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Review

Coastal nitrogen pollution: A review of sources and trends globally and regionally

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ABSTRACT

The past few decades have seen a massive increase in coastal eutrophication globally, leading to widespread hypoxia and anoxia, habitat degradation, alteration of food-web structure, loss of biodiversity, and increased frequency, spatial extent, and duration of harmful algal blooms. Much of this eutrophication is due to increased inputs of nitrogen to coastal oceans. Before the advent of the industrial revolution and the green revolution, the rate of supply of nitrogen on Earth was limited to the rate of bacterial nitrogen fixation, but human activity now has roughly doubled the rate of creation of reactive, biologically available nitrogen on the land masses of the Earth. Regional variation in this increase is great, and some regions of the Earth have seen little change, while in other areas, nitrogen fluxes through the atmosphere and through rivers have increased by 10–15-fold or more. Much of this increase has occurred over the past few decades. Increased use of synthetic nitrogen fertilizer and increased intensity of meat production has led the change globally and in many regions, and agricultural sources are the largest source of nitrogen pollution to many of the planet's coastal marine ecosystems. The rate of change in nitrogen use in agriculture is incredible, and over half of the synthetic nitrogen fertilizer ever produced has been used in the past 15 years. Atmospheric deposition of nitrogen from fossil fuel combustion also contributes to the global budget for reactive nitrogen and is the largest single source of nitrogen pollution in some regions. Technical solutions for reducing nitrogen pollution exist at reasonable cost, but implementation has been poor in many regions.

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The past few decades have seen a massive increase in coastal eutrophication globally, leading to widespread hypoxia and anoxia, habitat degradation, alteration of food-web structure, loss of biodiversity, and increased frequency, spatial extent, and duration of harmful algal blooms (Howarth et al., 2000; NRC, 2000; Boesch, 2002). Much of this eutrophication is driven by nitrogen, although phosphorus pollution can also contribute to coastal eutrophication (NRC, 2000; Howarth et al., 2005; Howarth and Marino, 2006). In this paper, I briefly review information on the sources of nitrogen in the landscape that contribute to coastal pollution, drawing on several recent analyses and more comprehensive reviews (Howarth et al., 2002a,b, 2005, 2006a; Howarth, 2006, 2008).

The vast majority of nitrogen on Earth is molecular N_2, most of which is in the atmosphere but some is dissolved in the world's oceans. Only 0.002% of nitrogen on the planet is present in living tissues and detrital organic matter (Schlesinger, 1997). The proportion of biologically available inorganic nitrogen such as nitrate, nitrite, and ammonium is orders of magnitude less yet. Because nitrogen is essential for life yet biologically available forms are such a small proportion of nitrogen on Earth, nitrogen limits primary productivity in many of the world's ecosystems (Vitousek and Howarth, 1991), including most of the world's estuaries and coastal marine ecosystems in the temperate zone (Howarth and Marino, 2006). Even in the tropics where phosphorus is often thought to be more limiting, nitrogen limits primary production in some systems such as the pelagic zone of the Caribbean Sea (Corredor et al., 1999) and can become limiting in seagrass ecosystems subject to moderate nutrient loading

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(McGlathery et al., 1994; Howarth et al., 1995). Coastal eutrophication is largely a result of excess nitrogen loads (NRC, 2000).

Before the industrial revolution, the only mechanisms whereby molecular N_2 gas was converted into reactive, biologically available forms were bacterial nitrogen fixation and chemical reaction with oxygen under high heat associated with lightning and volcanoes. Of these, creation of reactive nitrogen associated with lightning and volcanoes is minor, probably creating less than 10 Tg N $year^{-1}$ (Vitousek et al., 1997). Estimates for bacterial nitrogen fixation are uncertain, but prior to the large-scale alteration of the landscape by human activity, bacterial nitrogen fixation on land was probably in the range of 90–195 Tg N $year^{-1}$ (Vitousek et al., 1997; Cleveland et al., 1999). Nitrogen fixation by cyanobacteria in the world's oceans is probably in the range of 200–300 Tg N $year^{-1}$ (Karl et al., 2002). The total rate of creation of reactive, biologically available nitrogen due to natural processes globally thus probably is in the range of 300–500 Tg N $year^{-1}$, with between 25% and 50% of this fixed on land.

During the 20th Century – and particularly over the past few decades – human activity created additional reactive nitrogen at a rate that globally competes with the natural rate of creation (Fig. 1; Galloway et al., 2004). The exponential growth from 1960 to 1990 was particularly dramatic and is associated with an "explosive increase" in the incidence of coastal marine eutrophication globally (Boesch, 2002). By 2000, the human-controlled creation of reactive nitrogen occurred at a rate of ~165 Tg N $year^{-1}$. Thus, human activity has increased the total rate of formation of reactive nitrogen globally by 33–55% (if natural fixation both in the oceans and on land is considered). Most of the nitrogen loading to eutrophic estuaries comes from land-based sources and not the ocean (Nixon et al., 1996), so from the perspective of coastal nutrient pollution, the extent of the global alteration of nitrogen sources is perhaps best made by comparing the human-controlled creation with natural nitrogen fixation on the land. From this perspective, human-controlled processes may now exceed the natural processes (Fig. 1), or at least are approaching this if the natural rate of nitrogen fixation is at the higher end of the range suggested by Cleveland et al. (1999). As a reflection of this, approximately 60 Tg N $year^{-1}$ are estimated to flow to the world's oceans in rivers under current conditions (Boyer et al., 2006; Boyer and Howarth, in press), or about twice as much as is thought to have occurred in 1860 (Galloway et al., 2004).

Human activity creates reactive nitrogen through three mechanisms: (1) encouragement of biological nitrogen fixation associated with agriculture; (2) production of synthetic nitrogen fertilizer; and (3) inadvertent creation of reactive nitrogen through reaction with oxygen as fossil fuels are burned (Galloway et al., 2004). Of these, the largest change during the 20th Century was in the rate of production of synthetic nitrogen fertilizer (Fig. 1). Synthetic nitrogen fixation now dominates agricultural inputs of nitrogen, yet the Haber-Bosch process for reacting N_2 with H_2 to produce ammonia was only discovered less than 100 years ago, in March 1909 by German chemist Fritz Haber (Charles, 2005). This discovery fueled the green revolution, and allowed a massive expansion of global agriculture, with concomitant decrease in hunger and malnutrition (Smil, 2001). The rate of change is astounding, and half of all of the synthetic nitrogen fertilizer ever used on Earth has been produced since 1985 (Howarth et al., 2002a, 2005). Today, over 80% of the nitrogen in the protein of the average human on Earth is derived originally from the Haber-Bosch process.

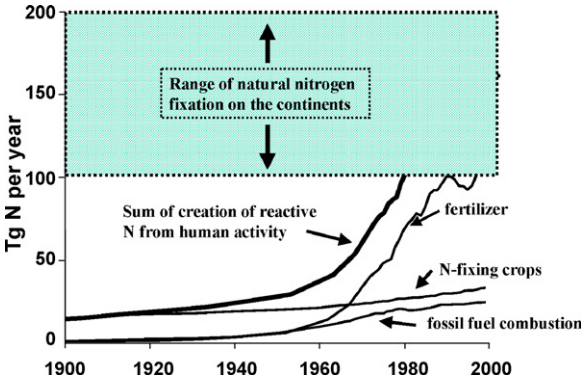

Fig. 1. Global trends in the creation of reactive nitrogen from human activity over the 20th Century compared to the natural rate of nitrogen fixation on the continents. Modified from Boesch (2002).

For the most part, the nitrogen cycle is not a global cycle, and reactive nitrogen generally moves on spatial scales of hundreds of meters to perhaps hundreds of kilometers through the atmosphere and in rivers and ocean currents. An exception is the gas N_2O (a potent greenhouse gas which also contributes to the destruction of ozone in the stratosphere), which has a residence time in the atmosphere of approximately 120 years (Schlesinger, 1997; Vitousek et al., 1997). In contrast, reactive compounds such as ammonium and nitrate have residence times on the planet of approximately 1 day due to rapid assimilation in biological processes (Howarth, 2002). Because most reactive nitrogen moves only over short distances and the amount of nitrogen created from both natural and human-controlled processes varies from region to region around the planet, the consequences of the human acceleration of the nitrogen cycle also differ on regional scales. Interestingly, most of the natural biological nitrogen fixation on the planet occurs in the tropics (Cleveland et al., 1999; Karl et al., 2002), while during the 20th Century most production and use of reactive nitrogen by humans occurred in temperate regions, the site of most industrial and agricultural activity (Holland et al., 1999). Thus human alteration of the nitrogen cycle to date has been much greater in temperate regions than in the tropics.

One measure of how humans have altered the nitrogen cycle is the deposition rate of nitrogen from the atmosphere. On average worldwide, deposition has probably doubled in the tropics and increased more than sixfold in the north temperate zone as a result of human activities (Holland et al., 1999). On the other hand, human activity has had less effect on nitrogen deposition in the temperate zone of the Southern Hemisphere because of much less agriculture and industry.

Another measure of the extent of change in the nitrogen cycle is the flux of nitrogen in rivers to coastal seas and oceans (Fig. 2). As is discussed further below, climate can influence riverine nitrogen fluxes, and fluxes are higher in regions with more precipitation and freshwater discharge (Howarth et al., 2006a). Nonetheless, human activity is the major influence on riverine nitrogen fluxes, and the natural background flux of nitrogen off the landscape in regions with little human influence averages approximately 100 kg N km^{-2} $year^{-1}$ (NRC, 2000; Howarth et al., 2002a, 2005). The riverine flux of nitrogen in the region of Labrador and Hudson Bay is close to that level (Howarth et al., 1996), whereas in regions with more agriculture and industry, nitrogen fluxes can be far higher. The flux from the Republic of Korea to the coast is 17-fold higher (Bashkin et al., 2002), while that from the watersheds flowing to the North Sea is 15-fold greater (Howarth et al., 1996). The nitrogen flux down the Mississippi River is elevated some fivefold to sixfold, while that from the watersheds of the northeastern United States from Maine through Chesapeake Bay are on average some 10-fold higher than background (Howarth et al., 1996).

The sources of human inputs of nitrogen vary greatly among different regions of the world. Globally, the production of synthetic

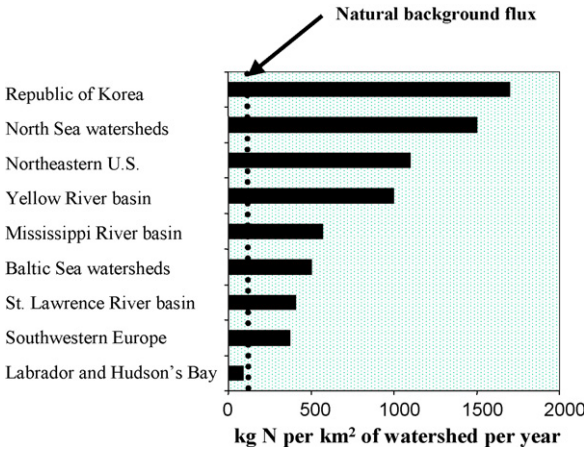

Fig. 2. Average annual flux from the landscape to coastal oceans in rivers for contrasting regions of the world in the temperate zone, expressed per area of watershed. Based on data in Howarth et al. (1996, 2002a, 2005) and Bashkin et al. (2002).

nitrogen fertilizer is the single biggest alteration of the nitrogen cycle by humans, and in many regions and watersheds, agriculture dominates the nitrogen flux. For example, the single largest input of nitrogen to the Mississippi River basin is synthetic nitrogen fertilizer, followed by nitrogen fixation associated with agricultural crops such as soybean (Fig. 3). Note that the nitrogen that leaves the Mississippi River basin in the export of food and feed for humans and animals – most of this is corn and soybeans coming down the River in barges – is actually greater than the amount of nitrogen pollution that flows down the River. In the northeastern United States (defined here as the region south through the watersheds in Virginia that flow into Chesapeake Bay), the single largest input of nitrogen is atmospheric deposition of oxidized nitrogen compounds (NOy), with the nitrogen originating from fossil fuel combustion (Fig. 3). The importation of food and feeds to the region is the second largest input, with agricultural inputs of synthetic nitrogen fertilizer and nitrogen fixation also contributing.

At these large spatial scales, the flux of nitrogen to the coast in rivers is a function of the net anthropogenic nitrogen inputs (NANIs), defined as the sum of nitrogen inputs as synthetic fertilizer, in nitrogen fixation associated with agriculture, as NOy deposition, and the net import or export of nitrogen in foods and feeds (Howarth et al., 1996, 2002a,b; NRC, 2000) (Fig. 4). Note that the nitrogen export in food and feed from the Mississippi is subtracted from the other inputs to get NANI, while the net nitrogen import in food and feed in the northeastern US is added to the other inputs. NANI is greater in the northeastern US than in the Mississippi, when expressed per area of watershed, and the riverine nitrogen flux to the coast is also correspondingly larger (Figs. 3 and 4). Non-point source fluxes dominate the riverine nitrogen flux, as is true of all the regions surrounding the North Atlantic Ocean (Howarth et al., 1996). The wastewater flux is greater in the northeastern US than in the Mississippi, reflecting a higher population density. At smaller spatial scales, wastewater can be the dominant input to some estuaries, such as the Hudson River estuary and New York City Harbor, but non-point sources dominate in most estuaries (Nixon et al., 1996; Alexander et al., 2000; Howarth et al., 2006b).

Agriculture contributes to nitrogen pollution of surface waters both from direct runoff from agricultural fields and from nitrogen in animal wastes. On average in the United States, nitrogen inputs to fields are used with a surprisingly high efficiency (Fig. 5). As of the late 1990s, the nitrogen removed from fields in harvested crops

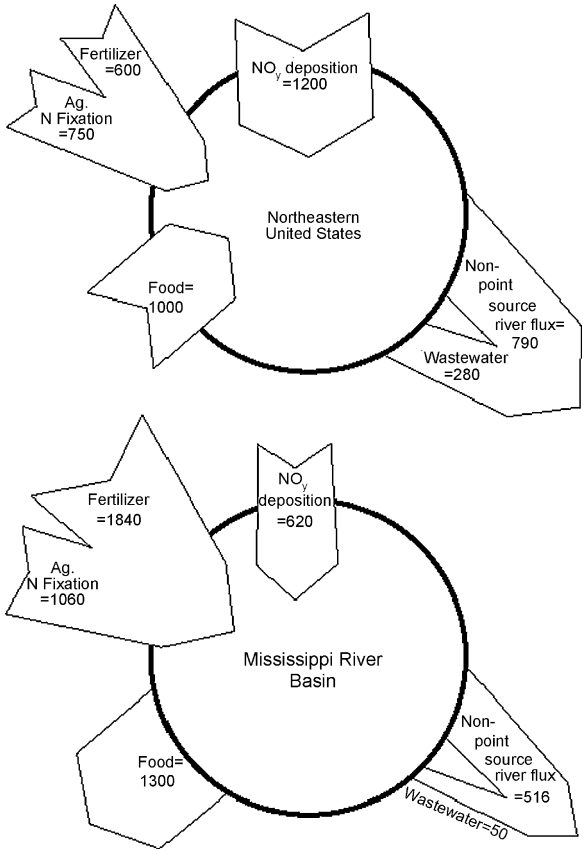

Fig. 3. A comparison of the regional nitrogen budgets for the northeastern United States from Maine through Chesapeake Bay (top) and for the Mississippi River basin (bottom). Units are kg N km^{-2} $year^{-1}$. Inputs to the Mississippi River basin are dominated by agriculture, while atmospheric deposition, the import of food and feed, and agricultural inputs of nitrogen as fertilizer and through nitrogen fixation all contribute in the northeastern United States. Note that while nitrogen is imported to the northeastern United States in food and feed, there is a net export of nitrogen in food and feed from the Mississippi basin. The riverine flux is greater for the northeastern United States than for the Mississippi River, when expressed per area of watershed. However, the area of the Mississippi basin is much greater, and so the total nitrogen flux is larger there. Reprinted from Howarth et al. (2002a).

was on average more than half of the total nitrogen inputs to the field and almost equal to the inputs of synthetic nitrogen fertilizer (Howarth et al., 2002b). Nonetheless, significant nitrogen leaves fields in surface runoff and groundwater, an amount estimated to be approximately 30% of the rate of synthetic nitrogen application or 20% of the total nitrogen inputs to the fields as of the late 1990s (NRC, 1993; Smith et al., 1997; Howarth et al., 2002b). Note, though, that the variability in nitrogen losses from fields is great, ranging from a low of 3% of the rate of nitrogen fertilization on grasslands with clay-loam soils to a high of 80% of the rate of nitrogen fertilization for row-crop agriculture on sandy soils (Howarth et al., 1996). Approaches for reducing nitrogen pollution from agricultural fields are discussed later in this manuscript.

Of the nitrogen that leaves fields in harvested crops in the United States, the majority is fed to animals: 70% of the harvest that is not exported from the country (Fig. 5). Thus, a major driver in the use of synthetic nitrogen fertilizer in agriculture is to grow crops to feed to animals. Per capita consumption of meat, meat products, and poultry in the United States is the second highest in the world, and continues to rise, increasing this trend (Howarth et al., 2002b). Further, the nitrogen in crops is converted into nitrogen in animal protein with relatively low efficiency, and so much of this nitrogen ends up in animal wastes and subsequently

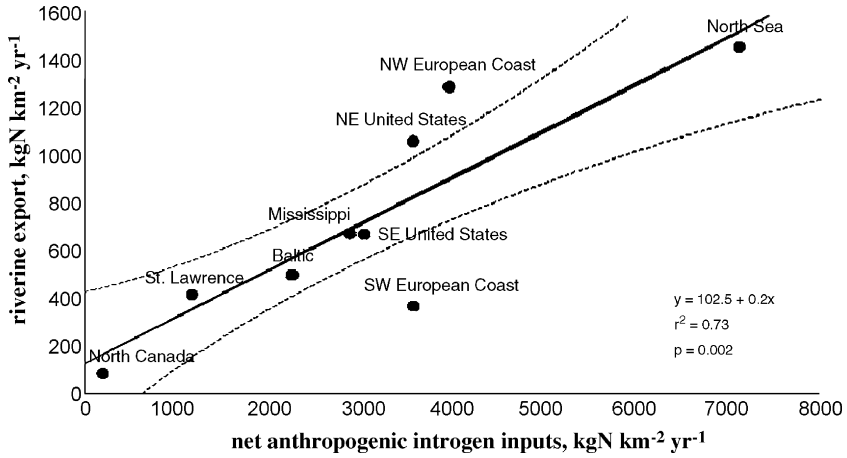

Fig. 4. Average annual nitrogen export per area of watershed from large regions around the North Atlantic Ocean to the coastal ocean as a function of net anthropogenic nitrogen inputs to the landscape per area. Reprinted from Howarth et al. (1996).

pollutes the environment. An estimated 30% is volatilized to the atmosphere as ammonia, with most of this deposited near the site of emission (Howarth et al., 2002b), Most of this is deposited near the site of emission. The fate of most of the nitrogen in animal wastes is poorly known, but it contributes directly or indirectly to water pollution (Fig. 5). An important point is that the widespread availability of synthetic nitrogen fertilizer has allowed animal agriculture to become physically separated from the site of crop production, as it is no longer necessary to return the nitrogen in animal wastes to fields to maintain adequate soil fertility. By the 1990s, most of the grain produced on farms in the United States left those farms to feed animals elsewhere, and the major animal-producing states imported more than 80% of the grains used in animal feeding operations (NRC, 2000; Howarth et al., 2002a). Prior to the 1960s, most animals were fed on locally produced crops, and most of the nitrogen in waste was returned to those nearby fields as essential fertilizer.

The amount of nitrogen inadvertently fixed during the combustion of fossil fuels is relatively small compared to the production of synthetic fertilizer on a global basis (Fig. 1). Nonetheless, as noted

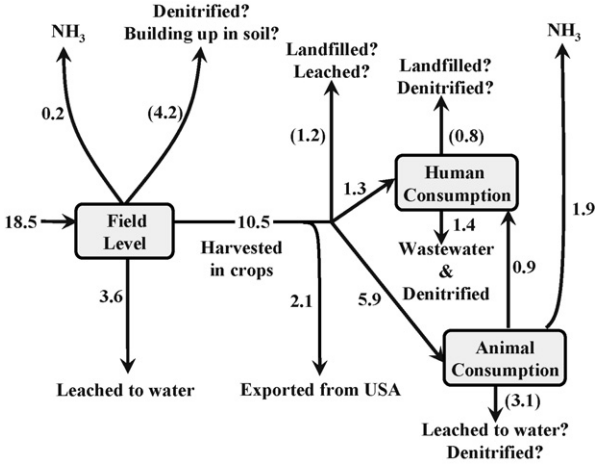

Fig. 5. Average fate of nitrogen inputs to agricultural fields in the United States as of the late 1990s (Tg N year^{-1}). Numbers in parentheses are calculated by difference, while the other numbers are direct estimates. The input of 18.5 Tg N year^{-1} includes inputs of synthetic nitrogen fertilizer (11.2 Tg N year^{-1}), nitrogen fixation associated with agricultural crops (5.9 Tg N year^{-1}), and atmospheric deposition of oxidized nitrogen from fossil fuel combustion that falls on agricultural fields (1.4 Tg N year^{-1}). Reprinted from Howarth et al., 2002b.

above, the resulting nitrogen pollution in the atmosphere is deposited onto the landscape or onto coastal waters directly and can be a significant source of nitrogen pollution in some regions. In the United States, both fossil fuel combustion and rates of emission of nitrogen pollution to the atmosphere are the highest in the world on a per capita basis (Howarth et al., 2002b). Nitrogen emissions from fossil fuel combustion are almost two-thirds of the rate of use of synthetic nitrogen fertilizer, and perhaps not surprisingly, on average for the United States approximately 30% of coastal nitrogen pollution is estimated to come from this source (Howarth and Rielinger, 2003).

The atmospheric nitrogen pollution deposited directly onto the surface of coastal waters can contribute between 1% and 40% of the total nitrogen inputs to coastal ecosystems (Nixon et al., 1996; Paerl, 1997; Howarth, 1998; Paerl and Whitall, 1999; Valligura et al., 2000). Direct deposition is most significant in very large systems, such as the Baltic Sea (Nixon et al., 1996), or in coastal systems with a small ratio of watershed to surface water area, such as Tampa Bay (Zarbock et al., 1996).

Coastal marine ecosystems also receive some nitrogen that is deposited onto their watersheds and then exported downstream, and often, this is a greater input that is the direct deposition onto water surfaces, although there is significant uncertainty in estimating these fluxes. For recent overviews on the sources of uncertainty in estimating the importance of deposition as a nitrogen load to coastal systems, see Howarth (2006, 2008). One issue is that the percentage of deposited nitrogen that is exported downstream as opposed to retained in terrestrial ecosystems is highly variable (Howarth et al., 2002a; Aber et al., 2003; Castro et al., 2007). Another problem is that total nitrogen deposition is poorly measured. Nitrogen that falls in precipitation (wet deposition) is well measured in the United States by the National Atmospheric Deposition Program (NADP), and good models exist for extrapolating the results over the landscape (Ollinger et al., 1993; Grimm and Lynch, 2005). However, spatial coverage for monitoring dry deposition – the nitrogen that is deposited as gases and in particles – is poor in both the United States and Europe (Holland et al., 2005), and the deposition of some major nitrogen gases (nitric oxide, nitrogen dioxide, peroxy- and alkyl-based organics, and ammonia gas) is not measured at all in monitoring networks. Recent model estimates suggest that total nitrogen deposition onto the watersheds of Chesapeake Bay may have been underestimated by 40% due to lack of consideration of the deposition of these gases (Robin Dennis, NOAA National Air Lab, pers. comm.).

Using a statistical model (SPARROW) to evaluate sources of nitrogen pollution from surface-water monitoring networks, Alexander et al. (2000) estimated that nitrogen deposition onto the landscape contributed between 4% and 35% of the total nitrogen inputs to a range of estuaries across the United States. Their analysis was based on NADP wet-deposition data; if dry deposition rates were correlated with wet deposition, the NADP data would serve as a fine surrogate for total deposition in the SPARROW model. However, gaseous dry deposition is not well correlated with wet deposition, and is much higher in urban areas and near highways due to local deposition of vehicle emissions (Robin Dennis, NOAA National Air Lab, pers. comm.; Cape et al., 2004; Howarth, 2006, 2008; Howarth et al., 2006a). The analysis of Alexander et al. (2000) therefore probably underestimates the importance of atmospheric deposition. Interestingly, their statistical model ascribed a large flux of nitrogen into many estuaries from "non-point, non-agricultural" sources (Table 1). Some of this – and perhaps most of it – may reflect dry gaseous deposition in urban and suburban areas (Howarth, 2008). Thus, the range of inputs of nitrogen to estuaries from deposition determined from the SPARROW model would be between the estimate for deposition from the NADP data and the sum of that estimate plus the non-point, non-agricultural source (Table 1). For Chesapeake Bay, this range is 28–50% of the total nitrogen inputs, which is in agreement with recent estimates from several other modeling approaches (Howarth, 2006, 2008; Gary Schenk, Chesapeake Bay Program, pers. comm.).

As noted above, the flux of nitrogen to the coast in large regions and river basins is a function of the nitrogen inputs. However, only a relatively small proportion of NANI (15–45%) flows to the coast (Howarth et al., 1996, 2006a,b; Boyer et al., 2002). The rest is retained in the landscape or is denitrified to N_2 or N_2O gas. A better understanding of these sinks in the landscape is essential to predicting future trends in coastal nitrogen pollution. If the major sinks are accumulation of organic nitrogen in trees, soils, or groundwater aquifers, then the sinks are likely to saturate over time. Given comparable nitrogen inputs, nitrogen fluxes to coastal ecosystems would increase in the future. On the other hand, denitrification rates might be expected to continue into the future without major change, as long as the sites of denitrification are protected. Unfortunately, it is extremely difficult to measure either denitrification at the landscape scale or the rate of accumulation of nitrogen in trees, soils, and aquifers. In the best analysis anywhere of these landscape-scale sinks of nitrogen, Van Breemen et al. (2002) used a variety of models to estimate the sinks for the major river basins in the northeastern United States. Their conclusion – which is highly uncertain – is that of the nitrogen input that is not exported to the coast in rivers, 27% is accumulating in soils and forest biomass and 73% is being denitrified (57% in the landscape, and 16% in rivers).

Nitrogen fluxes to coastal ecosystems in river flow also are related to climate. For the Mississippi River, McIsaac et al. (2001) demonstrated that the large annual variation in nitrate flow could be explained well by a simple model that considers NANI and river discharge; the model allows more storage of NANI in the watershed during dry years and greater export of the stored NANI in years when discharge was higher (Fig. 6). There is also evidence that climate affects the landscape-scale sinks of nitrogen and the average amount of NANI that is exported downstream in rivers over multi-year periods. For the 16 major river basins in the northeastern United States, the percentage of NANI that flows downriver to coastal ecosystems is up to 45% in the watersheds where average precipitation and discharge are higher (Howarth et al., 2006a). In drier regions, only 10–20% of NANI are exported (Fig. 7). There may also be a relationship to temperature, with greater percentage exports where the climate is cooler, but the relationship is not as strong as for precipitation and discharge (Howarth et al., 2006a). The greater export of nitrogen in wetter watersheds may reflect less denitrification in hot spots such as riparian wetlands and low-order streams due to a shorter water residence times in these systems.

The combined influences of NANI and either precipitation or discharge can explain the mean flux of nitrogen in the northeastern United States rivers with reasonably high precision (R^2-values of 8.87–0.90; Howarth et al., 2006a). Howarth et al. (2006a) used NANI and precipitation to predict possible climate change consequences on nitrogen fluxes for the Susquehanna River basin; the Susquehanna is the single largest source of nitrogen to Chesapeake Bay. Given the climate change predictions for increased precipitation (Najjar, 1999; Najjar et al., 2000), and assuming no change in NANI or land use, an increase in nitrogen flux down the Susquehanna of 17% by 2030 and 65% by 2095 is predicted (Howarth et al., 2006a). Clearly, in regions where climate change will lead to greater precipitation and discharge, coastal nitrogen pollution may be aggravated.

There is some good news in the coastal nitrogen pollution story: technical solutions exist to reduce nitrogen inputs from all potential sources, and generally at reasonable cost. Some such solutions as well as policies for implementing them effectively

Table 1
Estimates from the SPARROW model for the relative importance of atmospheric deposition, "non-agricultural non-point sources," and sewage wastewater as nitrogen inputs to several coastal marine ecosystems in the northeastern United States

	Atmosphere (%)	Non-ag non-point (%)	Wastewater (%)
Casco Bay	22	54	13
Great Bay	9	58	23
Merrimack River	28	43	20
Buzzards Bay	12	14	63
Narragansett Bay	10	19	62
Hudson River	26	21	40
Barnegat Bay	19	28	43
Delaware Bay	22	17	35
Chesapeake Bay	28	22	8

Note that the atmospheric deposition terms are estimated just from wet deposition monitoring data. Note further that the "non-agricultural non-point sources" may include a substantial amount of input from dry atmospheric deposition near emission sources in urban and suburban environments, and this would not be included in the SPARROW estimate of the atmospheric deposition input. See text for further discussion. Based on Alexander et al. (2000). Reprinted from Howarth (2008).

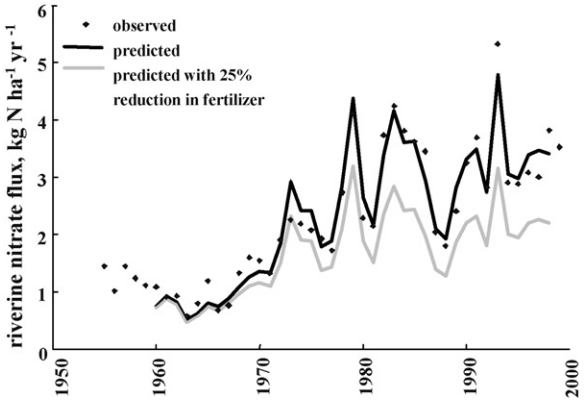

Fig. 6. Measured and predicted nitrate flux down the Mississippi River from 1955 to 2000. The predicted flux is based on a simple model of net anthropogenic nitrogen inputs and river discharge. Also shown is the predicted flux if the use of nitrogen fertilizer had been 25% less than actual use. Modified from McIsaac et al. (2001).

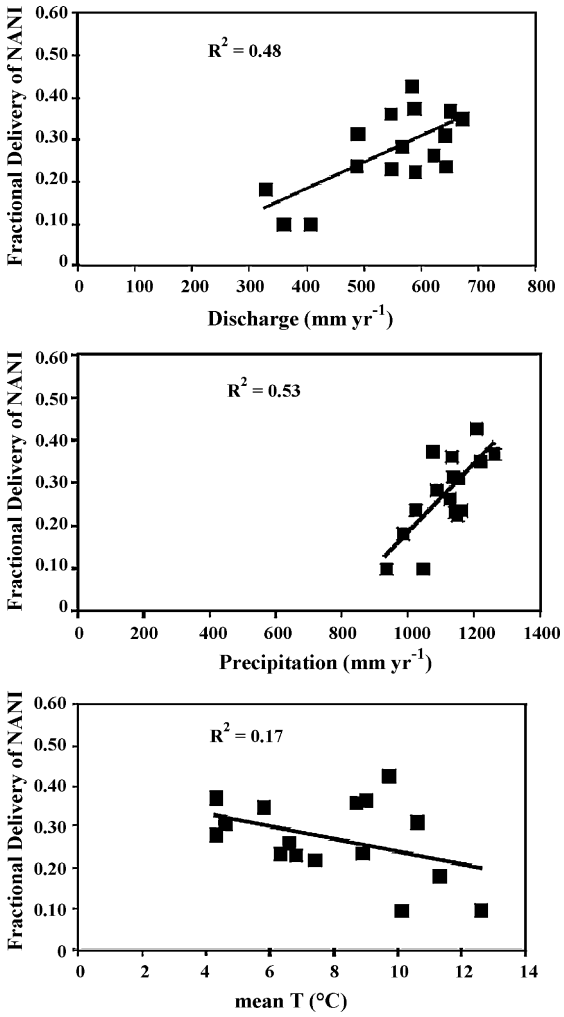

Fig. 7. The fractional delivery of net anthropogenic nitrogen inputs (NANI) for 16 major watersheds in the northeastern United States plotted as a function of mean discharge, mean precipitation, and mean temperature. The relationship for discharge and precipitation are highly significant ($p = 0.003$ and 0.0015, respectively); the relationship for temperature is weaker ($p = 0.11$). Reprinted from Howarth et al. (2006a,b).

both in the United States (Howarth, 2005) and globally (Howarth et al., 2005) have been reviewed recently for the International Nitrogen Initiative and for the Millennium Ecosystem Assessment. A few examples can illustrate the wealth of technical solutions. To reduce emissions of nitrogen to the atmosphere from fossil fuels, catalytic converter technology could be applied more aggressively to trucks and sports utility vehicles and "grandfathered" electric power plants built in the United States before the Clean Air Act could be brought up to modern standards (Moomaw, 2002). Incredible progress is being made on technologies for treating nitrogen from concentrated animal wastes, and these technologies are beginning to be applied in Asia and Europe (Choi and Yun, 2004). Nitrogen losses from agricultural fields can be reduced greatly by changing cropping systems; for example, land in Iowa and Minnessota planted in perennial grasses lost 30–50 times less nitrate than did fields planted in corn and soybeans (Randall and Mulla, 2001). Corn as a crop is particularly leaky of nitrogen, a condition aggravated by a tendency for many farmers to over-fertilize even when judged by economic return (Howarth, 2006). Unfortunately, the rush to produce more ethanol from corn for use as a fuel is increasing the land area planted to corn and may also be increasing the rate of fertilizer application, with potential to aggravate nitrogen pollution in regions such as the Mississippi River basin and the northern Gulf of Mexico (Simpson et al., 2008).

Unfortunately, the good news that technical solutions exist for nitrogen pollution must be tempered by the reality to date: while progress has been made in both Europe and the United States in reducing nitrogen pollution from municipal wastewater sources, little if any measurable progress has been made anywhere in reducing non-point source nitrogen pollution (NRC, 2000; Boesch et al., 2001; Howarth et al., 2005). One possible reason may be the long time needed for decreased nutrient losses from agriculture to show up as reduced riverine transport in some watersheds (Grimwall et al., 2000). Another reason is that some commonly used management practices are less effective than had been assumed and are not adequately assessed in actual practice (NRC, 2000; Boesch et al., 2001). Many managers have failed to recognize that nitrogen is much more mobile in the environment than is phosphorus, and management practices designed to reduce phosphorus pollution have been applied on the incorrect assumption that they also work effectively for nitrogen (Howarth, 2005). For example, no-till agriculture can reduce erosion and downstream phosphorus pollution but is not effective at reducing nitrogen losses (Randall and Mulla, 2001; Howarth, 2006). Failure to accurately determine the sources of non-point source nitrogen pollution have further aggravated the situation in some cases (NRC, 2000; Howarth et al., 2002a, 2005; Howarth, 2005, 2006). Clearly, reducing coastal nitrogen pollution will require better application of accurate information and adaptive management using quality science.

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