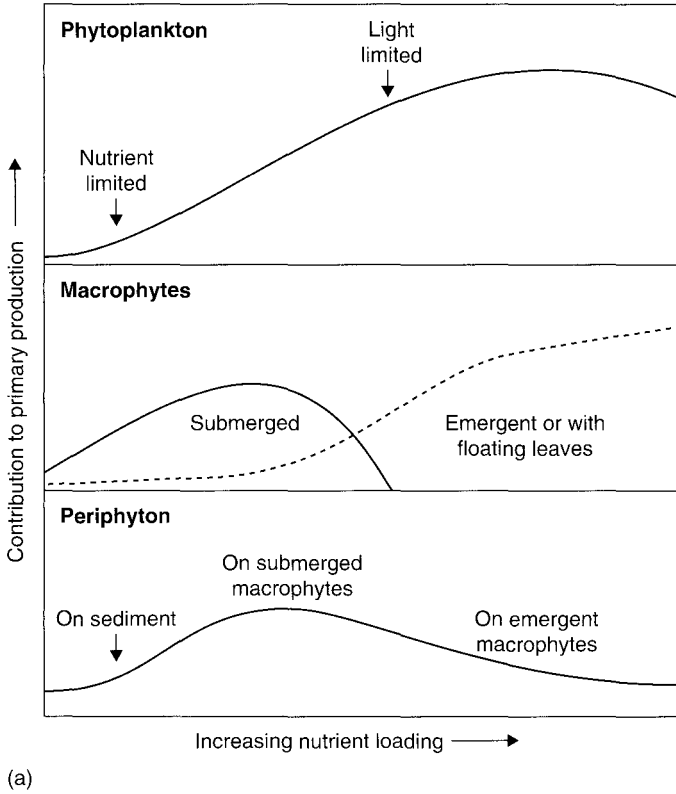
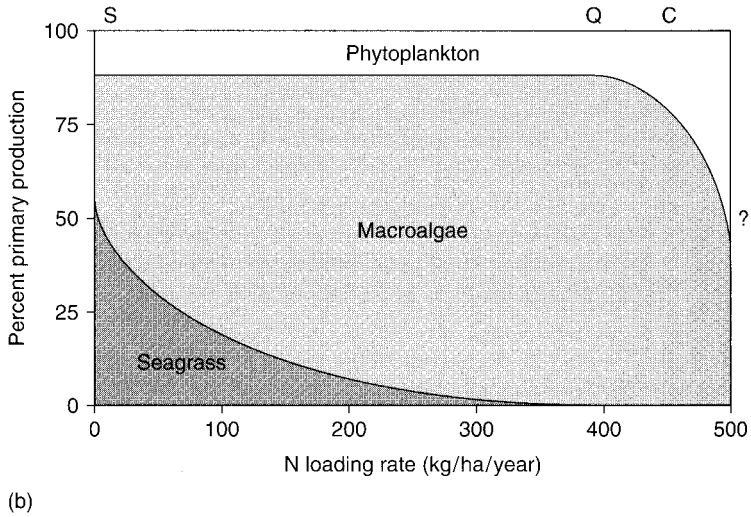


Figure 9a. General pattern of changes among primary producers with increased nutrient loading in shallow aquatic ecosystems. Redrawn from Sand-Jensen and Borum (1991). Conceptual progression based on summary of data for temperate lakes.

Figure 9a is fundamentally similar to trends established for a shallow southern Massachusetts estuary (Figure 9b), although the “emergent/floating” macrophyte forms are replaced by nuisance macroalgae (e.g., *Cladophora*, *Gracilaria* sp.) with high nutrient uptake rates. In Waquoit Bay, eelgrass was shown to decline from 1951 to 1987, from an extensive spatial coverage to restriction to a small patch near the mouth of the estuary. Duarte (1995) compared producer forms (seagrass, macroalgae, phytoplankton) in terms of various physiological properties in relation to the environment (light, nutrients) to suggest that macroalgal forms have physiological advantages over seagrasses in N-loaded systems, being more nutrient- and less light-limited. The Valiela et al. (1997b) trends in Waquoit Bay fundamentally followed the qualitative predictions of Duarte (1995). Sub-estuary data (Figure 9b)



(b) Figure 9b. Trends in primary producers with increased nutrient loading: an example from three subestuaries of Waquoit Bay, MA. Redrawn from Valiela et al. (1997b). The position of three short-residence time sub-estuaries (S, Q, and C) placed along a loading axis. Note: 1 kg/ha is about $\sim 7 \text{ mmol N/m}^2$. Eelgrass (*Zostera marina*) decline is rapidly promoted; $>50\%$ relative reduction is $>700 \text{ mmol N/m}^2$. A speculated threshold for full phytoplankton domination is $\sim 3500 \text{ mmol N/m}^2$.

suggest that at only modest N-loading enrichments, macroalgae replaced eelgrass, and increased watershed N inputs could be isotopically linked to all producers. Previously, Valiela et al. (1992) showed that enhanced macroalgal development facilitates development of anoxia, a mechanism to promote problems inherent to shallower systems that may not be captured in previous DO discussions (Table 1). Anoxia is among several ecological process/food web changes accompanying the shift to macroalgal dominance that are consequential to commercial shellfish populations (Valiela et al., 1992).

Consistent with a theoretical succession which shifts, essentially from nutrient to light limitation, various studies have noted that seagrasses colonize to a depth with a certain light level. Requirements are species-specific, but the light minima averages near 11% of surface light (Duarte, 1995). Negative relationships between nutrient concentrations and the depth limit of benthic macrophytes have been noted. Decline in seagrass beds sometimes has been observed from depth shoreward, as phytoplankton and epiphytes reduce available light. Related to this effect, the bathymetry of vegetated area can affect the pace and spatial distribution of seagrass decline and resultant patchiness in different systems. This phenomenon contributes to a lack of a general quantitative relationship between SAV declines and N loading (Duarte, 1995).

There is, however, some information and data such as given in Figure 9b can be used to place some bounds on thresholds for shifts between eelgrass and macroalgae, in response to N. A value of $<700 \text{ mmol/m}^2/\text{year}$ (converted from figure units of kg/ha/year to be consistent with other expressions in this chapter) is suggested for eelgrass decline when macroalgae become dominant. Given a very short water residence time ($\sim 1\text{--}4$ days) and shallow water depth (0.9 m) (Jay et al., 1997; Valiela et al., 1997b;) this value would equate to a residence-time and depth-normalized loading of about $4 \mu\text{M}$ (cf. data of Table 1). In a rough sense this expectation seems consistent with conditions in the main body of Waquoit Bay where there was a vestige of seagrass bed still remaining in 1987. This sub-area has higher salinity, low chlorophyll ($\sim 3\text{--}5 \mu\text{g/L}$), little NO_3 , and NH_4 concentrations averaging $\sim 2 \mu\text{M}$ in summer (Valiela et al., 1992). Nitrogen concentrations increase upstream into fresher water and this is where extensive *Cladophora* mats are found, where water concentrations average perhaps $20 \mu\text{M}$ (Valiela et al., 1992), or roughly consistent with expectations for a $3500 \text{ mmol/m}^2/\text{year}$ threshold implied as the edge of phytoplankton dominance (Figure 9b).

Another extended example for assessing thresholds comes from Chesapeake Bay and its subareas. Brush and Hilgartner (2000) present a record of SAV in the upper Bay since the 1600s, from paleo-evidence of SAV seeds in sediments. SAV distributions have high variability in space and over time. Nonetheless a distinct threshold is suggestible in response to land use change (with sediment/nutrient loading increases) that began in the 1700–1800s and intensified in the last half of the 20th century. The number of tributaries with SAV has decreased markedly in the 1900s. In 1983, Orth and Moore (1983) detailed a major loss of SAV. Studies at that time and soon after, including controlled microcosm and field experiments, demonstrated a connection between SAV, turbidity, and N (Kemp et al., 1983).

Subsequent studies have used an understanding of light requirements to define conditions where water quality is sufficient to support SAV (Dennison et al., 1993). By surveying nutrients, chlorophyll, turbidity, and light extinction at different depths and areas, studies established where SAV was present, or where transplants were able to survive. Using an experimental field study, Stevenson et al. (1993) transplanted plugs of living plants (*Ruppia maritima*, *Potamogeton perfoliatus* and *Potamogeton pectinatus*) to different areas in the Choptank River, and assessed survival. The following water quality thresholds for survival were indicated: $\sim 15\text{--}20 \text{ mg/L}$ total suspended solids, $15 \mu\text{g/L}$ chlorophyll, $\text{DIN} < 10 \mu\text{M}$, and $\text{PO}_4 < 0.35 \mu\text{M}$. Stevenson et al. (1993) emphasize that survival may occur at lower levels than that which would instigate declines. Concurrent work of Dennison et al. (1993) in higher salinity areas of the Bay (York River) indicated a similar range for patterns of eelgrass, *Zostera marina*.

Boynton (2000) provides a final Chesapeake Bay example, for the Patuxent River. He shows a precipitous seagrass decline concomitant with increased chlorophyll and decreased light penetration between ~ 1960 and 1980. During this time, chlorophyll rose from < 10 to almost $30 \mu\text{g/L}$. Total N loading increased from ~ 0.91 to

1.73×10^6 kg N/year from 1963 to 1985–1986 (Boynton et al., 1995). Using dimensions from Boynton et al. (1995) and the estuarine residence time recently estimated by Hagy et al. (2000), one can calculate the TN loading range as ~ 515 – 980 mmol/m²/year. With a mean depth of about 4.8 m and median estuarine residence time of 25 days, calculations suggest an increase in residence-time corrected, volumetric-expressed loading from ~ 7 to 14 μM over the period (see above, and Table 1). These values coincide with DIN concentrations near 10 μM in 1969 (cf. Boynton et al., 1982; Nixon and Pilson, 1983), increasing to an average DIN value of ~ 15 μM in 1985–86 (Boynton et al., 1995). Interestingly, these values all surround the survival conditions suggested by Dennison et al. (1993) and Stevenson et al. (1993).

There are other examples of trends, both response and the recovery, for both seagrass decline and macroalgal problems. For example, by lowering chlorophyll levels to a target of 8 $\mu\text{g/L}$, seagrass recovery appears to be proceeding, after a lag, in Tampa Bay (NRC, 2000). More examples should be used to develop appropriate comparisons of threshold levels of loading for different systems, SAV species, and geographic regions, but the few examples here suffice to put the problem in some quantitative perspective in relation to other effects (Section 5).

Before leaving SAV effects, it is worth considering in concept possible differences across systems. Valiela et al. (1997b) discuss two factors. First, they hypothesize that presence of fringing salt marsh may intercept groundwater and surface flows and lead to denitrification along the flow path; thus variations in the area of tidal salt marshes in an estuary could affect its vulnerability by affecting the eventual loading to the estuarine receiving waters. Second, Waquoit Bay has a short water residence time (~ 1 – 2.5 days); phytoplankton has less ability to respond to nutrients at these very short residence times. This may exacerbate the ability of macroalgae to replace seagrasses in this estuary, compared with those having longer residence times, where phytoplankton may more easily dominate and shade both seagrass and macroalgae, at relatively lower input rates.

4.5. Phytoplankton Species Response to Nitrogen, Stimulation of “Harmful Algal Blooms”

There are a variety of nuisance algal blooms (such as blue-green algae) which cause aesthetic and other problems, generally in only the oligohaline portions of estuaries (salinity of 0–5 PSU) (Paerl, 1988). Of more concern to this review are saline forms, which characteristically include dinoflagellates. There are nearly two dozen noted genera of phytoplankton that produce potent toxins, including ones historically called “red tide” dinoflagellates (Anderson and Garrison, 1997). There are species-specific toxins, which include those named for their symptomology in human consumers: paralytic, neurotoxic, amnesic, and diarrhetic shellfish poisoning (respectively, PSP, NSP, ASP, and DSP). There are endotoxins that accumulate through the food chain (and thus to commercially sought fish and shellfish species) and there are exotoxins that are exuded in the water. But not all “red tides” or dinoflagellates are harmful, not all toxic species are dinoflagellates (e.g., cyanobacteria

of the genus *Trichodesmium*, diatoms of the genus *Pseudo-nitzschia*, prymnesiophytes of the genus *Phaeocystis* – various references in Anderson and Garrison, 1997, such as Turner and Tester, 1997), and not all problem species discolor the water at all (or are “brown tides”). Blooms also can disrupt normal filter feeding or grazing, change the food chain, foul beaches, or cause acute DO problems through rapid accumulation and decay. Thus, the term “harmful algal bloom(s),” or HAB(s), was coined to include species-level growth that is toxic, hypoxia-inducing, or food-web disrupting. Many HABs are elusive in the sense that they exist in some type of resting stage (such as a cyst), which can lay dormant in sediments, until it “excysts” and provides a seed population in favorable conditions. The triggers for this action are not well understood, but cysts are a mechanism for remaining in a location for a long time once advected or carried there (in ballast water?) and established. The various known mortality modes and impact mechanisms of HABs are summarized in Anderson and Garrison, 1997 (cf. Smayda, 1997). In broadest use, HABs includes both microalgae and macroalgae; the latter has been included in Section 4.4. Many microplankton HABs problems occur in slightly deeper coastal waters, so there is often a physical separation in potential SAV/macroalgal and HAB effects, whereas DO effects can occur in both shallow and deep systems.

ECOHAB (1995) and Anderson and Garrison (1997) offer excellent summaries of the problem. Concerns for HABs have heightened principally because the types of observed problems (numbers of newly identified problems and problem species), the spatial extent or new locations of cases, and the incidence of reported occurrence all have expanded in the past few decades. This seems especially true in Western Europe and North America, where N loading increases are particularly notable. As an example, Paerl and Whitall (1999) examine the case for open coastal systems of the North Atlantic Ocean (Europe and North America), where new atmospheric inputs, in particular, have increased and form a substantial portion of the external N input. Concurrence of HAB events with high atmospheric N loading is part of the epidemiological evidence that has been compiled to suggest a linkage with increasing N inputs. Earlier, Smayda (1990) suggested a global epidemic of “novel” (~harmful) blooms and summarized evidence for increased spatial occurrence around the world. There is provocative epidemiological evidence, but strong direct linkages to N loading have not been confirmed and, certainly, no cross-system comparisons can be developed to suggest that a certain critical N load is involved.

Some sites with long-term data sets have reported an increased HABs occurrence frequency, coincident with a temporal increase in nutrient loading. One is Tolo Harbour, Hong Kong (Smayda, 1990). NRC (2000) cites another example from the inland Sea of Japan. Burkholder and Glasgow (1997) make an argument for a recently identified “phantom” dinoflagellate (with encysting form), *Pfiesteria*. They suggest that nutrients may foster outbreaks of these organisms, which can kill fish and also cause human health effects. Of course, there are areas of the world with increasing nutrient loading which do not have an increased occurrence of HAB

species. The challenge to epidemiological and time series evidence is that increasing reports could be due to increased attention and detection ability.

Smayda (1990) speculated that changes in observed N/Si/P ratios in some coastal areas (with increased N loading) over the last few decades may be promoting growth of forms that have low (or no) Si requirements (e.g., dinoflagellates, *Phaeocystis*) over more favorable bloom diatom species (expected from tenets of Officer and Ryther, 1980; Ryther and Officer, 1981). Rabalais et al. (2000) show N/Si ratio changes in the Mississippi and changes in the mix of diatom species. They note that some harmful forms (e.g., *Pseudo-nitzschia* spp. and maybe others) are more recently observed, but a wholesale shift to HAB forms has not been observed. Interestingly, various MERL mesocosm enrichment studies have never noted a shift to HABs or extensive HAB species development even though plankton biomass and productivity climb (Oviatt et al., 1986; Doering et al., 1989). Moreover, there is a contrasting case. Keller and Rice (1989) noted that a brown tide organism (*Aureococcus anophagefferens*) was present at a MERL experiment's start (from Narragansett Bay feedwater), in which nutrient levels and N/Si ratios were subsequently altered. After a brief response to initial enrichment, populations declined, appearing to be out-competed by diatoms; the organism did best in initial low nutrient conditions. Perhaps the simple message is that species-level response predictions are exceedingly difficult in complex ecosystems.

In all, we do know that relative increases of N might selectively favor some phytoplankton forms. It is possible that HABs could increase as part of a general increase in the phytoplankton community biomass and production associated with higher N loads, as is now somewhat described (Sections 4.1 and 4.2). It is far more controversial, as there seems meager evidence, to suggest that HAB species are being selectively stimulated. The contrasting physical (and chemical, ecological) conditions favorable to diatoms, dino- and micro-flagellates as *groups* have been outlined for marine systems (e.g., Pingree et al., 1975; Margalef, 1978; Demers et al., 1986; Legendre and Le Fevre, 1989). However, we cannot easily predict when any particular phytoplankton species among the community will actually flourish. In sum, a major concern exists, and HABs have been expanding according to available records, but a quantitative linkage to N for individual harmful species has not yet been confirmed.

5. A SUMMARY (CIRCA 2001) AND SPECULATION ON PROGRESSIONS WITH INCREASING ENRICHMENT

The evidence demonstrating a variety of effects of N on coastal systems is strong, and found at many levels of investigation. By most accounts, the scales of the problems have been growing rapidly throughout the 20th century. We have developed some general rules relating chlorophyll and production responses to N loading. By and large, the chlorophyll and production trends have strong similarity to those established for lakes. Chlorophyll and production increases are precursors to adverse secondary effects of concern (Figure 1), but even for these primary effects

we do not yet have site-specific predictability. I have three brief summary topics related to this.

One recurrent theme in this review has been the significance of physics, specifically water residence time. Recognition of the importance of residence time has long been woven into coastal studies, but only as specks of color here and there; it needs to be a dominant hue in the fabric of eutrophication research. We know, for example, that water residence time can affect how coastal systems remove N loading via denitrification losses (e.g., Nixon et al., 1996) and how it can influence the expression of benthic grazers on overlying plankton (e.g., Simenstad et al., 2000). We recognize its influence on water quality/biological dynamics and role in determining vulnerability to enrichment effects. Residence time has been a cornerstone of the concept of lake eutrophication, where general predictive relationships with residence-time normalized loading have been developed for chlorophyll, secchi depth, primary production, hypolimnetic oxygen depletion, and fish yield (e.g., Jones and Lee, 1986). This summary suggests that residence time plays a very similar role in coastal estuarine/marine production.

The second topic focuses on our understanding, both qualitative and quantitative, of the primary and secondary effects of N enrichment (e.g., Figure 1). In at least a handful of systems, enrichment progressions have been noted and linked to increasing nutrients, such as depicted in Figure 9a, b for shallow systems. Observation or historical reconstruction of change shows subtle-to-dramatic algal increases, sharp food-web shifts from benthic producers to planktonic producers and associated higher trophic-level organisms, and mortality from anoxia. From these, management strategies for specific systems have been formulated; there are examples where target reduction goals have been set which have helped with the problem (e.g., NRC, 2000).

There are, however, different levels of confidence in our general ability to link the response to N loading for the different categories of effects reviewed. Considering confidence, quantification, and generality of findings, I believe it reasonable to rank our overall understanding of effects (in decreasing order) as:

Chlorophyll > Primary production > DO > SAV ~ Macroalgae >> HABs
(phytoplankton)

Some will argue the exact order, the middle being the contentious ranking. The ranking is not to imply we have site-specific predictive capability for any effects. For even the best of the derived quantitative relationships, one always seems to be able to find new, outlier systems, as examples in this chapter illustrate.

The ranking, of course, suggests that the closer the effect is to the stimulus the greater our ability to couple the two. With "secondary" effects (such as DO, SAV, macroalgae, HABs; see Figure 1), and specific population responses, each highly dependent on many confounding factors, the requirements for details about the character, history, and structure of the system grow. Intensive studies and uniquely tailored simulation models should convince us that resolving such effects with

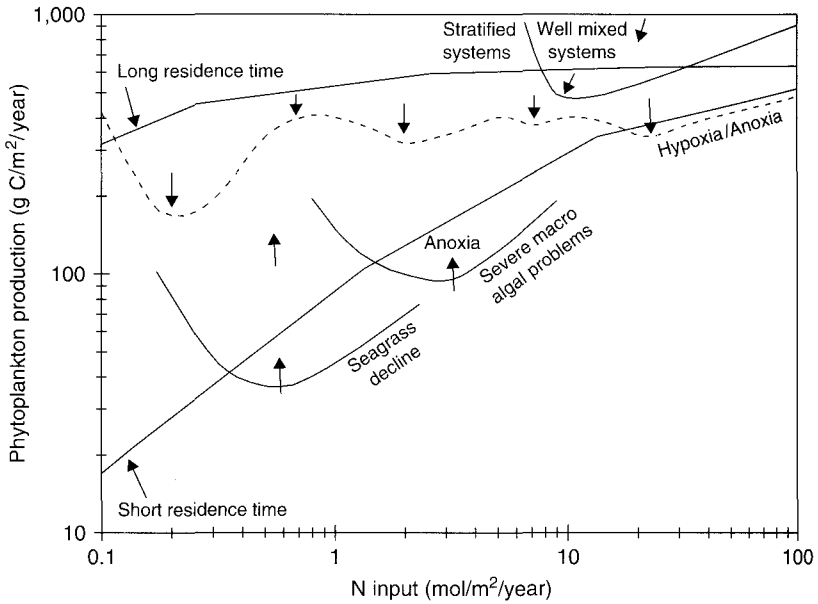


Figure 10. A speculative concept of the progression and thresholds for effects of N loading. Wavy lines and partial curves suggest possible threshold ranges for the combination of N loading and production based on where a noted effect has been reported. The sequence of arrows along a dotted line suggesting hypoxia/anoxia, represent (from left to right): Baltic Sea, Chesapeake Bay, Potomac River, Mississippi plume, and Providence River/Boston Harbor (Table 1). The partial curves for seagrass and macroalgae are based on description for Waquoit Bay and the Patuxent River (see text). The figure concept is borrowed from Figure 7, but ranges for axes have been narrowed to reflect the ranges reported only for marine coastal systems.

some level of predictive confidence is possible, but takes considerable effort. The demands and precision of research must be balanced against a coarser level of guidance that can be effective for management action, where setting targets within a range can help be protective.

With this thought, I have compiled information to illustrate development of quantitative thresholds (Figure 10). Increasing nutrients cause various algal changes affecting SAV, followed or accompanied by macroalgal changes (in shallow systems), with hypoxia/anoxia therefore one of the later effects. I have crudely attempted to map some boundaries for effects over the pattern and range of N loading, water residence time, and productivity observed for the bulk of coastal systems. These "thresholds" should, at best, be a speculative hypothesis to be tested and improved. Productivity data for systems with a DO problem are from Table 1. For SAV and macroalgae, the

domains are suggested without productivity data, that is, only from measured N input and residence times. It is clear that a given effect, such as hypoxia, occurs across a range of areal loading rates, probably being manifest more by productivity and affected by residence time. The array of north temperate systems with DO problems (Table 1) includes many with productivity $>300\text{--}400\text{ g C/m}^2/\text{year}$. But it is also clear that only the potential for an effect is suggested and not all at these levels have hypoxia (thus a wavy dotted line, Figure 10). High loading and very high productivity were necessary to induce a chronic DO problem in the well-mixed MERL mesocosms.

Some have wondered whether the suggested growing global incidence of hypoxia might be our figurative “canary in the coal mine.” This may be true only in the sense of presently providing warning of the increasing scale of coastal problems. At least for shallow systems, evidence suggests that before situations actually advance to a low DO problem arising from loading and plankton productivity, effects such as SAV loss and problem macroalgae blooms, will indeed appear. Based on the chlorophyll data for the systems with SAV loss or macroalgal bloom thresholds, one would expect associated plankton productivity to be very low, as is suggested (Figure 10). We do not know this to actually be the case in all systems, and the figure only begins to illustrate how variations in physics may modify a sequence of effects at a given loading rate. The illustration indicates a great deal of complexity still to be resolved, probably through further classification of systems and their responses to enrichment. Importantly, though, the figure reinforces the notion that it would help to have greater study of systems in different physical settings, and especially, more at the lower end of the coastal loading range. Many coastal areas being observed at their high, present-day levels of loading probably have passed already through a succession of changes.

With progressive enrichment comes consequential species change, SAV being our best example. We have the least information on the general topic of species compositional change, and have little to guide us as to whether there is any threshold stimulation point for a specific biological change, such as HABs. This raises the third related topic. Food webs and fisheries are a fundamental societal concern, but they are ecologically removed from the direct effects of nutrient loading. The world is not lacking for evidence of fish kills, but it is fascinating that, with hypoxia and benthic mortality documented at a huge scale in the northern Gulf of Mexico, analyses have difficulty showing the effect on total fish catch even though decline in important species (e.g., brown shrimp) has been noted (CENR, 2000). Reasons for this, include the difficulty of obtaining data on fisheries that reflect the actual conditions of the stock. There may also be time lags for expression of effects in longer-lived species. Unlike infaunal benthos, fish and epifaunal organisms (adult shrimp) can move to avoid hypoxia, but with such a large benthic food base affected, the concerns are large for the long-term sustainability of the fishery and fundamental shifts in the nature of the fish consumers in the food web (Caddy, 1993). In Caddy’s view (Figure 11), there are consumer food-web changes across the loading regime (often to less desirable, commercially sought species), many of

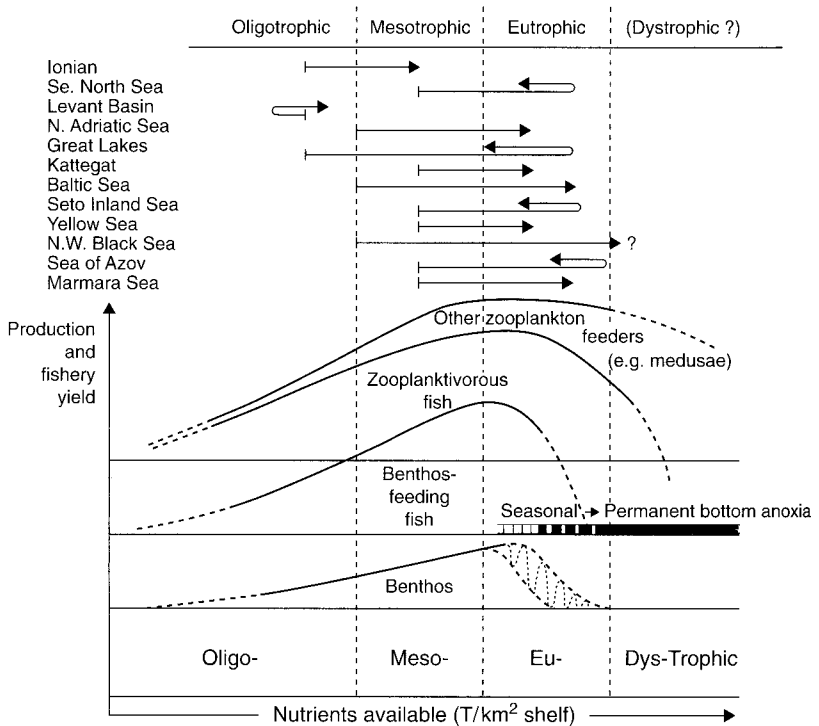


Figure 11. A speculative concept of fisheries change with nutrient enrichment. Redrawn from Caddy (1993). A qualitative progression was suggested from a review of patterns in different enclosed and semi-enclosed seas. The recent trajectory of different systems with respect to trophic status is indicated at the top. The bottom suggests a progression of change in the structure of food webs and composition of fisheries prior to a dramatic loss of yield with permanent bottom anoxia.

which become more consequential at high loading. The progression to anoxia, even in deeper, unvegetated systems, begins a decoupling of the functional connection of pelagic and benthic food webs (cf. Pearson and Rosenberg, 1978; Oviatt et al., 1986). Sediments become uninhabitable and only selected organisms thrive (less "choice" for some commercial fish species). Eventually, the benthos is lost totally, and further pelagic consumer food-web changes follow, sometimes with lags typical of longer-lived species.

Caddy's image of eventual collapse can be compared with some other trends. There appears to be a fundamental relationship between primary production in the water column and fisheries yield of different marine areas, as well as lakes (Nixon, 1998). Interestingly, with increasing production (such as is stimulated by higher

nutrients), the efficiency of conversion to fish appears to increase, not decrease. It has been suggested that this could relate to the nutritional quality (higher N for protein) of the phytoplankton (Iverson, 1990; Nixon, 1992). The trend compiled by Nixon does not in any way suggest a fisheries collapse at high production, although his summary does not include eutrophic/hypereutrophic areas with sustained productivity $>500 \text{ g C/m}^2/\text{year}$. Judging from this level compared with Figure 10, perhaps protecting against anoxia will generally prevent wholesale fisheries collapse due to eutrophication, but it will not prevent shifts in fish and shellfish, nor the loss of some species that are most valued by humans.

6. AFTERWORD, 2007

In the ~ 7 years since this original chapter was written, there has been high interest in marine eutrophication and countless measurements of N in the environment. In an estuaries chapter for an update of Capone and Carpenter's 1983 book, *Nitrogen in the Marine Environment* (Capone et al., in press), Boynton and Kemp rightfully describe this research area as "hyperactive." Major compendia have been published by several professional societies – at least two with missions that focus on aquatic science and coastal waters (Estuarine Research Federation, see Rabalais and Nixon, 2002; and American Society of Limnology and Oceanography, see Smith et al., 2006) and one that integrates across ecological and human health (*Journal of the National Institute of Environmental Health Sciences*, see McGeehin and Rubin, 2001). There are many new papers in the primary literature and there have been reviews and historical perspectives on individual systems or regions (e.g., Rabalais, 2002; Rabalais et al., 2002; Smith, 2003; Smith et al., 2003; Turner and Rabalais, 2003; Kemp et al., 2005). In all the frenzy, a cautious sense of consensus would be:

"...progress is being made in our ability to understand, manage, and perhaps mitigate the impacts of recent and widespread inadvertent fertilization of the coastal marine environment, [but that] ...quantifying the relationship between nitrogen or phosphorus inputs to coastal marine systems and particular responses remains a scientific challenge." (Rabalais and Nixon, 2002).

The results of new millennium's research are neither fully assimilated nor synthesized. I have chosen only to note several themes within the ~ 2000 –2007 literature. The themes, in part, reflect on Cloern's (2001) thoughtful classification of past, present, and future mental models for coastal eutrophication research and management. Cloern suggested a continuing evolution in thinking about the complexity, diversity, and perspective on coastal marine responses, from

- earliest (past) models (\sim simple input-response concepts for a few prime symptom parameters, borrowed from early limnological successes with phosphorus), to

- “present” models (~more complex sets of direct and indirect responses [Figure 1], with overall variability in different system responses as a function of system attributes that act as filters or moderators [some of which are referenced in Sections 3.3, 4, and 5 including tidal influence, water residence times, turbidity/light, food-web structure]), and toward
- “future” models (which consider N as one of many interacting stressors, and which view the coastal zone in a larger earth system perspective, more firmly embedded within social and economic frameworks).

6.1. Increasing Scale of the Issue

Humans have significantly altered the global N cycle, nearly doubling the amount of bio-available (reactive) N in circulation compared with pre-industrial times (Galloway et al., 1995; Galloway and Cowling, 2002). One consequence is an increasing N input from terrestrial systems through both watershed and airshed sources. Since 2000, there are many new records of local, watershed/catchment-level scales of atmospheric, surface, and groundwater inputs to estuaries (e.g., Cloern, 2001; Bowen and Valiela, 2001; Valiela and Bowen, 2002; Kemp et al., 2005; Clarke et al., 2006; Boynton and Kemp, in press). Increases in N loading to the coastal marine environment have been confirmed at more regional levels and suggested at the global scale, with projections of particular vulnerability in certain regions of the globe (e.g., Howarth et al., 2002; Seitzinger et al., 2002; Paerl et al., 2002). Human population density and nutrients through runoff (affected by human activities on the landscape and on hydrological cycles) are the key independent variables of the global predictive regression equation for loading of Smith et al. (2006) who suggest, “Apparently human activities have increased DIP and DIN (Dissolved Inorganic P and N, respectively) above natural fluxes by more than a factor of three, and those changes appear to be recognizable on time scales as short as two decades.”

While the scale of *effects* also seems to have increased with N loading, so has our recognition of the scale of the issue. Eutrophication of freshwater and marine systems is unequivocally termed a “global problem” (Rabalais, 2002; Smith, 2003). Scavia and Bricker (2006) use assessments to “document that N-driven coastal eutrophication is widespread and increasing in the US.” Others see some prime symptoms such as lowered DO as: (a) very likely due to excess N enrichment, (b) occurring worldwide, and (c) having increased rapidly within the past two decades. Diaz (2001) suggested “Oxygen deficiency (hypoxia and anoxia) may very well be the most widespread anthropogenically induced deleterious effect in estuarine and marine environments around the world.” Kemp et al. (2005) provide a detailed chronology and synthesis for one of our best studied systems, Chesapeake Bay, and another revealing chronology has been reconstructed for the Adriatic Sea (Sangiorgi and Donders, 2004); each of these systems has had hypoxia/anoxia issues, a response seen as intermediate to late in a progressive enrichment sequence. *These and other studies continue to support and refine the image of a progression of autotrophic and secondary effects and thresholds reached with increased*

N (e.g., Havens et al., 2001; Rabalais, 2002; Scavia and Bricker, 2006; Boynton and Kemp, in press). If DO characteristically does decline to levels of hypoxia/anoxia at loading rates beyond which other “significant” and/or “undesired” ecological changes have already occurred, then concurrent with Diaz’s and other evidence (Section 4.3) of DO losses worldwide, there must be substantial growth of many other unmeasured, undocumented effects.

6.2. “Simple” Models and Evolving Perspectives for Aquatic Ecosystems

A strong consensus exists that N is the primary cause of eutrophication in many coastal ecosystems, even though there are locations and times where excess P and the availability of Si play a role (Howarth and Marino, 2006). Many studies suggest control of both N and P is wise. There is no dispute of the more prominent, but again not exclusive, role for P in freshwaters (Schindler, 2006; see also Blomqvist et al., 2004).

There continue to be interesting contrasts in freshwater and marine perspectives on eutrophication. Estuarine science has been more reluctant to draw cross-system generalizations that long ago stimulated freshwater eutrophication research (see Introduction). The continuing perspective of some estuarine scientists on application of simple limnological models was noted by Smith (2006):

“In a synthesis of our knowledge of coastal marine eutrophication, Richardson and Jorgensen (1996) concluded that there are essential differences between freshwater and marine environments that prevent us from simply applying knowledge gained from limnological studies to the marine environment. ...Sharp (2001) took a much stronger view, asserting that limnological studies and concepts and observations often lead to incorrect conclusions when they are applied to estuarine and coastal marine waters.”

Smith (2006) counters that marine coastal plankton growth from N and P enrichment is highly consistent with the general pattern previously reported in the limnological literature for freshwater lakes and reservoirs. Sections 4.1 and 4.2 and Figure 7 in this chapter report a similar general predictive response in aquatic primary production, if freshwater and marine systems are scaled for the more limiting nutrient of each system and the water residence time is taken into account. Section 4 noted that it is harder to find marine coastal systems of low loading rates, compared with lakes, and this is a theme of Guildford and Hecky (2000). Havens et al. (2001) found support for the general model of transition from vascular plant to algal dominance with enrichment of shallow water systems (Section 4.4) when several detailed *freshwater and marine* examples were compared.

Recent estuarine studies *have* successfully applied the basic concepts of a limnological model to coastal marine settings. These include Boynton et al. (1996) and Dettman (2001). Dettman was able to relate annual N inputs and N concentrations, using freshwater residence time to model first-order internal loss and net export;

the model was adapted directly from the Dillon and Rigler (1974)/Vollenweider (1976) approach. Dettman's simple model broadly applied to a group of north temperate estuaries with a range of physical and morphometric characteristics, loading rates, and biogeographical settings. Meeuwig et al. (2000) developed a Chl *a* – TP regression model in the Baltic and also empirical regression models of Chl *a* with land use/estuarine mean depth for systems dominated by nonpoint source inputs. Similarly, Meeuwig (1999) successfully derived a simple regression model, with Chl *a* described as a function of land use/estuarine morphometry for a set of systems in eastern Canada. Kauppila et al. (2003) report success in relating DO to land use, mean depth, and fetch, as well as near-bottom TN. These studies have provided some new regional examples to complement those in Sections 4 and 5. We are indeed able to develop some simple regression models relating nutrient sensitive response parameters to nutrient inputs and concentrations in many regional settings, and these make use of simple, fundamental mass-balance concepts of limnological models. A trick is in knowing if, or how finely, we must divide regions, and which additional parameters or modifying aspects (Cloern 2001) might have particular relevance in given cases. A fundamental utility is there through correlations and other evidence, even if satisfying predictive power in each individual case remains elusive.

There are indeed interesting and important differences between lakes and marine systems (cf. Nixon, 1988). But in part, a “but estuaries-are-so-different-and-each-so-unique” view comes from focusing on site to site variation (see also Howarth and Marino, 2006), and sometimes from using only two (or few) quite different settings for comparison, rather than a large suite of systems. Two lakes are often as different as a lake and an estuary, so there is a problem in that perspective. Yet there are also opposing perspectives that come from viewing the same information. Cloern (2001) cites the results of Meeuwig, 1999; (see also Meeuwig et al., 1998) as a part of case for freshwater/marine differences that limit utility of the most simple limnological models, whereas Meeuwig's data are used by Smith (2006) to conclude there are fundamental similarities in freshwater and marine autotrophic responses to nutrients. Guildford and Hecky (2000) suggest there is a similarity of marine and freshwater plankton growth, where differences may relate to local stoichiometric conditions more than an underlying fundamental freshwater/marine difference.

There are a number of issues that perpetuate an apparent dichotomy in view as to the similarity/dissimilarity of freshwater and estuarine responses. First, it's hard for many to think of response as “similar” when one class of systems is more generally based on P and one more generally based on N. Second, coastal marine inspections have not always included estuarine residence times and their impact on simple input-response predictions, which is necessary to be consistent with the early and successful limnological models (Dillon and Rigler, 1974; Vollenweider, 1976; see also Section 4.2). These models, although still simple, may edge into Cloern's (2001) description of a more complex, “present” conceptual model because they

have physical modifiers of response embedded within them. Third, it has always been true that the simplest freshwater empirical models work generally, but fail to be highly predictive in site-specific cases (see Section 4). Fourth, because of those limits to predictability, and as a parallel to the evolving model of complexity for coastal marine eutrophication (Cloern, 2001), there has also been an evolving model for freshwater eutrophication. The limnological model concept and evolution is summarized by Schindler (2006). Over the decades, additional modifying factors and interacting stressors in lake settings have been examined, including: the significance of internal loading (especially in shallow systems with nutrients stored in sediments), food-web structure and trophic cascades, interacting stressors, stoichiometric issues, history, biogeography and related biogeochemistry, climate, etc. In the past few decades, perhaps we have lived through a pendulum swing. We have changed the level of confidence desired for prediction, and/or the precise time and space objectives. Read a recent perspective of Schindler (2006) on predicting nutrient-related responses across different trophic structures:

“Overall there are enough differences in the responses of biomanipulated lakes to suggest that the result will depend greatly on the complexities of the individual lake communities.”

Contrast that with a very recent estuarine view:

“One could make a statement that sufficient data are now available to initiate a more comprehensive, comparative synthesis of estuarine primary production (both pelagic and benthic) that considers a wider selection of independent variables and uses dimensional scaling to the extent needed to ensure adequate comparability among different estuarine locations. In short, estuarine ecologists need to take a lesson from the limnologists who began doing that some 30 years ago (e.g., Vollenweider, 1976) and developed tools useful for both scientific understanding and for lake management” (Boynton and Kemp, in press)(See Figures 7 and 10 for an attempt at a primitive scaling for some factors.)

If one were to compare these admittedly select snippets of comments with freshwater and marine eutrophication scientists' statements in the late 1970s, one could get the impression that the implied predictive capability relating to enrichment in lakes versus estuaries has almost gone topsy-turvy over three decades. The truth is, both fields have long appreciated the diversity of systems and complexity in responses, but have been somewhat independently pushing the limits to predictability.

In the past half decade or so, there has been ever growing evidence that factors like estuarine morphology, water residence times, water column stratification/mixing, turbidity and light, and nutrient stoichiometry all can influence the expression of responses to nutrient loading. It would be hard to capture all the new efforts and emerging ideas on other factors that modify responses, particularly as more systems

are being studied outside the traditional north temperate areas of study. Suffice to say that research to extend effects of N to secondary production, and to higher trophic levels including fish, and to relate these with habitat degradation, are now more vibrant research areas (cf. Section 5). Papers on these topics include several in Rabalais and Nixon (2002). For example, Deegan (2002) and Breitburg (2002) provide views on the interaction of nutrients with habitat and DO, as they may affect fish. Reports have detailed changes in benthos or other secondary consumer and substantiated these changes as a very connected response to N enrichment (Nixon and Buckley, 2002; Tewfik et al., 2005; Boynton and Kemp, in press). These efforts provide data that may ultimately clarify the conceptual image and stages of Caddy's (1993) fisheries perspective (presented in Section 5 and Figure 11).

A landscape perspective, and the role of watersheds in driving progressive increases in N loading and effects to shallow coastal systems continue to become more explicitly recognized. Some significant papers on this subject relate directly to DO, but also to seagrasses (e.g., Bowen and Valiela, 2001; Martinetto et al., 2006; Turner and Rabalais, 2003). For example, Hauxwell et al. (2003) use historical conditions to link watersheds and *Zostera* decline. They report loading thresholds that are very low, but still consistent with previous work summarized in Section 4.4 and Figure 10.

Landscape changes often bring a host of other, co-occurring stressors into clearer view, so studies in this area force multi-stressor concepts of understanding how humans can change the coastal zone. Interacting stressors also include climate change (Cloern, 2001; see also Schindler, 2001). Oviatt's (2004) studies on coastal food web and fisheries changes during decades of climate warming should serve to remind us: it is daunting to unravel a singular effect of a decade or so of increasing nutrients against other synergistic or antagonistic long-term trends. The interaction of climate, nutrients, primary producers, and enrichment symptoms has been reported (e.g., Paerl, 2006; Paerl et al., 2006) in studies examining large storms and unmistakable (not at all subtle) hydrological forcing events that deliver nutrients in huge pulses. Storms also alter mixing and stratification dynamics in estuaries, fundamentally altering the time and space scales of overenrichment symptoms. A number of studies have recognized interactions with river flow, nutrients, and hypoxic volume over historical time (e.g., Turner and Rabalais, 2003; Hagy et al., 2004; Kemp et al., 2005). Fittingly, with Boynton and Kemp's notion of "hyperactivity," the study of multiple-stressor interactions, which were once part of Cloern's (2001) "future" conceptual model of eutrophication, has already edged well within the present area of active research.

6.3. A Widening of Perspective in Considering Nitrogen Enrichment Effects

Recent worldwide trends and some newer research findings seem to be acting to seriously broaden thinking about how we should study and evaluate the responses and consequences of coastal marine nutrient enrichment. In general, this is consistent with a course that Cloern (2001) suggested.

A major contributor to this broadening, I believe, is the continuing confirmation of *connectedness* between human activities at landscape and regional levels, resultant nutrient loading, and cumulative-scale effects. The link between landscape-level properties and N effects has been confirmed and sometimes quantified through new isotopic indicators, in different regions, and with respect to different effects (e.g., Meeuwig et al., 2000; Bowen and Valiela, 2001; Valiela and Bowen, 2002; Hauxwell et al., 2003; Kauppila et al., 2003; Martinetto et al., 2006). In the coastal freshwaters of the Great Lakes Basin, linkage between watersheds/basins and their coastal habitat/ecosystem condition is a similarly active research area (e.g., Danz et al., 2007). Moreover, the recognition that connectedness can occur on very large watershed scales is notable (e.g., Turner and Rabalais, 2003). An important recognition of Smith et al. (2003) is that *both* human population and runoff are important to describing and forecasting coastal loading increases. Human *activities* alter the landscape and through this, and in other ways, impact hydrologic cycles and thus runoff.

The connectedness notion, and also some documentation of the history of enrichment over long time frames (e.g., Kemp et al., 2005), opens up the scale to be examined in both time and space. All acting together, it forces us all to get up out of the water and see the estuary as within an airshed–landscape–hydroscope, which is a view that brings interacting factors such as hydrology, climate, and so on a bit more easily into focus. That opening up of scale, along with a recognized reality of continued human population growth, seems to have engendered a qualitatively new perspective. The questions previously being asked – What might be “undesired” in the “local effects” sense (e.g., HABs, SAV, DO, food-web change), how do we quantify those, and how do we curtail them? – can shift to: What “trade-offs” we are willing to accept on broader scales?

The emerging new frame includes a perspective of terrestrial benefits that may outweigh their costs in downstream systems. With it comes a stronger recognition of coastal waters as part of larger and more coupled systems. Scientists are thrust into considering: Do we know all these linkages well enough to be confident in the “trade-offs” that might be evaluated? A variety of economic and societal issues thus have been entering into this kind of picture (see Boesch, 2002; Doering, 2002; Hoagland et al., 2002). Consistent with this larger perspective, the beneficial effects of fertilization (e.g., increased fisheries yield; see Caddy, 1993; Nixon and Buckley, 2002) that might occur at different scales come better into view even though this has, of course, always been a prime management issue. “Over”-enrichment (Rabalais and Nixon, 2002) too, is hardly a new thought, but an apt term. It makes clear that the focus is on acceptable and unacceptable effects and the thresholds between them. But scientists now more actively recognize that the “system” to be examined in this context is not restricted to narrowly defined boundaries of the water and sediments within the coastal marine system. A fascinating time – some old themes and challenges writ ever larger and ever more complex.

I believe the startlingly rapid broadening in perspective is one that is much needed scientifically, and is appropriate to a management context. But I have to admit

that I want to hold on to my naiveté. We could too easily be shifting from a view that humans live fully within the ecosystem and almost spiritually value its condition, to a view that humans live distinctly outside the ecosystem. And in the new perspective one could develop a hubris that we really have an ability to decide how best to engineer its services for our use.

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- Figure 11 (from Caddy, 1993) – the author and CRC Press.

Thanks also to the EPA-ORD-NHEERL group working on nutrients in coastal systems, for keeping me informed of some developments in the saltier world. This 2007 chapter has drawn from an outrageously large body of work and insights of many researchers on coastal eutrophication, but the interpretation of trends and editorializing are my own. Although I did not attempt a comprehensive review of new work for the 2nd edition, it was hard to limit the new citations (post 2000–2001) to only 50 or so.

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