1. Introduction

The availability of nitrogen (N) most frequently limits autotrophic production of organic matter in the world’s estuarine, near-shore coastal and open ocean waters (Dugdale, 1967; Nixon, 1986, 1995; Ryther and Dunstan, 1971). As such, the fertility of these waters is often closely controlled by N inputs, provided either internally by regeneration of pre-existing N and biologically-fixed atmospheric N$_2$, or supplied externally (i.e., “new”) as combined N sources delivered via surface runoff, sub-surface groundwater or atmospheric deposition (Fig. 11.1). Because marine ecosystems lose some fixed N via denitrification and burial, “new” N supply is essential for compensating for these N losses, and ensuring productive conditions. However, during the past century humans have significantly altered the balance between “new” N inputs and N losses in the marine environment through large-scale use of synthetic N fertilizers in agriculture, rapid expansion of industrial and transportation-related fossil fuel combustion and coastal urbanization (Codispoti et al., 2001; Rabalais, 2002; Vollenweider et al., 1992). Over this time frame, terrigenous discharge and atmospheric N emissions have increased 10-fold (Holland et al., 1999; Howarth et al., 1996) and continue to grow as human development expands in coastal watersheds (Vitousek et al., 1997).
For at least 50 years, researchers have recognized this growing imbalance, especially in estuarine and coastal waters where N over-enrichment has fueled accelerated primary production, or eutrophication, which Nixon (1995) defined as “an increase in the rate of supply of organic matter to an ecosystem.” From a food web perspective, eutrophication is a condition where nutrient-enhanced primary production exceeds the ability of higher ranked consumers to utilize it. D’Elia (1987) characterized this condition as “too much of good thing,” or over-fertilization of N-limited marine ecosystems with “new” N, the bulk of it being anthropogenic (Galloway and Cowling, 2002; Howarth et al., 1986; Vitousek et al., 1997). Symptoms of N-driven eutrophication include: (1) subtle increases in plant production, to (2) changes in primary producer community composition, (3) rapidly accelerating algal growth, (4) visible discoloration or blooms (Jørgensen and Richardson, 1996; Paerl, 1988; Smetacek et al., 1991), (Fig. 11.2), (5) losses in water clarity, (6) increased consumption of oxygen leading to ecosystem-scale oxygen depletion (hypoxia, anoxia) (Hagy et al., 2004) (Fig. 11.3), which is stressful to resident fauna and flora, and can lead to (7) elimination of habitats (Diaz and Rosenberg, 1995; Rabalais and Turner, 2001). Lastly, excessive N enrichment, relative to other nutrient inputs, can alter nutrient input balances and cycling, including shifts to phosphorus and silicon limitation of

Figure 11.1 Schematic diagram illustrating sources, transformations and fates of nitrogen (N) along the estuary to ocean continuum. Surface, subsurface and atmospheric pathways of externally-supplied or new N inputs attributable to anthropogenic activities are shown as are internal N cycling. The combined anthropogenic N inputs are shown as a thick arrow (upstream), which decreases in thickness downstream as a portion of the N inputs settles to the bottom sediments and is buried and/or denitrified (DNF). NF represents nitrogen (N₂) fixation, a biologically-mediated new N input. The linkage of anthropogenically-enhanced N inputs to accelerated primary production or eutrophication, and its trophic and biogeochemical fate are also shown. In many estuarine and sound systems, primary production and phytoplankton biomass are maximal in mid-system locations, where adequate new N loads and decreasing rates of flushing (increasing residence times) overlap. The resultant chlorophyll a maximum (Chl a max) is characteristic of estuarine systems in which residence times are long enough to allow periodic phytoplankton blooms to accumulate.
primary production in receiving estuarine and coastal waters (Dortch et al., 2001; Lohrenz et al., 1999; Sylvan et al., 2006).

Anthropogenic or cultural eutrophication has been closely linked to population densities in coastal watersheds (Nixon, 1995; Peierls et al., 1991; Vitousek et al., 1997; see also Chapter 9 by Seitzinger and Harrison, this volume; Chapter 18 by Boynton and Kemp, this volume). A significant, and in many instances increasing, proportion of “new” N input can also be attributed to remote sources delivered through airsheds and subsurface aquifers outside the immediate watershed (Galloway and Cowling, 2002; Jaworski et al., 1997; Paerl et al., 2002). The area of an airshed can greatly exceed that of a watershed for a specific estuary or coastal region. For example, the airshed of the Baltic Sea includes much of Western and Central Europe (Hov et al., 1994), and the airsheds of the US’s two largest estuarine ecosystems, the Chesapeake Bay and Albemarle-Pamlico Sound, are from 15 to over 30 times the size of their watersheds (Paerl et al., 2002) (Fig. 11.4). Thus, the airshed of one region may impact the watershed and receiving waters of another (Dennis, 1997) (Fig. 11.4). Furthermore, atmospheric N inputs do not stop at coastal margins. Along the North American Atlantic continental shelf and Mediterranean coast, atmospheric N inputs more than match riverine inputs (Herut et al., 1999; Jaworski et al., 1997; Paerl et al., 2002), underscoring the fact that this is a regional, if not global, issue (Galloway and

**Figure 11.2** Estuarine and coastal phytoplankton blooms symptomatic of nitrogen-enhanced primary production or eutrophication. Clockwise, starting with upper left; (A) cyanobacterial blooms in the Gulf of Finland, Baltic Sea (photo courtesy of P. Moisander); (B) dinoflagellate red tide bloom off the coast of Japan (photo courtesy of ECOHAB Program); (C) dinoflagellate/diatom bloom near the coast of Hong Kong, China (photo courtesy ECOHAB Program), and (D) cyanobacterial bloom on the St. Johns River Estuary, Florida (photo courtesy J. Burns).
Figure 11.3  Low oxygen, hypoxic (less than 2 mg L$^{-1}$ O$_2$) conditions in the bottom waters of the N-enriched, eutrophic Neuse River Estuary, located in coastal North Carolina (upper left hand side). Upper right hand side shows the lateral extent of hypoxia along the central channel of the estuary during summer (data from the Neuse River Monitoring and Modeling Program, www.marine.unc.edu/neuse/modmon, see Buzzelli et al., 2002). The lower part of this figure shows both the spatial and temporal extent of hypoxia in this estuary (adapted from Buzzelli et al., 2002).
For example, in truly oceanic locations (e.g., Bermuda), North American continental atmospheric N emissions (reduced and oxidized N) are commonly detected and significant (Luke and Dickerson, 1987; Prospero et al., 1996). Likewise, islands in the North and mid Pacific receive N deposition originating on the Asian continent (Prospero et al., 1989).
It has been estimated that riverine and atmospheric N inputs in the North Atlantic Ocean basin are at least equal to and may exceed N inputs by biological N$_2$ fixation (Howarth et al., 1996; Paerl and Whitall, 1999). Therefore, our understanding of marine eutrophication dynamics needs to consider a range of scales including ecosystem, watershed, regional and global.

2. The Evidence for the Role of Nitrogen in Marine Eutrophication

A vast number of studies conducted over the past 50 years have shown that N enrichment is a primary causative agent of marine (estuarine and oceanic) eutrophication (Dugdale, 1967; D’Elia et al., 1986; Nixon, 1989, 1995; Ryther and Dunstan, 1971; Smetacek et al., 1991). These include:

- *In situ* evidence of the spatial and temporal relationship of N inputs vs. primary production responses (Fig. 11.5)
- Nutrient addition bioassays where N enrichment has been shown to stimulate primary production (D’Elia et al., 1986; Dugdale, 1967; Fisher et al., 1992, 1999; Oviatt et al., 1995; Paerl et al., 1987, 2003; Pennock et al., 1994; Piehler et al., 2004) (Figs. 11.6 and 11.7)
- Paleoecological studies showing that historic increases in anthropogenic nutrient (N-dominated) loading have led to eutrophication (e.g., Chesapeake Bay, Cooper and Brush, 1993)
- Uptake studies which have shown that at ambient concentrations and supply rates, N limitation is widespread (Harrison and Turpin, 1982; Harrison et al., 1987; Syrett, 1988)
- Correlative budgetary studies in which N loading or supply rates were directly related to daily or annual rates of primary production in diverse coastal ecosystems (Nixon, 1986, 1995) (Fig. 11.8)
- Stoichiometric analyses that have shown, relative to the availability of carbon (C), phosphorus (P), silicon (Si) and other nutrients, N often falls below the nutrient supply ratio needed to sustain balanced plant growth (i.e., Redfield ratio of 105:16:1 for C:N:P, Redfield, 1958; Smith, 1990)

*Figure 11.5* Relationships, in time and space, between dissolved inorganic N (as nitrate, the dominant new N source), dissolved organic N, inorganic phosphorus (as orthophosphate) concentrations and phytoplankton biomass (as chlorophyll a) along the axis of the Neuse River Estuary, North Carolina. The locations shown on the y axis represent the distance between an upstream location at Streets Ferry Bridge and a downstream location below Cherry Point (both are shown as ♦ on map). Phytoplankton accumulate as distinct concentrations of biomass, which, depending on chlorophyll a levels, may be as visible blooms. During much of the year, nitrate concentrations decreased sharply concurrent with increased chlorophyll a concentrations forming the chlorophyll a maxima, while a similar relationship between orthophosphate and chlorophyll a is absent. This serves as in situ evidence that dissolved inorganic N supply is most likely controlling or “limiting” phytoplankton production. Dissolved organic N concentrations appear to closely track phytoplankton biomass, suggesting that phytoplankton may be a source of this form of N.
Neuse River Estuary
North Carolina

October 2000

January 2001

April 2001

July 2001

(For legend see opposite page)
Specific case studies (e.g., Kaneohe Bay, HI, Chesapeake Bay, Neuse River-Pamlico Sound, Long Island Sound, Narragansett Bay, Baltic Sea, Coastal North Sea, Northern Adriatic Sea, Northern Gulf of Mexico) have shown that increasing N loads are directly linked to accelerated eutrophication (cf., Boesch et al., 2001a,b; Elmgren and Larsson, 2001, Fisher et al., 1999; Nixon, 1995; Paerl et al., 1998, 2004; Rabalais, 2002; Smith et al., 1981; see also Chapter 18 by Boynton and Kemp, this volume).

Geochemists have pointed out that theoretically, nitrogen (N$_2$) fixation should compensate for N-limitation in the worlds oceans. According to this argument, P availability (which is assumed to control N$_2$ fixation) is ultimately limiting primary production (cf., Doremus, 1982; Tyrell, 1999). This argument operates over geological time scales and requires predictable and consistent biology (i.e., N$_2$ fixation is solely and consistently controlled by new P inputs). However, the theory does not seem to be compatible with biological time scales and the complex environmental controls of N$_2$ fixation beyond phosphorus availability (Paerl, 1990; see also Chapter 3 by Carpenter and Capone, this volume and Chapter 7 by Mulholland and Lomas, this volume). In many estuarine and coastal systems, N$_2$ fixation does not automatically “turn on” when P is adequate and N is limiting. Experimental data indicate that other factors, including N:P supply ratios, iron (Fe) limitation, organic matter availability, physical constraints such as turbulence, irradiance, and potentially “top down” consumption processes control N$_2$ fixation (Howarth, 1988; Paerl, 1990). As a result, this argument has limited application to managing coastal eutrophication.

Figure 11.6 Results from seasonal in situ bioassays in the southwest basin of Pamlico Sound. Bars are means of 5 replicates and error bars are on standard deviation. The top panel shows the response of primary productivity of the natural phytoplankton community to the addition of nitrate (+N, 20 μM-N), phosphate (+P, 5 μM-P), nitrate and phosphate (+NP), and the un-amended control. The bottom panel shows the response of chlorophyll a to the same treatments. Relative to controls, strong N limitation was observed in Pamlico Sound despite high load of N to the upstream Neuse River Estuary.

- Specific case studies (e.g., Kaneohe Bay, HI, Chesapeake Bay, Neuse River-Pamlico Sound, Long Island Sound, Narragansett Bay, Baltic Sea, Coastal North Sea, Northern Adriatic Sea, Northern Gulf of Mexico) have shown that increasing N loads are directly linked to accelerated eutrophication (cf., Boesch et al., 2001a,b; Elmgren and Larsson, 2001, Fisher et al., 1999; Nixon, 1995; Paerl et al., 1998, 2004; Rabalais, 2002; Smith et al., 1981; see also Chapter 18 by Boynton and Kemp, this volume).

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Figure 11.7 Time course of nutrients and chlorophyll a in the nutrient-addition bioassay performed on a Chesapeake Bay above-pycnocline water sample from station TF1.5 (Patuxent Tidal Fresh Region) in September 2000. Nutrients were added as one dose just after the initiation of the bioassay (0 days). N was added as NH$_4^+$ (NH$_4$Cl) and P was added as PO$_4^{3-}$ (NaH$_2$PO$_4$). Added NH$_4^+$ was rapidly depleted, whereas excess P was present in all treatments except +N. The bioassay was incubated at 60% of surface irradiance. Chlorophyll a responded strongly to any N addition, but not in the control nor to P additions (adapted from Fisher and Gustafson, 2004).
Figure 11.8  Upper frame: Direct relationships between dissolved inorganic N input and primary production in a various estuarine and coastal ecosystems*. Figure adapted from Nixon et al., 1996. Lower frame: Direct relationship between dissolved inorganic N input and phytoplankton biomass, as mean annual chlorophyll a content of several Western Australian estuarine systems. Figure adapted from Twomey and Thompson, 2001.

*Details of systems: The open circles are for large (13 m3, 5 m deep) well-mixed mesocosm tanks at the Marine Ecosystems Research Laboratory (MERL) during a multi-year fertilization experiment (Nixon, 1995; Nixon et al., 1986). Natural systems (solid circles) include; (A) Scotian shelf—DIN from Houghton et al. (1978), production from Mills and Fournier (1979); (B) Sargasso Sea—DIN from Jenkins (1988), production from Lohrenz et al. (1992) mean of 1989 and 1990 values of 110 and 144 g C m⁻² year⁻¹; (C) North Sea—DIN from Laane et al. (1993) assuming that the ratio of DIN/TN
The extent to which N loading promotes eutrophication varies greatly among marine ecosystems. Receiving waters exhibit varying sensitivities to N and other nutrient (P, Fe, Si) loads that are controlled by their size, hydrologic properties (e.g., flushing rates and residence times), morphologies (depth, volume), vertical mixing characteristics, geographic and climatic regimes and conditions. In addition, the magnitude and distribution of N in relation to other nutrient loads can vary substantially. In waters receiving very high N loads relative to requirements for sustaining primary and secondary production, other nutrient limitations may develop. This is evident in estuarine and coastal waters downstream of rivers draining agricultural regions that are highly enriched in N. Examples include the Po (Italy), Rhine (W. Europe), Yangtze (China) and Mississippi (USA), the Ganges (India) and the Nile (Egypt) rivers that over the past century have experienced very large increases in N loading (cf. Nixon, 2003; Rabalais, 2002). Here, excessive N loading may be saturating in-shore primary production, leading to either P and Si co-limitation or exclusive P or Si limitation (Conley, 2000; Dortch and Whitledge, 1992; Ferber, 2004; Lohrenz et al., 1999, Sylvan et al., 2006). Further offshore, receiving waters have remained N undersaturated and hence exhibit chronic N limitation (Justic et al., 1995a,b; Rabalais et al., 1996; Smetacek et al., 1991). Because they have remained N limited even in the face of accelerated N loading, off-shore waters can support additional N-driven eutrophication (i.e., they are still in the initial stages of eutrophication) (cf. Codispoti et al., 2001; Smetacek et al., 1991).

N-driven eutrophication can exert feedbacks on internal N cycling, which may alter within-system N availability, N budgets and subsequent eutrophication potentials. Numerous studies have shown organic matter loading, sedimentation and the extent of bottom water/sediment hypoxia resulting from eutrophication can regulate key N transformations, including nitrification and denitrification (Fear et al., 2005; Heggie et al., 1999; Henriksen and Kemp, 1988; Boynton and Kemp, 2000; Seitzinger and Giblin, 1996; Smith and Hollibaugh, 1989, 1998; Twilley et al., 1999). These within-system feedbacks can significantly affect N availability, and hence subsequent eutrophication potentials (Eyre and Ferguson, 2002; Smith and Hollibaugh, 1998). For example, in the Baltic Sea the extent of hypoxia formation is thought to control denitrification rates and hence the ability of the system to deplete itself of fixed N (Elmgren and Larsson 2001; Wulff et al., 2001). Lastly, top down effects such as grazing, and removal of grazers by overfishing (Jackson et al., 2001) can significantly alter the flux, availability, utilization and manifestation of N and other nutrient inputs.
3. NUTRIENT, PHYSICAL AND CLIMATIC CONTROLS OF MARINE EUTROPHICATION

While anthropogenic N enrichment has been linked to N-driven eutrophication of estuarine and coastal waters (cf. Boesch et al., 2001a,b; Nixon, 1995; Smetacek et al., 1991), the manner and extent by which eutrophication affects production, nutrient cycling, trophodynamics, and habitat condition are complex products of nutrient inputs, physical features of receiving waters and climatic conditions. In arid regions, the generation, transport and fate of nutrients are quite distinct from wet regions. Similarly, polar, temperate and tropical coastal and oceanic regions can respond in contrasting manners to N and other nutrient inputs. Episodic climatic events, such as large storms and hurricanes, and extreme temperatures strongly modify the degree to which eutrophication is manifested in coastal waters.

As a result, marine waters exhibit highly individualistic responses to N loads over seasonal and longer (multi-annual, decadal) time scales. The degree to which these systems are exposed to tidal exchange and vertical mixing is critical for determining and predicting how they respond to specific N loads (Cloern, 1999, 2001; Nixon, 1995; Vollenweider et al., 1992). Another variable is the manner in which N is delivered, ranging from acute pulsed events such as storms and associated flooding, to longer-term gradual (chronic) increases in N loading associated with more predictive seasonal, annual and interannual hydrologic cycles. There are striking contrasts in estuarine response to N inputs that reflect a range of hydrodynamic, optical and climatic conditions (Cloern, 1999, 2001; Sharp, 1994, 2001). North American examples include the contrast between strong tidally-driven estuarine systems like Delaware Bay and San Francisco Bay, moderately tidally-driven systems like Chesapeake Bay, micro-tidal, lagoonal systems like North Carolina’s Pamlico Sound and Texas’s Laguna Madre, and semi enclosed coastal systems like Florida Bay and the Long Island Sound (Bricker et al., 1999). While these systems may receive comparable amounts of N, they exhibit differential biological responses. Interactions between the biological response and physical forcing features are significant. If the rate of flushing exceeds the growth and doubling rates of the resident phytoplankton communities, the system may be able to tolerate a relatively high N load without showing symptoms of eutrophication. In contrast, under similar N loading but relatively long residence time conditions, phytoplankton growth rates may be able to keep up with or exceed the rates at which populations are flushed from the system. Under these conditions, symptoms of eutrophication (algal blooms, hypoxia) are most strongly felt within the estuary, while under more highly-flushed, short residence time conditions, eutrophication may manifest itself more in downstream coastal waters (Fig. 11.9).

Variation in ecosystem sensitivity and response due to hydrologic variability can be seen in North Carolina’s Pamlico Sound, the US’s largest lagoonal ecosystem and second largest estuary (1700 km²), draining numerous sub-estuaries. This system exhibits N limitation from its upstream oligohaline riverine components to the inlets.
Spatial relationships between the phytoplankton biomass, as chlorophyll $a$, and estuarine freshwater discharge conditions in the Albemarle-Pamlico Sound estuarine system North Carolina, USA. Surface water chlorophyll $a$ concentrations were estimated using an aircraft-based SeaWiFS remote sensing system (Harding and Perry, 1997), calibrated by field samples that were analyzed fluorimetrically (Paerl et al., 1995). Under relatively low flow and long residence time conditions, phytoplankton biomass is concentrated in the upstream reaches of the estuarine tributaries. Under moderate flow, phytoplankton biomass maxima extend further downstream. Under high flow (i.e., short estuarine residence time) phytoplankton biomass maxima are shifted further downstream into the open Pamlico Sound.

mediating exchange with the Atlantic Ocean (Peierls et al., 2003; Pielcher et al., 2004; Twomey et al., 2005). If high flow coincides with the period of maximum productivity (early spring through summer), the zone of maximum phytoplankton biomass accumulation (chlorophyll $a$ “maximum”) tends to be in the downstream estuarine segments (Valdes-Weaver et al., 2006). In contrast, if low flow accompanies maximum productivity, or if drought conditions follow periods of high nutrient discharge, maximum biomass accumulation typically tends to occur in the upstream oligo- to mesohaline regions of the estuaries (Figs. 11.5 and 11.9). In either case, productivity may be enhanced by similar N loads, but depending on the means of N delivery (high vs. low flow discharge), ecosystem responses and impacts vary.

The Baltic Sea exemplifies the strong interactions between physical and chemical forcing features in terms of how eutrophication is expressed. There is general
agreement that excessive P and N loading have led to the eutrophication of this large brackish water system (Larsson et al., 1985), although the history of productivity and algal boom responses to increasing levels of nutrient loading remains unclear (Elmgren, 1989; Elmgren and Larsson, 2001). On interannual and decadal time scales, the magnitudes, duration, biogeochemical and trophic ramifications of eutrophication vary considerably, depending on these interactions. Riverine and atmospheric inputs in the Eastern and the Southern Baltic regions have large impacts on amounts and ratios of new N and P loads controlling productivity (Conley et al., 2002; Elmgren and Larsson, 2001; Savchuk and Wulff, 1999; Turner et al., 1999). Furthermore, the freshwater runoff associated with large rivers draining the eastern, southern and northern Baltic influence both salinity and vertical stratification regimes throughout the sea. On the western side, dense, saltwater inflow from the North Sea is a key determinant of vertical salinity stratification, nutrient fluxes and budgets (Elmgren and Larsson, 2001; Humborg et al., 2002; Savchuk and Wulff, 1999). The interaction of these physical-chemical forcing features in time and space determines the nature, distribution and duration of nutrient limitation, the magnitude and extent of manifestations of eutrophication, and the hypoxic volume of the Baltic basin. This interaction can vary substantially on interannual and decadal time scales (Elmgren and Larsson, 2001; Humborg et al., 2002; Turner et al., 1999). In single or consecutive years of high freshwater runoff combined with high nutrient delivery, productivity and biomass accumulation tends to be relatively high (e.g., spring non-cyanobacterial blooms, followed by large summer cyanobacterial blooms). High delivery of freshwater also maximizes vertical stratification, which in turn favors water column stability and buoyant cyanobacterial bloom species. Under these conditions maximum expression of eutrophication takes place (Wulff et al., 2001). Conversely, relatively dry years tend to reduce N loading, and phytoplankton productivity and biomass accumulation are relatively low. The resultant ecosystem effect is a reduction in eutrophication potential (Conley et al., 2002; Wulff et al., 2001).

Another example of the interaction of physical forcings and nutrients modulating coastal eutrophication was provided by Bledsoe and colleagues (2004). In a 3-year study of the Suwanee River estuary in Florida, excessive phytoplankton growth was controlled by both nutrient loading and residence time. Maximum phytoplankton biomass was recorded during periods when loading from the river was elevated and prevailing on-shore winds decreased the flushing rate from the estuary. Their work illustrated the difficulties of managing nutrients in the context of mitigating natural circumstances such as meteorological conditions.

Very large climatic events, such as hurricanes and associated floods, can overshadow more subtle, longer-term (chronic) trends in eutrophication and water quality decline. The three sequential hurricanes (Dennis, Floyd and Irene) that struck coastal North Carolina during a six week period in fall 1999 and provided a striking example of the interacting effects of anthropogenic N loading and climatic forcing. These storms delivered up to 1 m of rainfall and caused record flooding in the largely agricultural watershed of the Pamlico Sound. The floodwaters rapidly mobilized nutrients and turned the Sound’s sub-estuaries into raging rivers,
discharging nutrient enriched floodwaters directly to the Pamlico Sound (Paerl et al., 2001; Peierls et al., 2003). During normal hydrologic years, these N-limited estuaries remove much of the terrigenously-supplied N, either by uptake and assimilation or denitrification and dissimilation, to the extent that only a fraction of the externally supplied new N makes it past the estuarine filter into Pamlico Sound. During these years, symptoms of eutrophication (algal blooms, hypoxia and anoxia, fish kills) are largely confined to the estuaries themselves (Buzzelli et al., 2002; Paerl et al., 1998, 2006a,b,c) (Fig. 11.9). In contrast, floodwaters following large storm events flush the algal filter and nutrient-rich, low-salinity waters into Pamlico Sound and adjacent coastal waters (Paerl et al., 2006a,c). From a eutrophication perspective, the productivity and algal bloom response takes place in the Sound and coastal waters during storm-prone periods, while during storm-free periods, the productivity response is more confined to the upstream estuarine regions (Paerl et al., 2001; Peierls et al., 2003, Valdes-Weaver et al., 2006). Where and how this largely-anthropogenically generated N load manifests itself is a product of climatic forcing features such as storms and the resultant freshwater discharge patterns (see also Chapter 9 by Seitzinger and Harrison, this volume).

A single large summer-fall hurricane like Fran (1996) or Floyd (1999) can deliver about as much “new” N as the more predictable winter-spring wet period (Fig. 11.10). Thus, during a year supporting “normal” winter-spring runoff, accompanied by one or several summer-fall hurricanes, approximately twice the expected annual N load is delivered to downstream estuarine and sound waters. The eutrophication response to these large N loads can be seen in the form of above-average phytoplankton biomass production in the downstream waters of Pamlico Sound (Fig. 11.9) (Paerl et al., 2006a,c). This demonstrates that the eutrophication gradient can be displaced in response to hydrologic discharge.

The manner and extent to which climatic conditions affect coastal N loading and eutrophication are, to some extent, dependent on land-based N sources and discharge characteristics. In rural agricultural watersheds, a large proportion (as much as 80% in coastal North Carolina) of anthropogenic N is discharged as diffuse, non-point source (NPS) loading. In contrast, in mixed urban and agricultural watersheds such as those in the Chesapeake Bay region, NPS loads account for about half the N discharge, while in largely urban watersheds such as those discharging to the Long Island Sound, point sources (PS) tend to dominate N discharge (80% PS, 20% NPS). NPS-dominated watersheds tend to be far more “leaky” in response to storm-driven N discharge events than PS dominated watersheds. On the other hand, point sources tend to dominate N discharge to estuarine and coastal waters during relatively dry periods (Howarth, 1998). The ecological and management ramifications of these contrasting scenarios are considerable. For example, if receiving waters have a tendency to develop blooms during high flow early spring-summer or fall periods, NPS controls are likely to be more effective at stemming eutrophication. On the other hand, PS controls may prove critical in controlling blooms under mid-summer, low flow conditions. In most instances, combined NPS and PS N input reduction is the most prudent and effective N management strategy for controlling eutrophication on a year-round and longer-term basis (see section VII) (Paerl et al., 2006b).
Figure 11.10 Annual patterns of dissolved inorganic nitrogen (DIN: nitrate/nitrite plus ammonium) loading to the Neuse River Estuary, NC. Years were chosen that represented different hydrologic conditions, including a drought year (1994), a year with relatively high rainfall during the typically rainy winter-spring periods, followed by low rainfall summer conditions (1998), years in which large summer-fall hurricanes affected the watershed (Fran in 1996, Dennis and Floyd, in 1999) and relatively wet years that were impacted by a coastal hurricane that delivered relatively little rainfall (Bonnie in 1998, Isabel in 2003). In years where hurricanes delivered large amounts of rainfall to the watershed (1996, 1999), hurricane-based N loads were on the same order of magnitude as spring runoff based N inputs. It is evident that rainfall amounts and periodicity have very large impacts on the delivery of land-based N to the lower estuary, receiving sound and coastal waters.
Is Nitrogen Nitrogen? Roles of Different Nitrogen Sources in Marine Eutrophication

Externally-supplied N comes in a various forms, including inorganic and organic, reduced (ammonium) and oxidized (nitrate, nitrite) inorganic N, all of which are potentially available to support new production and eutrophication. Laboratory experiments on phytoplankton isolates and bioassays with natural phytoplankton communities have indicated that these contrasting forms may be differentially and preferentially utilized, indicating that, depending on composition of the affected phytoplankton community, some forms are more reactive than others (Collos, 1989; Riegman, 1998; Stolte et al., 1994; see also Chapter 7 by Mulholland and Lomas, this volume). Phytoplankton community composition can also be altered by varying proportions and supply rates of different forms of N (Dortch, 1990; Harrington, 1999; Piehler et al., 2002; Pinckney et al., 1999; Stolte et al., 1994). In addition, specific N compounds may interact with light availability, hydrodynamics and other nutrients, most notably P, Si, Fe, and trace metals, to influence phytoplankton community growth rates and composition (Dortch and Whitledge, 1992; Harrison and Turpin, 1982; Smith, 1990; see also Chapter 38 by Hutchins and Fu, this volume and Chapter 37 by Kudela, this volume). These findings raise important questions about the quantitative and qualitative linkages between new N inputs and marine eutrophication:

- How does the composition of the new N pool influence new production and biogeochemical cycling in N-sensitive waters?
- To what extent does enrichment with certain N forms affect algal bloom potentials, specifically dominance by harmful algal bloom taxa (dinoflagellates, cyanobacteria, etc.)?
- Are some forms of N more potent catalysts of eutrophication than others?
- How are projected changes in human activities in coastal water- and air-sheds related to shifts in the composition of new N inputs, and can we link these shifts to predictable changes in the composition and ecological roles of primary producers catalyzing marine eutrophication?

One example of shifting N inputs is the proliferation of intensive livestock operations in coastal watersheds, which has led to large increases and changes in chemical composition of nitrogenous compounds discharged to estuarine and coastal waters via runoff, groundwater and atmospheric deposition (Galloway and Cowling, 2002; Howarth, 1998; Paerl, 1997). In general, coastal waters under the influence of these operations are experiencing increases in total N loading as well as a shift toward more reduced N (ammonium, organic N) relative to oxidized N (nitrate, nitrite) (Galloway and Cowling, 2002; Howarth et al., 2002). These increases, combined with increases in hypoxia and anoxia in receiving waters, are leading to more ammonium-rich conditions, which will favor algal species able to best exploit this N form. Alternatively, conversion of agricultural to urban lands tends to lead to more nitrate-enriched conditions, potentially favoring plant taxa best able to exploit this N form. Lastly, disruption or destruction of wetlands adversely affects their important N filtering capabilities, resulting in changes in both the amount and composition of N compounds that are processed by these ecosystems.
To improve our understanding of the biogeochemical and trophic roles “new” N composition plays in marine eutrophication, we must combine observational and experimental approaches focused on microalgal and higher plant communities responsible for eutrophication. Approaches include linking loading characteristics of specific N compounds and groups of compounds (e.g., dissolved organic matter) to microalgal growth responses in space and time. This approach relies on: (1) intensive monitoring of N inputs, their distributions and uptake characteristics in relation to microalgal community dynamics (Fig. 11.11), (2) nutrient manipulations (additions or reductions) where the community growth and compositional responses are assessed using various groups of species-specific growth/composition parameters, including microscopic observations, biochemical, molecular and automated identification and enumeration techniques (e.g., image analysis, diagnostic photopigment analyses, flow cytometry, genetic screening using probes, fluorescence in situ

![In situ nutrient addition bioassays](image)

**Figure 11.11** Results from a set of *in situ* nutrient addition bioassays conducted at three locations along the axis of the Neuse River Estuary that was routinely monitored for ambient nutrient (ammonium, nitrate, phosphate) concentrations and chlorophyll *a* as an indicator phytoplankton biomass. All nitrogen forms were added at 20 μM-N, while phosphate was added at 5 μM-P. The locations of bioassays are shown (symbols) on the map: (A) just upstream of the chlorophyll *a* maximum (C_{max}); (B) at the C_{max} and (C) downstream of the C_{max}. Strong N limitation was encountered at the C_{max} location. Downstream of the C_{max}, N limitation dominated and there was a strong preference for ammonium over nitrate as a DIN source.
hyridization, microautoradiography, microarrays), (3) a combination of the above, in which observational and experimental results are coupled to predictive modeling over a range of complexities, from species to communities to habitat and ecosystem levels.

The ability of some algal groups to utilize organic N, which often is the most plentiful new N source (Cornell et al., 1995; Peierls and Parel, 1997; Seitzinger and Sanders, 1999; see also Chapter 7 by Mulholland and Lomas, this volume), can also provide a competitive advantage (Antia et al., 1991). For example, some heterotrophic dinoflagellates, cyanobacteria, diatoms, and brown algae (e.g., the brown tide species, *Aureococcus anophagefferens*) can assimilate diverse organic compounds over a range of naturally-occurring concentrations (Antia et al., 1991; Gobler et al., 2002; Kana et al., 2004). In addition, members of these groups are capable of photoheterotrophy, which could be advantageous in illuminated surface waters where bloom species often thrive (Parel, 1991). Lastly, it is well known that bloom-forming phytoplankton exchange a variety of organic and inorganic N compounds with bacterial and other algal epiphytes in consortial relationships (Parel and Pinckney, 1996). Such exchanges can benefit the growth of both host and epiphyte, and may be particularly effective in the development and persistence of blooms in low-nutrient oligotrophic waters experiencing incipient eutrophication. Essential to these consortial associations is metabolic flexibility, particularly the ability to utilize and exchange a variety of N-containing compounds if and when they become available for supporting growth and bloom formation. This is relevant for planktonic and benthic marine cyanobacterial nuisance bloom genera (e.g., *Nodularia, Aphanizomenon, Lyngbya*), which can utilize inorganic and organic new N inputs and can also resort to N₂ fixation as an N source when external N sources are depleted. Using this “switch hitting” capability, these taxa are able to take advantage of intermittent or pulse nutrient loading events such as periods of riverine runoff following storm events, periodic upwelling and deepening of the mixed layer, atmospheric deposition (rainfall and dryfall) and groundwater loading events. This metabolic flexibility may help initiate and sustain massive blooms of these diazotrophic cyanobacteria in the eutrophied Baltic Sea, nutrient enriched coastal lagoons, and some estuaries (Parel, 1988) as well as shallow water benthic environments (tropical reefs and intertidal flats) experiencing nutrient enrichment (La Pointe et al., 1997).

Some allochthonous organic N compounds may also have chemotactic and allelopathic functions (Gallucci and Paerl, 1983; Keating, 1977; Legrand et al., 2003). In these capacities, specific compounds that are excreted by some phytoplankton taxa may attract or repel other phytoplankton or bacterial taxa, and serve as cues for establishing specific associations between consortial and symbiotic partners. In this manner, organic N compounds (e.g., amino acids, peptides) may play a central and crucial role in determining microbial community composition, including establishment and proliferation of bloom species (Suikkak and et al., 2004).

These results provide insight into the potential biogeochemical and trophic impacts of shifting compositions and quantities of new N compounds implicated in marine eutrophication. Both the amount and makeup of new N play critical roles in determining the magnitude and composition of primary producers. Structural and functional modifications of the primary producer community strongly influence
biogeochemical cycling and trophic (heterotrophs, grazers, herbivores and carnivores) responses to nutrient over-enrichment and eutrophication of estuarine, coastal and open ocean waters.

5. The Role of Nitrogen in Relation to other Nutrients

While excessive N loading has been implicated in marine eutrophication, it also plays a role in the balance, availability and ecological manifestations of other potentially-limiting nutrients, most notably P, Si and Fe (Hecky and Kilham, 1988; see also Chapter 37 by Kudela, this volume and Chapter 38 by Hutchins and Fu, this volume). N loading from anthropogenic sources has dramatically increased over the past 50 years (National Research Council, 2000), to the extent that in some receiving waters N over-enrichment has led to shortages in other nutrients required for metabolic activity and growth. Under these conditions, N may be supplied in excess of the stoichiometric nutrient ratios needed to support phytoplankton and higher plant growth (i.e., Redfield ratio, Redfield, 1958). If N over-enrichment persists for days to weeks, other nutrient limitations may, at times, dominate. Examples can be found in the Northern Gulf of Mexico and the Northern Adriatic Sea, regions receiving very high N loads discharged via the Mississippi and Po Rivers respectively. Here, P limitation, N and P co-limitation and Si limitation (of diatom growth) have been observed in the fresh and brackish water components of riverine plumes that can extend well over 100 km into the receiving waters (Lohrenz et al., 1999; Sylvan et al., 2006). In these instances chronic increases in N loading from the vast amount of agricultural land in the Mississippi and Po river basins appear to have induced P and other nutrient limitations (Ferber, 2004, Justić et al., 1995a,b). A similar scenario is evident in the Chesapeake Bay, where elevated N loading accompanying the spring maximal freshwater runoff period increases the potential for P limitation (Fisher and Gustafson, 2004) (Fig. 11.12).

These shifts in nutrient limitation can lead to alterations in phytoplankton community composition, such as a shift from diatoms towards flagellates and dinoflagellates (Justić et al., 1995b; Rabalais et al., 2001; Turner et al., 1998). Planktonic invertebrates, shellfish and finfish consumers whose diets are highly dependent on the composition and abundance of specific phytoplankton food species and groups may then be affected (Turner et al., 1998). In the Baltic and Black Sea basins, the combined effects of enhanced watershed N loading and increased Si retention due to damming of rivers has led to dramatic decreases in Si:N ratios in freshwater discharge to these seas. These changes may affect nutrient cycling and food web dynamics (Humborg et al., 2000).

Biogeochemical and trophic responses to altered nutrient loading stoichiometry are intimately linked (Conley, 2000). For example, if phytoplankton composition and biomass are altered in response to relatively high N:P and N:Si loads, the trophic fate of this new production may change. In the case of diatoms, which are readily grazed and packaged as fecal pellets by zooplankton, changes in their availability and abundance may affect the downward flux and accumulation of such nutrient-rich pellets. Furthermore, Si sedimentation via such pellets may be altered, resulting in
Figure 11.12 Seasonal variations in light, P, and N as limiting resources for phytoplankton at mesohaline Chesapeake Bay station CB4.3C during 1990–2003. Responses to nutrient addition bioassays were transformed to indices as described in Fisher et al. (1999). Values of the index >0.5 indicate the primary limiting resource. Upper panel is the average monthly Susquehanna River discharge during 1968–2001 (long term mean) and the monthly average discharge during the study period (Aug. 1990–Dec. 2003), with the 30 year mean discharge shown as the horizontal line (Figure adapted from Fisher and Gustafson, 2004).
changes in biogenic Si cycling in sediments and availability to future diatom populations. The composition of other phytoplankton groups may also be affected by anthropogenically-altered inputs and fluxes of growth-limiting nutrients. For example, taxa that are favored by increased levels of N supply relative to P and Si are likely to benefit (Dortch and Whitledge, 1992). These would include flagellates and dinoflagellates (Elmgren and Larsson., 2001; Humborg et al., 2000), and groups that include toxic and other bloom-forming nuisance species (Anderson and Garrison, 1997; Hodgkiss and Ho, 1997). Toxic, ungrazed nuisance species are likely to increase the downward flux of carbon and other nutrients, since they will be less effectively cycled in the water column. This, in turn may exacerbate hypoxia and anoxia potentials in vertically stratified waters. The ecosystem-level ramifications are multifold, including altered cycling of oxygen and nutrients, which would impact nutrient supply and availability to primary and secondary producers and trophic relations between them (including the microbial loop).

At nearshore and offshore oceanic locations, nitrogen enrichment can also affect primary production, nutrient cycling, and nutrient limitation. Nutrient addition experiments conducted on natural phytoplankton communities over a range of scales (microcosms to large-scale surface water fertilization experiments) have shown that N additions that mimic upwelling, atmospheric and riverine inputs are capable of stimulating primary production and may help initiate blooms (Paerl, 1997; Paerl et al., 1999; Zhang, 1994) (Fig. 11.13). These studies have also shown the potential for N-driven eutrophication to exist in open ocean waters. However, aside from some coastal seas such as the North Sea, Baltic, N. Adriatic, Northwest Pacific coastal regions, Sea of Japan and W. Mediterranean, the rates of N loading appear to still be too low to elicit the most obvious signs of eutrophication, such as noticeable “greening” of these oligotrophic waters and persistent algal blooms. The slow delivery of N and other potentially-limiting nutrients (e.g., Fe) to these waters most likely prevents such events from occurring. In contrast, acute N loading events, such as fertilizer spills from ships, riverine floodwaters extending over continental shelves (e.g., Gulf of Mexico, Amazon, Yangtze and Yellow Rivers), and experimental large-scale fertilization experiments (Kolber et al., 2002) are capable of eliciting significant increases in primary and potentially secondary production. Therefore, the potential for open ocean eutrophication exists and should be carefully monitored.

Fortunately, remote-sensing platforms specifically designed for detecting and quantifying phytoplankton biomass (as chlorophyll and other photopigments) are routinely collecting data over a range of temporal and spatial scales. In particular, NASA’s satellite-based SeaWiFS ocean color sensing system can examine long-term, large-scale trends in estuarine, near-shore and oceanic primary production and producer biomass (http://seawifs.gsfc.nasa.gov/SEAWIFS.html) (Harding and Perry, 1997). This system, as well as aircraft and satellite-based multispectral (e.g., MERIS, MODIS) and hyperspectral (e.g., AVIRIS) remote sensing platforms offer an excellent opportunity to document coastal and open ocean phytoplankton (and higher plant) production baselines, against which change can be gauged over appropriate spatial and temporal scales.

An interesting result of open ocean nutrient limitation/enrichment research is the discovery that primary production in certain oligotrophic ocean gyres (Northern and
Central Pacific, Southern Ocean) can be Fe rather than N or P limited (Behrenfeld and Kolber, 1999; Martin et al., 1994). These regions have dissolved inorganic N and P concentrations in excess of phytoplankton requirements, indicating that N enrichment is not likely to impact eutrophication potentials of these waters and that any eutrophication effect of N enrichment is reliant on parallel new Fe inputs (Kolber et al., 2002).

**Figure 11.13** Effects of the addition of dissolved inorganic N (NH$_4^+$ and NO$_3^-$, 5 μM each), N-containing rainfall (collected at Morehead City, NC), iron (as EDTA-chelated and non-chelated FeCl$_3$; 0.2 μM each; EDTA, and the combination of NO$_3^-$ and EDTA-chelated FeCl$_3$) on coastal Atlantic Ocean phytoplankton communities. *Indicates treatments were significantly ($p < 0.05$) from controls, using Bonferroni *a posteriori* comparisons of the means. See Paerl et al. (1999) for details of the experimental procedures.
New Fe inputs to these regions are largely atmospheric, originating from continental
dust, volcanic sources, fossil fuel combustion and industrial sources (Duce et al., 1996;
GESAMP, 1989). For subtropical N. Atlantic oceanic waters, African Saharan dust
storms appear to be a major Fe input source supporting algal bloom formation,
including red tides (Walsh and Steidinger, 2001). New Fe inputs from this and other
eaolian sources may in part control the assimilation of externally-supplied N as well
internally-supplied N via N₂ fixation and upwelled regenerated N. A similar scenario
has been proposed for a region of the Indian Ocean (Takeda et al., 1995).

In more coastal/shelf regions, such as the W. Atlantic Gulf Stream off the US East
Coast and off western Florida, the potential for iron limitation has also been demon-
strated (Paerl et al., 1999; Walsh and Steidinger, 2001). However, in these regions, Fe
and N limitations appear to be closely linked and interdependent. For example, while
dissolved inorganic nitrogen (DIN) additions, as either ammonium or nitrate, stimu-
lated primary production and phytoplankton productivity (as CO₂ fixation) and
biomass (as chlorophyll a) in oligotrophic Gulf Stream and Sargasso Sea waters off
the North Carolina coast, per amount of N, ammonium often proved to be more
stimulatory than nitrate (Paerl et al., 1999) (Fig. 11.13). This differential response was
attributed to the Fe required by phytoplankton to reduce nitrate (nitrate reductase)
prior to its use as an N source to support growth. This requirement does not exist for
ammonium assimilation. These results suggest that, in the case of nitrate enrichment,
there may be a parallel Fe requirement, indicating that for eutrophication to proceed
additions both nutrients are required. This synergistic enrichment may be provided by
certain anthropogenic pollution sources, such as industrial and urban atmospheric
emissions or riverine inputs enriched in both N and Fe (Church et al., 1984; Duce
et al., 1986). Thus, the connection between human activities and marine eutrophica-
tion may need to include consideration of simultaneous multiple nutrient enrichment
processes, which are influenced by agricultural, urban, and industrial activities.

6. Human Activities in the Coastal Zone
and Eutrophication

Human activities in and modifications of coastal watersheds significantly affect
estuarine and coastal N-driven eutrophication (Boesch et al., 2001a,b; Vitousek et al.,
1997; see also Chapter 36 by Boyer and Howarth, this volume and Chapter 9 by
Seitzinger and Harrison, this volume). Most prominent are coastal urbanization,
human wastewater discharges, the destruction and modification of coastal wetlands,
and large-scale agricultural and aquacultural operations (Fig. 11.1). As population
density continues to grow in the coastal zone, these activities and associated pollution
are becoming increasingly significant sources of N and other nutrients to coastal waters.

6.1. Human development of the coastal landscape

With increases in coastal population, and the accompanying shifts in land use
(Vitousek and Mooney, 1997), come several changes that have significant impacts
on the delivery of nitrogen to coastal waters. Sources of N change, and generally
increase, as human use of a coastal region intensifies. Wastewater, agricultural
discharge, stormwater and industrial sources of N are among the major contributors.
Another change that accompanies human development of the coastal landscape is an
alteration of the transport mechanisms of N to coastal waters. Imperviousness,
ditching, channelization of streams and rivers and removal of native vegetation all
contribute to increased rates and quantities of N transport from the land to the
adjacent water bodies. Finally, human activities in the coastal region often lead to
significant loss of the natural landscapes that either retain or remove N. Wetland loss
and watershed-scale landscape alterations are well documented, and contribute to the
increased transport of N from human affected coastal regions. There is a cumulative
impact on N transport to coastal waters from human development of the coastal
landscape that includes the effects of increased sources, modified transport and altered
biological processing of N (Fig. 11.1).
Sources of N that increase with human activity include sewage treatment plant
discharges, fertilizers associated with agriculture and other activities (e.g., lawns,
gardens, golf courses), increased atmospheric N deposition and increased stormwater
carrying myriad non-point source pollutants associated with urbanization (Boesch
et al., 2001a; Paerl, 1997). In the first stages of human development of coastal
regions, the shifts in land use are generally from forest to agriculture and residential.
Once an area is populated, the shifts in land use are then away from open space
(forest and agriculture) toward residential and industrial (Beach, 2002). Through this
sequence of land use alteration the N sources change and the trend is generally
towards more and more N transported to coastal waters (Line et al., 2002).
Human modifications in the transport of water from land to coastal waters occur
throughout coastal watersheds (Schueler and Holland, 2000). If sources of N are
present, transport of water from the landscape to the aquatic system usually translates
into enhanced N mobilization and loading. Urbanization leads to increased imperv-
iousness through the presence of roads and rooftops, and imperviousness leads to
accelerated transport of water from the landscape. However, imperviousness alone is
not responsible for the changes in transport of stormwater and entrained pollutants
(Beach, 2002). Poorly planned urban areas often exacerbate the problems associated
with imperviousness through sprawling development. Development characterized
by sprawl tends to have more and wider roads that have the added effect of more
driving, and thus larger sources of N and other potential pollutants (Trobulak and
Frissell, 2000). Improving the quality of planning during development can signifi-
cantly reduce the negative impacts on water quality. A three phased approach that
includes strategies to reduce the area covered by streets and parking lots, improved
site design and conserve natural areas is required to address the impacts of urbaniza-
tion on stormwater transport to adjacent waters (Schueler and Holland, 2000).
The retention and removal (i.e., attenuation) of N during transport through the
watershed can be significantly affected by human activities. Ditching, channelization
and increased imperviousness all facilitate movement of water, and thus N, to coastal
waters. Reduced residence time often equates to reduced biological attenuation of
the N load from a watershed. Larger quantities of faster moving waters are less likely
to be effectively filtered in small headwater streams, wetlands and riparian areas.
Human activities also lead to direct negative effects on natural systems that attenuate
the N load from the landscape. Destruction and degradation of headwater streams
that often occurs in areas with intensive human use is likely to significantly reduce N removal and retention (Peterson et al., 2001). In the US, wetland loss is on the order of 50% since the pre-settlement era and despite legal protections, this loss is likely to continue to increase (Mitsch and Gosselink, 2000). Even with laws in place to protect wetland loss, management of wetland resources presents a significant challenge (La Peyre et al., 2001).

Wetlands are known sinks for N transported to them from watersheds (Reddy and Gale, 1994). The loss of wetland transformations of N (as opposed to removal of N) are also potential contributors to water quality by transforming N from highly biologically available inorganic forms to possibly less labile organic forms (Craft et al., 1989). Riparian areas are also sites of significant biological removal of N via denitrification (Jacobs and Gilliam, 1985; Spruill, 2004). Their function is frequently either lost or significantly compromised by human activities, diminishing the ability of the landscape to remove N (Groffman et al., 2003). Urbanization is common enough in coastal regions that nitrogen cycling research in the urban landscapes is required to understand the transport and transformation of N in these areas (Groffman et al., 2002)

6.2. Agricultural activities

World-wide, coastal regions are experiencing unprecedented agricultural growth, in the form of conversion of forests, swamps and wetlands to row-crop farmland and the conversion of small family farms to intensive livestock (poultry, hogs, cattle) operations (see also Chapter 9 by Seitzinger and Harrison, this volume and Chapter 36 by Boyer and Howarth, this volume). The rapid rise in commercial fertilizer use accompanying agricultural expansion in coastal watersheds has accelerated N loading from these lands. This trend has been evident in developed regions of Europe, North America, and Australia for at least half a century (i.e., post World War II; Chambers et al., 2001; Howarth, 1998; Nixon, 1995; Vitousek et al., 1997). However, currently the most rapid rise in N loading from agricultural lands is occurring in developing nations in Asia, Africa, South and Central America. In addition, countries that have had traditional agricultural sectors (e.g., China, India) are using ever-increasing amounts of synthetic N fertilizers to bolster production. Not surprisingly, these countries are experiencing some of the largest increases in N discharge to N limited estuarine and coastal waters (Vitousek et al., 1997). There is substantial evidence that symptoms of advanced eutrophication are proliferating in these waters, including a troublesome rise in harmful (toxic, hypoxia-inducing) algal bloom events (Hallegraeff, 1993; Richardson, 1997), hypoxia and anoxia (Rabalais, 2002).

In many watersheds along the US East and Gulf Coasts, as well as throughout Canada, agriculture is responsible for at least half the estuarine and coastal new N (and P) inputs (Table 11.1, Fig. 11.14) (Castro et al., 2003; Driscoll et al., 2003). This percentage is growing in regions that have yet to experience forest and wetland conversion to intensive agriculture. Clearly, this sector of human activity is a prime target for N input reductions. A variety of best management strategies aimed at reducing N losses from agricultural operations is either in place or being formulated for coastal watersheds in North America and Europe. In most mandated N reduction
Figure 11.14 Relative contributions of atmospheric deposition, human sewage, forests, urban non-point sources and agriculture to “new” N loading in a set of representative US Atlantic (East Coast) and Gulf of Mexico estuaries. Figure adapted from Castro et al. (2003).
strategies that have been formulated for watersheds where agriculture constitutes a significant fraction of new N input, a 30–50% reduction of N discharge from agricultural lands can be expected. There are numerous efforts in place to meet such mandated reductions. They include; construction of riparian vegetative buffers, no-till farming practices, restrictions on timing and amount of N fertilizer application, use of organic fertilizer and the use of natural sources of new N, such as soybeans and other leguminous plants that are rotated with N demanding crops such as cereals (corn, wheat, rice), fiber crops (cotton, flax) and vegetables.

While nitrate is a dominant new N source from agriculture, ammonium and organic N can also be significant waste products, especially from animal operations where reduced N tends to accumulate in waste stored in “lagoons” or sprayed as fertilizer on surrounding fields. Discharge from agriculture is dominated by surface runoff (Boesch et al., 2001a,b). Subsurface and atmospheric N inputs can also be large from intensive animal operations and feedlots (Paerl, 1997; Walker et al., 2004; Whitall and Paerl, 2001). Each of these sources is significant enough to warrant their inclusion in basin-wide and regional N reductions aimed at improving water quality (e.g., EPA’s Total Maximum Daily Loads or TMDLs; US EPA, 1999b).

6.3. Aquaculture/mariculture

As a means of sustaining coastal fisheries-based economies and as a source of animal-based protein, coastal aquaculture and mariculture (jointly termed aquaculture) have grown rapidly. In part, the need for and success of aquaculture can be attributed to the depletion of traditional estuarine and coastal fisheries in many regions of the world, including Asia, Europe, North and South America. Aquaculture has also proven more profitable and attractive than many sustenance fisheries because highly-desired, high-income specialty fisheries are often amenable to aquaculture. These include shellfish (oysters, clams, mussels), crustacean (shrimp, lobster, crab, crayfish), and finfish (salmon, trout, catfish, striped bass, tilapia) operations that are expanding worldwide. A vast majority of these operations depend on fertilization and/or input of food supplies to sustain productivity. As with intensive agricultural operations, ensuring adequate N supply is critical to maximizing aquacultural production. Invariably, this translates into administering high doses of fixed N, of which a significant fraction, ranging from a few percent to well over 50% is released to the surrounding waters (Leung et al., 1999; Tovar et al., 2000; Wu, 1995). While overall, aquaculture represents a much smaller N input source than agriculture, from an N retention perspective, aquaculture operations can constitute localized intensive sources of N fertilization and potential drivers of eutrophication in nearby N-limited waters. The unintentional eutrophicating effects of aquaculture have already become evident in coastal net-pen fish (salmon in particular) operations in semi-enclosed (i.e., poorly-flushed) lagoons and embayments (fjords) in Northern Europe (Denmark, Norway, Sweden, Scotland), North and South America (Western Canada, New England and the Canadian Maritime provinces, Colombia, Brazil and Chile). In particular, the Scandinavian fjords have experienced symptoms of advanced eutrophication in response to excessive fertilization associated with these as well as agricultural operations (Christensen et al., 2000). Additionally, the
eutrophication related impacts that result from these operations are complicated by
the use of antibiotics in feed formulations. Antibiotic introduction could affect
microbial antibiotic resistance, and hence alter microbial community composition
and activities in response to nutrient enrichment. In Asia and South America,
brackish to full-salinity coastal lagoonal ecosystems are being used to produce shrimp,
shellfish and a variety of finfish species. These systems represent concentrated nutrient
sources adjacent to low-nutrient-adapted estuarine and coastal habitats, including
mangroves, mudflats, marshes, coral reefs, and oligotrophic near-shore surface waters
(Wu, 1995). Although little work has been done to detect and characterize ecological
responses to the nutrient enriched effluent from these operations, previous exper-
imental work indicates that both increases in benthic and planktonic microalgal
production, epiphytization of seagrasses and corals, shifts in plant in producer and
grazer community composition are among the responses accompanying eutrophica-
tion of receiving waters (D’Elia et al., 1981; Johannes, 1975; LaPointe et al., 1997;
Laws and Redalje, 1979). The global expansion of aquaculture represents a potential
threat to both water quality and habitat integrity of ambient waters, many of which
support flora and fauna adapted to oligotrophic conditions.

7. The Future and Nitrogen Management

It is clear that N over-enrichment due to human population growth in coastal
water- and air-sheds will remain a key driver of marine eutrophication (Peierls et al.,
1991; Vitousek et al., 1997) well into the foreseeable future, especially in the rapidly-
developing regions of Asia, Africa, and South America. The growing N “glut” that
accompanies anthropogenic encroachment on the coastal zone will continue to enhance
primary production and accelerate eutrophication. Because few receiving waters show
any indication of becoming N saturated, the potential exists for coastal and oceanic
waters to exhibit further N-enhanced eutrophication. While some upstream segments of
estuarine ecosystems that have experienced a lengthy history of excessive N enrichment
at times experience N saturation and show spring N & P co-limitation or even P or Si
limitation (Fisher and Gustafson, 2004, Fisher et al., 1999; Justic et al., 1995a,b; Sylvan
et al., 2006), they continue to show an overall high degree of sensitivity to N enrichment.
This is especially true for the broader downstream mesohaline and euhaline portions of
estuaries, where the combined effects of denitrification and relatively abundant and
readily recycled marine P and Si supplies preclude limitation by these elements.

Therefore, preservation of acceptable water quality and reversal of eutrophication
in these systems will invariably include reductions of current loads and setting limits
on new N input, either alone or in combination with other nutrient (e.g., P) input
reductions. Freshwater nutrient management has, in many instances, successfully
arrested and reversed eutrophication by reducing P loadings (Edmondson, 1970;
Likens, 1972; Vollenweider, 1982). Indeed, the upstream freshwater portions of
many estuaries are also P-limited (Boynton and Kemp, 2000; Fisher et al., 1999),
and P input constraints in these waters have been quite successful in reducing the
unwanted symptoms of eutrophication (nuisance algal blooms, toxicity, hypoxia,
As estuaries drain to the seas, both salinity and N limitation increase (Ryther and Dunstan, 1971; Nixon, 1986, 1995). However, reducing N alone can create favorable conditions for N\textsubscript{2} fixing phytoplankton and present the possibility of the return of N to the system through N\textsubscript{2} fixation (Horne, 1977; Piehler et al., 2002). Therefore, along the freshwater–estuarine–coastal transitional zone, both N and P input constraints are likely to be needed if we are to control and manage marine eutrophication (Cloern, 2001; Conley, 2000; Paerl et al., 2004, Sylvan et al., 2006). This calls for addressing nutrient management along the integrative watershed-to coastal ocean (and beyond) scale; a scale that is no doubt challenging, but essential if we are to consider effective, long-term nutrient management aimed at preserving the entire suite of biotic resources involved in and affected by marine eutrophication.

Coastal waters are sensitive to both human and natural perturbations. For example, accelerating nutrient, sediment, and toxics inputs, may be accompanied by climatic, geochemical (i.e., volcanism, subsidence) and other forms of natural change. There is evidence that certain manifestations of climate change, including tropical storm and hurricane frequency, may also be increasing (Goldenberg et al., 2001, Webster et al., 2005). It is therefore useful to develop management approaches and indicators of ecosystem response that could help distinguish human from natural perturbations.

Determining and evaluating the ecological response of marine ecosystems to N enrichment and interacting physical–chemical perturbations is highly dependent on the indicators used and sampling sites selected. Benthic indicators may tell quite a different story from planktonic ones in ephemerally-stratified system (e.g., Chesapeake Bay, Pamlico Sound, Mobile Bay, Florida Bay). In well-flushed (short water residence time) systems, phytoplankton growth responses to nutrient enrichment may not be nearly as profound as those for benthic microalgae. Here, benthic microalgae and associated infauna may be more sensitive and meaningful indicators of ecosystem response to nutrient enrichment. In addition, indicators of community structure (i.e., diversity indices, keystone species) may gauge ecosystem conditions quite differently than indicators of function (primary and secondary production, respiration, nutrient cycling). Indices of biotic integrity (IBI) and habitat suitability indicators (HBI) are specific examples of indicators that in combination can assess structure, physical–chemical quality and biological measures.

To link N and other nutrient inputs to eutrophication dynamics across diverse coastal and oceanic systems, specific, broadly-applicable and integrative indicators that can couple biotic community structure to function in the context of ecological condition and change are needed (cf. Bartone, 2005; Committee on Environment and Natural Resources, 1997). Using case studies representing a range of impacted habitats, we need to develop and assess indicators of water quality, food web and habitat condition in response to a range of physical, chemical and biotic perturbations in marine ecosystems varying in water residence time, climatic regime and trophic state. Emphasis should be placed on microbial, plant and animal taxa and assemblages capable of providing qualitative and quantitative responses to nutrient and other stressors. These indicators should have sufficient detail to help clarify mechanisms underlying ecological change in response to external perturbations, with the ultimate goal of identifying indicators that will help researchers and...
managers quantify roles of key organisms mediating production, energy flow, nutrient and oxygen cycling aspects of eutrophication. Combining these specific indicators with modeling efforts will help clarify and distinguish anthropogenic from natural stressors in diverse marine environments.

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REFERENCES


Redfield, A. C. (1958). The biological control of chemical factors in the environment. Am. Scientist 46, 205–222.


