Unlike toxic chemicals such as pesticides that are intended as biocides (to cause biological harm), nitrogen pollution in coastal bays and estuaries is primarily a consequence of nitrogen fertilization to increase agricultural production and fossil-fuel combustion associated with a variety of human activities. It is a bit counterintuitive to think that the Earth, with an atmosphere containing \(3.96 \times 10^{15}\) t of nitrogen, would be deficient in nitrogen. This is because nitrogen in the atmosphere exists predominantly as biologically inert dinitrogen molecules. Biological nitrogen fixation, mediated by bacteria, has remained the mainstay of agricultural production throughout human history. However, the natural processes of nitrogen fixation and mineral sources of usable nitrogen cannot sustain food requirements for human population growth, which in the 20th century rose from 1.65 billion to more than 6 billion. The Haber-Bosch process, developed in 1913, allowed for industrial-scale production of nitrogen-based fertilizers and now greatly supplements the fertilizer required to meet human demands. In recent years, it has accounted for more than 90 million t of nitrogen fertilizers used worldwide (2004–2005 data), of which 11 million t were used in the United States, 27 million t in China, and 11 million t in India.¹

Only about 14% of nitrogen used as fertilizers results in crops and an even lesser amount in human food.² The remaining amount is lost during food production, including transportation and application of fertilizers, seepage to groundwater and surface water streams, spoilage and waste, crop residue, animal wastes, and volatilization to the atmosphere. The direct and indirect delivery of large

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amounts of reactive nitrogen into coastal bays and estuaries has become a matter of global concern, largely due to widespread eutrophication and its adverse consequences on the structure and functioning of coastal ecosystems.3,4

**SOURCES OF NITROGEN**

Source attribution of nitrogen input to coastal waters and knowledge of its transport and sequestration in different environmental compartments are important requisites for establishing pollution abatement or mitigation strategies. Fertilizer applications; municipal sewage and certain industrial effluent discharges; leaking sewer and septic systems; aquaculture operations; fossil-fuel combustion (i.e., power plants and automobiles); runoff from farms, forests, and urban areas; atmospheric deposition of emissions from farms and fields; and inflow from adjacent coastal waters all contribute measurable amounts of reactive nitrogen to coastal bays and estuaries. The relative contributions of different nitrogen sources to a number of estuaries on the U.S. East Coast are summarized in Figure 1.

There is a considerable dearth of data on direct atmospheric deposition of nitrogen to coastal bays and estuaries, since all nitrogen-related atmospheric monitoring programs are land-based. In comparison, data on atmospheric deposition of nitrogen and its mass balance in coastal watersheds are considerable. For the forests in the Northeast, for example, it has been noted that more than 90% of nitrogen deposition would be retained in the watershed if the deposition rate was less than 7 kilograms per hectare per year (kg/ha/yr), which is among the highest values of inorganic nitrogen wet deposition calculated for the United States for 2005 (see Figure 2).5 According to this scenario, transport of nitrogen from the watershed to the estuary would be minimal. Atmospheric nitrogen deposition rates in the Midwest and Far West are much lower, predominantly less than 2 kg/ha/yr, and are not viewed as a significant source of impact on estuarine ecosystems.

The data noted above are for "wet deposition"; dry deposition (not associated with precipitation) can add substantially to total nitrogen inputs.3,6 Estimates of atmospheric deposition of nitrogen to coastal watersheds, based on the sum of wet and dry deposition, range from 3 kg/ha/yr (low values mostly in the West) to
14 kg/ha/yr (high values in Delaware Bay, Chesapeake Bay, and some estuaries in the Gulf of Mexico). However, dry deposition monitoring data are few and deposition rates are often derived from modeling estimates; it is estimated that on average 40% of the total nitrogen deposition to the watershed is dry deposition.\(^7\) There are even fewer monitoring data and modeling estimates of direct atmospheric deposition of nitrogen to the estuaries, but it is assumed to be lower than direct deposition on the watersheds, in part, because dry deposition is expected to be lower over water.

In comparison with atmospheric deposition of nitrogen, agricultural fertilizer application adds considerably to surface water runoff and groundwater flows. Fertilizer application is dependent on soil conditions, as well as the targeted crops, and varies considerably. For example, 298 kg/ha for wheat, 320 kg/ha for corn, and 434 kg/ha for potatoes in intensely managed farms in the Columbia Basin, OR.\(^8\) More current data on fertilizer nitrogen use in the United States can be accessed from the U.S. Department of Agriculture (www.ers.usda.gov/datasets/fertilizeruse). Additional fertilizer nitrogen sources include losses from turf grass and home gardens for which data are generally not available.

A portion of the unutilized nitrogen in fertilizers seeps into shallow groundwater and aquifers near farms. Although nitrogen concentration in pristine groundwater should be negligible, levels less than 8 µg-at N/L are considered typical of shallow groundwater unaffected by direct human activities.\(^3\) Nitrogen concentrations in groundwater vary considerably, typically ranging from nondetectable to 400 µg-at N/L, with isolated observations being an order of magnitude higher.\(^10\)\(^11\) Soil and hydrological characteristics that would impact nitrogen flow and seepage at smaller spatial scales, include fractions of artificially drained soils, irrigation intensity, as well as denitrification of nitrate by microbes in soil and vegetative cover. In addition, regional physiography plays an important role in determining the relative amount of nitrogen delivered to an estuary via groundwater. As an example, base flow nitrate load accounts for approximately 26–100% of total nitrate load in Chesapeake Bay.\(^12\)

As part of a major study to assess environmental and socioeconomic factors influencing the size and persistence of the hypoxic zone in the northern Gulf of Mexico, it was noted that large amounts of nitrogen fertilizers in the Mississippi River basin that are not used up by plants or fixed in the soil can leach into groundwater or run off into surface waters during rainfall events. Major contributions are from fertilizers and mineralized soil nitrogen (58%), animal manure (16%), municipal and industrial point sources (9%), and other sources, including atmospheric deposition, groundwater, soil erosion, and urban runoff (16%).\(^13\)

### Nitrogen Concentrations

A general range of observed concentrations of nitrogen and nitrogenous compounds in coastal waters and estuaries is given in Table 1. While dissolved nitrogen gas is the most abundant form, it is of little consequence in terms of biological productivity; its concentration is determined by its solubility constant, temperature, salinity, and pressure.

<table>
<thead>
<tr>
<th>Nitrogen Type</th>
<th>Concentration (µg-at/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nitrogen (gaseous)</td>
<td>900 – 1100</td>
</tr>
<tr>
<td>Nitrous oxide (gaseous)</td>
<td>0 – 0.25</td>
</tr>
<tr>
<td>Nitrate (dissolved)</td>
<td>0 – 200</td>
</tr>
<tr>
<td>Nitrite (dissolved)</td>
<td>0.03 – 10</td>
</tr>
<tr>
<td>Ammonium (dissolved)</td>
<td>0.03 – 100</td>
</tr>
<tr>
<td>Dissolved organic N</td>
<td>3 – 20</td>
</tr>
<tr>
<td>Particulate organic N</td>
<td>0.1 – 30</td>
</tr>
</tbody>
</table>

Notes: Nitrogen concentrations are often given as the amount of nitrate (NO\(_3\)-N), amount of nitrogen present in nitrate (NO\(_3\)-N), or gram-atOMIC weight of nitrogen (quantity of element equivalent to its atomic weight in grams). Using current standardized units, 1 mg NO\(_3\)-N/L is equivalent to 0.23 mg NO\(_3\)-N/L, which, in turn, is equal to ca. 16 µg at NO\(_3\)-N/L.

However, concentrations of different reactive nitrogen species vary considerably as a result of biochemical processes, chemical transformations, and mixing of water layers with different concentrations.

Ambient concentrations of reactive nitrogen in near-surface coastal waters are generally less than 2 µg-at/L, while in estuaries that receive large amounts of municipal sewage, agricultural runoff, industrial effluents, and groundwater discharge, nitrogen concentrations typically approach or exceed 200 µg-at/L, with even higher concentration near the sources.

There are no quantitative criteria for nitrogen compounds in U.S. coastal waters and estuaries to assess levels of nutrient enrichment above which impairment of the ecosystem function, resources, and overall condition could occur. Quite often, categories of good, fair, and poor are derived from quartiles or percentiles of accumulated data, or based on individual opinions. Table 2 provides thresholds for dissolved inorganic nitrogen that were used to characterize the condition of estuaries under the U.S. Environmental Protection Agency’s (EPA) National Estuary Program. This assessment was based on analysis of filtered surface water samples collected in late summer.\(^17\)

### Table 1. Nitrogen concentration in coastal waters.\(^4\)\(^-\)\(^6\)

### Table 2. Thresholds of dissolved inorganic nitrogen concentrations [µg-at N/L] used in EPA’s National Estuary Program.
EFFETS OF NITROGEN OVER-ENRICHMENT

Linking Nitrogen Over-Enrichment to Eutrophication

Both nitrate and ammonium are readily utilized during photosynthetic production of organic matter by phytoplankton, macroalgae, and submerged aquatic vegetation. Utilization of ammonium is energetically more advantageous, especially under light-limited conditions. Excessive ammonium concentrations can also reduce nitrate uptake. Other forms of nitrogen, including organic nitrogen, can become available for plant growth, either through direct uptake or through use of inorganic forms of nitrogen resulting from breakdown of organic nitrogenous compounds.

It is widely assumed that increased supply of nitrogen enhances primary productivity of coastal bays and estuaries because nitrogen is often the limiting nutrient for algal growth. To some extent, the underlying linearity in such a relationship is based on a simple “lake model” developed in the early 1960s, a correlation between phosphorus inputs and eutrophication in lakes worldwide. Such a model is not reflective of the considerably more complex relationships between the highly varied amounts and forms of nitrogen and primary productivity in estuaries and coastal bays. Analysis of data from 51 estuaries showed that only 36% of the variance in phytoplankton could be explained by nitrogen loading, and underscored the difference in response to nutrient enhancement between lakes and estuaries, as well as complexity of such a relationship in different estuaries and coastal bays. The study also showed considerable scatter in the data and nonlinearity in the relationship between annual nitrogen loading and phytoplankton production with many outliers. For example, less than 5 mol of nitrogen loading per square meter was associated with phytoplankton primary production, ranging from less than 1 to 600 g C/m²/yr. This observation does not imply lack of a cause-effect relationship between a micronutrient and primary productivity, but underscores the complex interactions of several factors that control the timing and levels primary productivity, including light availability, tidal mixing, flushing, sediment resuspension, nutrient regeneration, and grazing.

To complicate matters further, different forms of nitrogen (e.g., ammonium and nitrate ions) affect primary productivity and accumulation of phytoplankton biomass differently. Estuaries that receive large amounts of wastewater discharge and little freshwater inflow result in large concentrations of ammonium, which is often taken up preferentially by phytoplankton during photosynthesis and, as such, can out-compete nitrate as a nitrogen source, especially when light availability is restricted. This has been shown in turbid parts of San Francisco Bay where nitrate uptake occurs only rarely (i.e., when ammonium concentration is reduced due to dilution or other factors to approximately 4 µg-at/L or less). Thresholds for ammonium impacts on primary productivity have not been established.

It has become evident that each estuary represents a mosaic of habitats and responds to nitrogen loading differently. For example, Narragansett Bay, RI, has distinct zones extending from riverine habitats off Providence to the seaward opening of the Rhode Island Sound bay. Within the bay, nutrient-defined enrichment, depuration, and advection zones have been defined that have different properties in terms of light limitation, silicon deficiency, nitrogen sources and influx, and dissolved oxygen concentration. Responding to ambient conditions, the bay also exhibits strong gradients in primary productivity, with the highest values in the Providence River estuary.

The establishment of nonindigenous grazers during the past two decades has had a profound effect on levels of phytoplankton biomass in the water column, which, in some areas, has reduced primary productivity or shifted primary productivity to benthic macroalgae due to increased light availability and access to underutilized nutrient pool. A five-fold decrease in the primary production and disappearance of phytoplankton bloom in Suisun Bay (northern part of San Francisco Bay), following the introduction of the Asian clam, and nuisance growth of a filamentous green alga in Lake Michigan are respective examples. Furthermore, climate fluctuations and changes, such as the North Atlantic Oscillation or periodic increase in hurricanes, strongly impact phytoplankton biomass, composition, and productivity.

Linking Nitrogen Over-Enrichment to Harmful Algal Blooms

Exceptionally high algal growth in coastal waters and estuaries sometimes results in massive accumulation or “blooms” of species that may be directly harmful to humans, fish, and wildlife or cause substantial ecosystem perturbations. Estimates of the economic impact of such blooms average US$75 million annually, including public health costs, loss of commercial fishing and aquaculture, recreation, and tourism. Largely as a result of increased input of reactive forms of nitrogen, notably ammonium and nitrate, and increased number of observations of harmful algal blooms (HABs) in recent years, nitrogen-related issues are stated or implied to include HABs. However, a direct relationship between nitrogen over-enrichment of coastal waters—mostly reported as concentration of dissolved inorganic nitrogen in water—and incidence or duration of HABs has been difficult to quantify. Several estuaries having exceptionally high nitrogen concentrations in the water (e.g., Delaware Bay) have no history of HAB occurrence. In such cases, factors other than nitrogen availability, such as vertical mixing, turbidity, flushing characteristics, and grazing control algal growth and bloom outbreaks.

In addition to the well-documented role of dissolved inorganic nitrogen in promoting algal growth and bloom dynamics, dissolved organic nitrogen may play an important role. This could be as regenerated nitrogen and as nitrogenous excretion sources, including urea and amino acids. Only recently has the role of organic nitrogen in promoting harmful algal blooms been documented in detail. Further, several pelagic and benthic dinoflagellates that are associated with HABs are recognized as “mixotrophs,” meaning that they have a mixed nutritional strategy that includes utilization of organic molecules. Heterotrophic dinoflagellates are an example of enzyme-mediated epidermal feeding.
Recent insights into effective utilization of organic molecules by several HAB species are significant since coastal runoff and effluents from urbanized and industrially developed areas typically contain large amounts of organic nitrogen in dissolved and particulate forms and this fraction is generally not accounted for in “nutrient monitoring” in most coastal waterbodies.

Impacts on Coastal and Inland Bays

Due to their small size, low freshwater runoff, and restricted circulation, coastal and inland bays can be particularly vulnerable to impacts of wastewater streams, sewage flows, and toxic chemicals from rapidly growing human populations on the coast, increasing recreational uses, and, in some instances, intense agricultural operations. Examples on the northeastern coast of the United States include Great South Bay (NY), Barnegat Bay (NJ), Rehoboth Bay (DE), Newport Bay (MD), and southern Chincoteague Bay (VA). They often show multiple signs of habitat impairment, including high chlorophyll levels, occurrence of harmful and nuisance algal blooms, depleted dissolved oxygen, loss of submerged aquatic vegetation, and impaired biotic communities and harvestable fisheries. Progressive eutrophication, fairly well documented in such bays, can eventually lead to permanent loss of essential habitats, diminished life support, and a marked decline in human use of the coastal resources and amenities.36

Several coastal bays with long-term data have demonstrated probable impacts of nitrogen enrichment by a shift in dominant primary producers, from the slower growing seagrasses to faster growing macroalgae (seaweeds) and even faster growing phytoplankton. Further, there is increasing evidence of the prevalence of brown tide, caused by a cyst forming alga Aureococcus anophagefferens that may have been introduced in the region via ballast water in the 1980s. Such blooms, though not toxic, are often quite dense and can cause discolored water, reduced seagrass growth due to decreased light penetration, and recruitment failures of commercially important shellfish.37 Occasionally, high concentrations of raphidophytes (microalgae) have been noted in Delaware’s Inland Bays, situated on the Atlantic Coast. That may be a consequence of selective grazing by zooplankton along with other environmental co-factors that may converge to offer a competitive advantage to the microalgae.38

It is important to note that moderate or highly eutrophic conditions may occur in certain bays, despite low levels of dissolved inorganic nitrogen (DIN) in the water column. For example, in the Barnegat Bay-Little Egg Harbor Estuary in New Jersey, nitrate-plus-nitrite concentrations are typically < 4 µg-at N/L and the highest ammonium concentration are usually < 2.5 µg-at N/L.36 Despite these low DIN concentrations, this estuary is plagued by high expressions of eutrophic symptoms, most notably negative biotic responses. Low DIN levels in the system largely reflect rapid uptake by algae and vascular plants, rather than steady-state conditions. Therefore, biotic processing of DIN, notably plant assimilation and sequestration, as well as the role of considerable, but largely unmonitored, flux of organic matter must be accounted for when assessing eutrophic conditions in coastal and inland bays.

Impacts on Human Health

Extensive data exist concerning the adverse effects of nitrate on human health, most notably the “blue baby” syndrome in infants. There is also concern about other health hazards (e.g., cancer, thyroid disease, diabetes) that could be associated with chronic exposure to high levels of nitrate. However, these data relate to drinking water contamination, and have no relevance to nitrate levels in estuaries and coastal bays.

Oxides of nitrogen that form in fertilized farms and fields, automobile and aircraft exhausts, emissions from factories and power plants, and burning of biomass are important contributors to the formation of smog and ozone, which, in turn, can lead to a variety of respiratory and cardiovascular ailments. Nitrogen emissions also contribute to the formation of fine particulate matter that exacerbates respiratory symptoms associated with air pollution. On occasions, high levels of nitrogen (inorganic or organic) may be an important co-factor for the onset, severity, and duration of certain harmful algal blooms, which, in turn, may pose human health risk through inhalation of toxic aerosols or dermal exposure to toxins or ingestion of tainted seafood.

Developing water quality criteria for nitrogen has remained a daunting task.

NITROGEN CONTROL STRATEGIES

Regulatory Control and Abatement

Historically, federal environmental laws and regulations for coastal waterbodies and estuaries have focused on point-source discharges of toxic chemicals and water-related habitat impairment issues. The Clean Water Act Amendments of 1977 and provisions of the Great Lakes Critical Program Act (1990) required development of standards, criteria, and pollution abatement strategies to address critical water pollution issues in the United States, including nitrogen and phosphorus over-enrichment. Criteria guidance is often expressed as pollutant concentration that will assure attainment of designated uses of the waterbody (e.g., drinking water supply), propagation of fish and wildlife, and for recreational, industrial, agricultural, and navigational purposes. Criteria can be numeric (e.g., 1 mg/L of nitrate-N) or narrative (e.g., free from conditions injurious to human health). However, the Clean Water Act stipulates protection for “existing” uses of the waterbody (i.e., those that were attained as of November 28, 1977; the date when the original water quality standards took effect). In other words, states may not remove designated uses if they are existing uses, and are expected to assure support for such uses through water quality criteria and other antidegradation measures.
It has been well recognized in the scientific community that observations and inferences from eutrophication in lakes had limited applicability to coastal bays and estuaries. As a follow-up, EPA has produced documents on technical guidance for developing nutrient criteria in estuaries and coastal waters.[39,40] The strategy also promoted dialog and partnership among the federal government agencies, state and tribal governments, and local organizations to address scientific issues, as well as technological feasibility of controlling excessive nutrient input coastal water bodies. However, developing water quality criteria for nitrogen has remained a daunting task, in part, because of complex biogeochemical cycling of nitrogen and its delivery to coastal bays and estuaries in vastly different amounts, chemical forms, and rates, and, in part, due to its multitude of effects, which include unsafe drinking water, soil acidity, smog, eutrophication, ozone depletion, and greenhouse warming at different concentrations.

As noted earlier, presently there are no nationwide numerical criteria to limit nitrogen pollution in coastal bays and estuaries in the United States. As of May 2007, only a handful of states had approved criteria for entire classes of rivers and streams (www.epa.gov/waterscience/criteria/nutrient/value/status.html).

The EPA document, National Strategy for the Development of Regional Nutrient Criteria, outlines the preferred regional approach for developing nutrient criteria and standards for different types of water bodies and provides guidance on measurements and indicators to state governments and recognized tribes.[39] A National Estuaries Experts Workgroup was formed in 2006 to provide guidance on this matter, and is expected to produce its observations and recommendations by the end of 2007.

Also, there are no toxicological benchmarks for nitrogen concentrations in the United States for protection of coastal and estuarine organisms. In Canada, an interim guideline intended for protection of coastal fauna and flora from direct toxicity of the nitrate ion, but not necessarily from impacts of eutrophication, is 16 mg NO$_3^-$/L.[10]

Nitrogen Management at Local and Regional Levels

Concerned about harmful or undesirable consequences of excessive nitrogen input to the estuaries, a number of regional or local communities and government agencies have taken steps to control nitrogen delivery to coastal bays and estuaries. One of the earliest such steps was taken by the Town of Falmouth, MA, which, through a by-law, tied watershed development to water quality in adjacent water bodies and laid the foundation of the PondWatch monitoring program in 1987. In recent years, several computer-based tools and models have been developed for predicting nutrient loading to estuaries and evaluating impacts of different nitrogen management options, including reducing wastewater inputs, preserving forested tracts, reducing fertilizer use, and altering zoning ordinances.[41]

Restoration of seagrass acreage in Tampa Bay, FL, through nutrient load reductions and lowered phytoplankton biomass is an example of collaboration among government entities and private industries. A Nitrogen Management Consortium coordinated development of nitrogen load targets that were estimated to improve water clarity that would be sufficient for allowing natural recovery of seagrasses. The actions started with wastewater treatment upgrades, and subsequently involved improved stormwater treatment, changes in fertilizer manufacturing processes and agricultural practices, increased use of reclaimed water (thus reducing surface water discharge from wastewater treatment plants), and significant reductions in atmospheric deposition of nitrogen from local electric power plants. Seagrass cover in the bay has increased more than 30% in comparison with estimated seagrass cover in 1982. A subarea of the bay, Old Tampa Bay, has not shown such recovery and El Nino events have occasionally impeded seagrass recovery.[42] It is anticipated that federal and state initiatives to promote cleaner-burning fuels, improve fuel economy for automobiles, and expand mass transit systems will further reduce nitrogen input to the bay and facilitate seagrass recovery in the area.

Nutrient Trading

The 2000 Chesapeake Bay Agreement and related multistate cooperative and regulatory initiatives established allocations for nitrogen and phosphorus delivered to the Chesapeake Bay and its tidal tributaries to meet applicable water quality standards and placed caps on the loads of these nutrients that may be discharged into the Chesapeake Bay watershed. Such initiatives require point source dischargers of nitrogen and phosphorus to achieve significant additional reductions of these nutrients to meet the required cap on the total quantity of nutrients entering a waterbody. For several reasons, nutrient trading in coastal waters and estuaries has remained remarkably slow and restrained since the publication of water quality trading policy by EPA in 2003. The reasons may include adequate nutrient control technologies already in place in major wastewater treatment plants, government subsidies and “green payments” to farmers for controlling nonpoint source discharges, uncertain trading ratios (e.g., between a treatment plant effluent and crop cover planting), and unexplored issues of buyer liability. It is also evident that nutrient trading may not be effective in coastal bays and estuaries where nonpoint source discharge of nutrients is predominant.

SUMMARY

The presence of excessive amounts of biologically reactive nitrogen in coastal bays and estuaries has been recognized as a major environmental issue. For the most part, this is due to its role in enhancing phytoplankton growth that, in turn, can lead to algal blooms, oxygen consumption in seawater and on the seabed, altered patterns of primary productivity, changes in species composition, and shading effects on macrophytes and seagrass beds. Of particular concern is the occurrence and magnitude of harmful algal blooms, for which nutrient over-enrichment could be the primary or an important co-factor.

Nitrogen sources to coastal waters and estuaries are diverse and include wastewater effluents, certain industrial
discharges, surface water runoff, groundwater discharge, the atmosphere, and influx from the coastal ocean. Atmospheric deposition can be a relatively major source of nitrogen to estuaries, particularly in areas where other nitrogen sources are relatively small. There are very few estuaries for which a balance sheet of nitrogen input and export has been developed. This is due to a lack of coherent monitoring programs that can provide a continuum of observations from the watershed to the coastal ocean, and in different environmental matrices to calculate or simulate source attribution of analytes of concern.

The apparent complexity of nitrogen’s role in promoting eutrophication has occasionally rekindled debate on the comparative influences of nitrogen and phosphorus on primary productivity in coastal and marine waters, particularly where mixing between surface and deeper waters is more pronounced. Corollary to this debate is the question of whether nitrogen is the scarcest of the nutrients in coastal bays and estuaries and whether a singular focus on nitrogen controls would be fruitful. Occasional reviews of nitrogen limitation of algal growth and nitrogen to phosphorus ratios in marine and freshwater environments suggest that nitrogen limitation can occur when the N:P ratio is less than 20, and phosphorus limitation when the ratio is more than 50, rather than whether the system is marine or freshwater. Even after years of focused research on the consequences of nitrogen over-enrichment in U.S. coastal waters and estuaries, several key issues remain, including the magnitude, spatial, and temporal extent of eutrophication and associated water quality and habitat impairments attributable to increasing nitrogen inputs to estuaries and coastal waters. 

REFERENCES