6. Atmospheric Deposition and Nitrogen Pollution in Coastal Marine Ecosystems

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Since the first appreciation of the widespread occurrence of acid rain in North America (Likens and Bormann 1974), most public attention has focused on the acid component rather than effects from the associated elements in atmospheric deposition. The emphasis has been on freshwater ecosystems and forests in sensitive regions with relatively low buffering capacity. Effects of acid deposition on coastal marine ecosystems have usually not been considered, which makes sense in the context of acidity. Marine ecosystems are very well buffered, since they contain large amounts of dissolved carbonate and bicarbonate, and consequently are quite insensitive to acid inputs. Similarly, marine waters contain huge quantities of sulfate (~ 28 mM) and thus are not sensitive at all to inputs of sulfate associated with acid deposition. On the other hand, nitrogen (N) pollution can cause severe degradation in coastal marine ecosystems, and the role of atmospheric deposition as a contributor of nitrogen to coastal waters has received increasing scrutiny over the past 15 years since Fisher and Oppenheimer (1991) noted that the nitrate anion associated with nitric acid in acid rain may be a major source of nitrogen to Chesapeake Bay.

The effects of nitrogen in marine ecosystems are due to its fertilizing effect. In most coastal marine ecosystems, rates of primary production are limited by the supply of available forms of nitrogen (primarily nitrate and ammonium). As nitrogen availability increases, so do rates of primary production by phytoplankton (Nixon et al. 1996). In moderation, this can be viewed as beneficial, since it can also lead to greater rates of production higher up the trophic structure leading, for example, to greater fish production (Nixon 1988; Caddy 1993). However, excess inputs of nitrogen lead to eutrophication and associated deleterious ecological changes (Figure 6.1; Caddy 1993; Nixon 1995; National Research Council 2000; Rabalais 2002). These changes include hypoxia (low oxygen zones) and anoxia (zero oxygen zones), alterations in community structure, degradation of habitat quality, loss of biotic diversity and increased incidences and duration of harmful algal blooms (National Research Council 2000; Rabalais 2002). Two thirds of the coastal rivers and bays in the United States are moderately or severely degraded from nitrogen pollution (Bricker et al. 1999), and excess

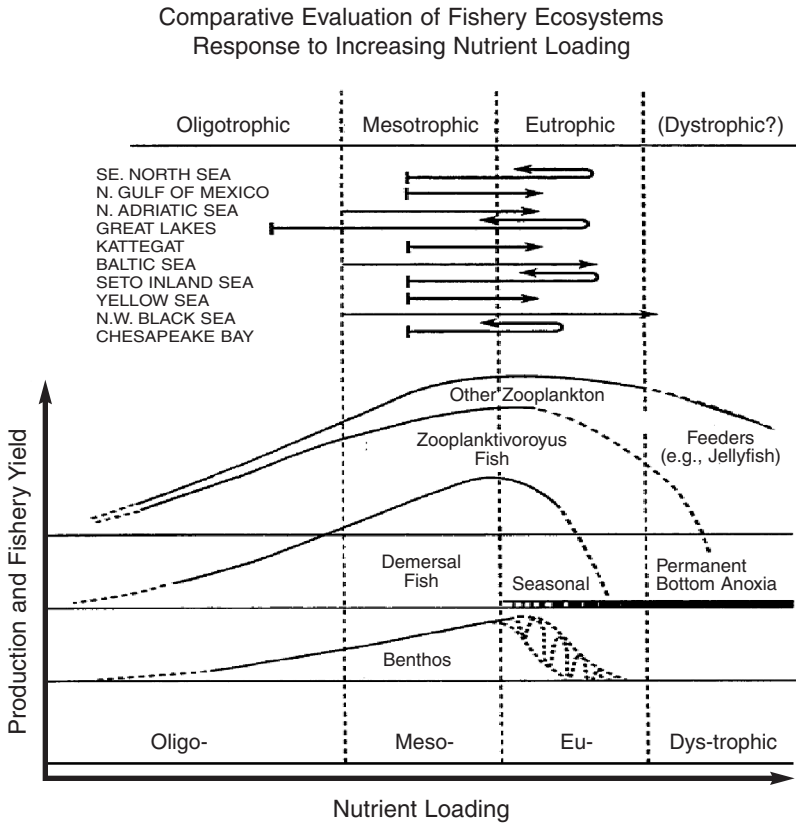

Figure 6.1. The response of secondary production and fishery yield in coastal marine ecosystems to increased nutrient loads (primarily nitrogen). Note the non-linear responses, with production at all trophic levels increasing as nutrient inputs increase at lower levels of loading, but declines in secondary production and fishery yields at higher nutrient loads. Demersal fisheries (those that are dependent upon the bottom waters and sediment) decline before fish dependent on zooplankton in the water column for their food. Modified from Caddy (1993)

Nitrogen inputs represent the largest pollution problem in the nation's coastal waters and one of the greatest threats to the ecological integrity of these ecosystems (National Research Council 1993, 2000; Carpenter et al. 1998; Howarth et al. 2000, 2002a, 2005). The inputs of nitrogen to the coastal waters of the United States are projected to continue to increase over future decades (Howarth et al. 2002a, 2005), in part due to rapid population growth in the coastal zone (Paerl 1997).

1. How Important is Atmospheric Deposition as a Source of Nitrogen Pollution to Coastal Waters?

For the United States as a whole, we have estimated that atmospheric deposition of nitrogen that originates from fossil-fuel combustion contributes on average 30% of the total nitrogen inputs to coastal marine ecosystems, while another 10% of these nitrogen inputs come from ammonia volatized into the atmosphere from agricultural sources (Howarth and Rielinger 2003). The rest of the nitrogen inputs to coastal waters come from runoff from agricultural sources and from municipal and industrial wastewater streams. The variation in both the fluxes of nitrogen to the coast and the source of the nitrogen is great across regions (see below), and in some regions and watersheds atmospheric deposition may be the single largest source of nitrogen.

Some of the nitrogen inputs from atmospheric deposition is nitrogen deposited directly onto the surface of coastal waters. This direct deposition to surface waters often contributes between 1% to 40% of the total nitrogen inputs to coastal ecosystems (Nixon et al. 1996; Paerl 1997; Howarth 1998; Paerl and Whitall 1999; Valigura et al. 2000). The direct deposition is most significant in very large systems, such as the Baltic Sea (Nixon et al. 1996) or in coastal systems which have relatively small watersheds in comparison to the area of the surface waters, such as Tampa Bay (Zarbock et al. 1996).

In most coastal marine ecosystems, the major route whereby atmospheric deposition contributes nitrogen is not the direct deposition onto surface waters but rather deposition onto the terrestrial landscape with subsequent downstream export in streams and rivers. As discussed below, these fluxes are difficult to measure, leaving significant uncertainty and debate about their magnitude. In the northeastern United States as a whole (Gulf of Maine through Chesapeake Bay), our studies have suggested that atmospheric deposition is the single largest source of nitrogen to coastal waters (Howarth et al. 1996; Jaworski et al. 1997; Boyer et al. 2002), while other studies have concluded atmospheric nitrogen deposition is the second largest source after wastewater discharges from sewage treatment plants (Driscoll et al. 2003). Our approach leads to the conclusion that atmospheric deposition of nitrogen onto the landscape—considering only the deposition of oxidized nitrogen compounds that originate from fossil fuel combustion (NOy)—contributes between 25% to 80% of the nitrogen flux in the different major rivers of New England (Figure 6.2; Boyer et al. 2002; Howarth and Rielinger 2003) and approximately 25% of the nitrogen flux in the Mississippi River (National Resource Council 2000; Howarth et al. 2002b). Using another approach—SPARROW, or Spatially Referenced Regressions

on Watershed Attributes model—Alexander et al. (2000) concluded that atmospheric deposition onto the landscape contributed between 4% and 35% of the nitrogen flux in 40 major coastal watersheds across the United States, with the highest contribution in the northeastern and mid-Atlantic regions.

The uncertainty over the contribution of atmospheric deposition as a nitrogen source to coastal marine ecosystems stems from two issues: uncertainty over the magnitude of nitrogen deposition onto watersheds (particularly from "dry deposition"), and uncertainty over the amount of the deposited nitrogen that is subsequently exported downstream (National Resource Council 2000; Howarth et al. 2002b). Each of these is discussed in some detail in the following sections.

2. How Large is Dry Deposition?

The vast majority of measurements of nitrogen deposition in the United States—including those made by the National Atmospheric Deposition Program (NADP)—measure only "wet deposition" (i.e., N in rainfall and

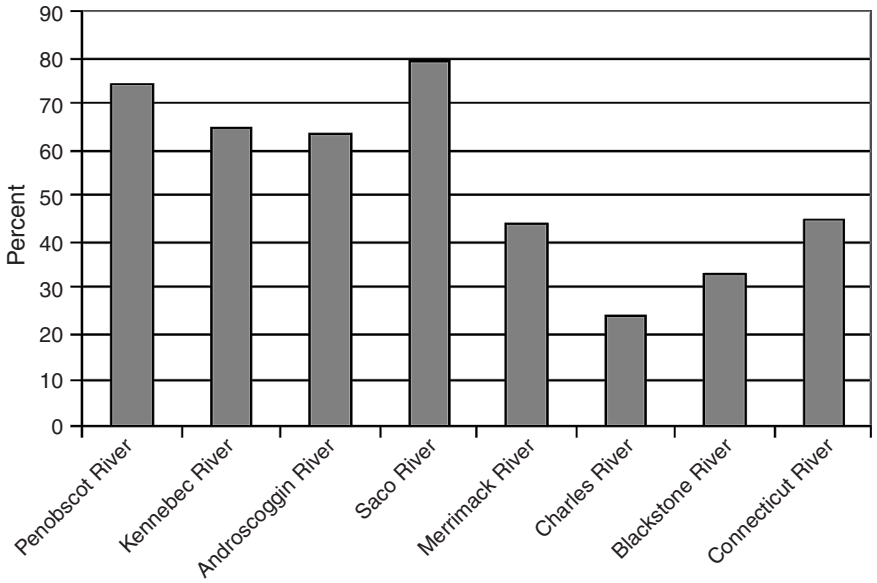

Figure 6.2. Percentage of nitrogen in major New England rivers that originates from fossil-fuel derived atmospheric deposition onto the landscape. Reprinted from Howarth and Rielinger (2003), based on data in Boyer et al. (2002)

snow). To estimate wet deposition onto an entire watershed, data at particular monitoring sites are extrapolated statistically considering factors such as local topography and precipitation (Ollinger et al. 1993; Grimm and Lynch 2005).

Substantial quantities of nitrogen can be deposited from the atmosphere as "dry deposition," which includes aerosols and other particles and uptake of gaseous forms of nitrogen by vegetation, soils, and surface waters. Both in the United States and Europe, the extremely sparse spatial coverage in networks for measuring dry deposition severely limits estimation of this process (Holland et al. 2005). In the United States, dry deposition is routinely measured only at sites that are part of the CASTNet and AIRMon-Dry programs. At the peak of these programs in the 1990s, these networks consisted of a total of 93 sites across the country, but the number is now substantially less (pers. communication, Richard Artz, Deputy Director of the NOAA Air Resources Lab). In the watersheds of Chesapeake Bay—an area of 165,000 km^2 that includes land in six states—there are only eight stations for monitoring dry deposition. In practice, given the sparse coverage of dry deposition monitoring, rates are often estimated as some percentage of wet deposition, based on comparisons of these two processes at a relatively few sites (Ollinger et al. 1993; Holland et al. 1999, 2005).

In addition to the limited spatial extent of the dry deposition monitoring networks, these networks do not measure all of the components that can be deposited. For example, particulate nitrate and particulate ammonium are routinely measured, as is nitric acid vapor. However, other gaseous nitrogen compounds that may play a significant role in deposition (i.e., nitric oxide, nitrogen dioxide, peroxy and alkyl based organics, and ammonia gas) are not measured. Nitric oxide and nitrogen dioxide are the major gases emitted from fossil fuel combustion, while ammonia is the major form of air pollution from agricultural sources. Ammonia is also released in vehicle exhaust, although at lesser amounts than for nitric oxide and nitrogen dioxide (Baum et al. 2001; Cape et al. 2004). To the extent these compounds are deposited, the dry depositional monitoring networks are underestimating total deposition. As currently measured, the dry deposition at the eight CASTNet sites in the Chesapeake Bay watershed ranges from 23% to 38% of total deposition, but the actual contribution when all forms of nitrogen gases are considered must certainly be higher.

There are other potential sources for the underestimation of nitrogen dry deposition. The manner in which dry deposition rates are calculated—multiplying concentration data obtained at the monitoring sites by "depositional velocities"—may also result in underestimation. For the AIRMon and CASTNet sites, these deposition velocities are estimated as a function of

vegetation and meteorological conditions (Clarke et al. 1997). Our knowledge of depositional velocities is based on studies in flat, homogenous terrain; as noted by Bruce Hicks (former Director of the NOAA Air Resources Lab), when estimating dry deposition "we are simulating the world on the assumption that our understanding of [these] special cases applies everywhere … We often display unwarranted confidence" in our estimates (Hicks presentation to the annual meeting of the American Society of Meteorology, October 2005). Complex terrain is likely to substantially increase depositional velocities. Vegetative cover is also important, and different models can vary in their estimates of spatial integrated dry deposition by more than five-fold depending upon different assumptions of the effect of vegetation (particularly coniferous forests) on depositional velocities (Weseley and Hicks 1999; Holland et al. 2005).

3. Two Approaches for Estimating Total Nitrogen Deposition in the Northeastern United States

Boyer et al. (2002) estimated the average deposition of oxidized nitrogen (NOy) onto the landscape of the major rivers of the northeastern United States, including both wet and dry deposition, following the approach of Ollinger et al. (1993) in using a statistical extrapolation of wet deposition monitoring data and relating dry deposition to wet deposition at particular sites. We found a range of values across these watersheds from ~ 360 kg N km^{-2} yr^{-1} in the Penobscot River basin in Maine to ~ 890 kg N km^{-2} yr^{-1} in the Schuylkill River basin in Pennsylvania (Boyer et al. 2002). The average value for this set of watersheds is ~ 680 kg N km^{-2} yr^{-1}.

Another approach for estimating nitrogen deposition onto the landscape can be obtained from models based on emissions to the atmosphere, with consideration of reaction and advection in the atmosphere, followed by deposition (Figure 6.3). We used one of these models (the GCTM model; Prospero et al. 1996) to estimate nitrogen deposition in all of the regions—including the northeastern United States—that surround the North Atlantic Ocean (Howarth et al. 1996). The GCTM model predicts depositional patterns globally at a relatively course spatial scale using emission sources as inputs and modeling atmospheric transformations and transport (Prospero et al. 1996). For the northeastern United States, the GCTM model yielded an estimated total nitrogen deposition (wet plus dry) of ~ 1,200 kg N km^{-2} yr^{-1}, a value 80% greater than that derived by Boyer et al. (2002) from extrapolation of wet-deposition monitoring data (Figure 6.4; Howarth et al. 2006). A similar, more recent emission-based model (TM3) developed by Frank

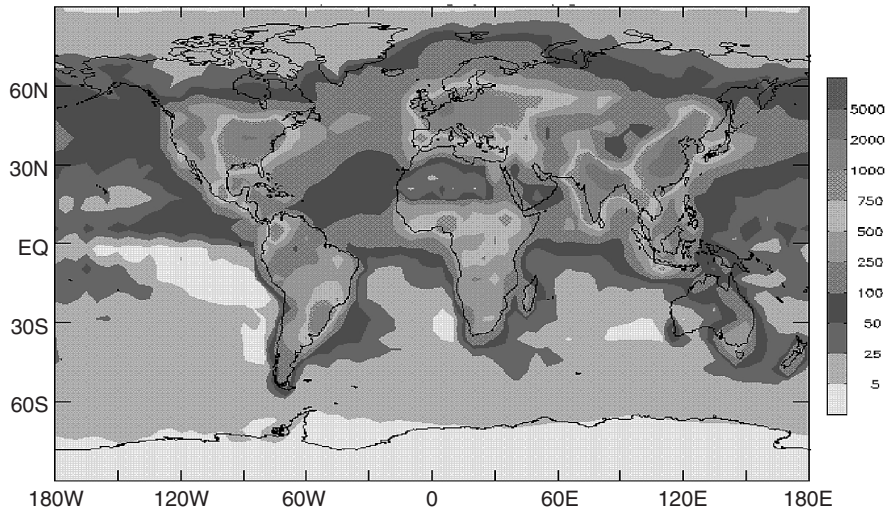

Figure 6.3. Pattern of average total deposition of inorganic nitrogen as of the early 1990s based on the TM3 model of Frank Dentener. Units are kg N km^{-2} yr^{-1}. Modified from a color graphic in Galloway et al. (2004)

Dentener and colleagues and used by Galloway et al. (2004) for their global and regional nitrogen budgets yields a comparable estimate for the northeastern United States as did the GCTM model (Howarth et al. 2006). These emission-based models are attractive, in that at least at very course spatial scales, they are as accurate as the emission data. However, they cannot easily be applied at a spatial scale fine enough to give estimates for the 16 individual northeastern watersheds.

Why is the estimate from the emission-based model (Howarth et al. 1996) so much greater than that from estimates based on extrapolation of the wet deposition monitoring data (Boyer et al. 2002)? There are three possible explanations, which are not mutually exclusive:

1. Deposition on the relatively urbanized coastal plain may be much greater than in the watersheds away from the coast. The watershed areas considered by Boyer et al. (2002) end upriver from the coast and tend to be more rural than is the coastal plain downstream (see Figure 6.4). Recent studies have found evidence that deposition near emission sources can be much greater than deposition away from emission sources; deposition within New York City was more than twice as high than in more rural areas to the North of the city (Lovett et al. 2000), and deposition in the immediate vicinity of

roads was much higher than a few hundred meters away (Cape et al. 2004; unpublished data of Howarth, Bettez, Marino, and Davidson);
2. The estimate based on wet deposition monitoring data (Boyer et al. 2002) may underestimate total deposition. This is of course likely, to the extent that dry deposition is underestimated. As noted above, not all of the important gases that may be deposited are routinely measured by the dry deposition monitoring networks, and depositional velocities may be underestimated in regions with major terrain features. Further, these networks are not designed to measure deposition in the immediate vicinity of emission sources (point #1, above). In fact, most of the NADP wet deposition monitoring sites are intentionally located far away from urban emission sources ; and
3. The estimate from emission-based modeling (Howarth et al. 1996) may overestimate total deposition. This could occur if emissions are overestimated, which may well be true for ammonia emissions but probably not for emissions of oxidized nitrogen to the atmosphere in

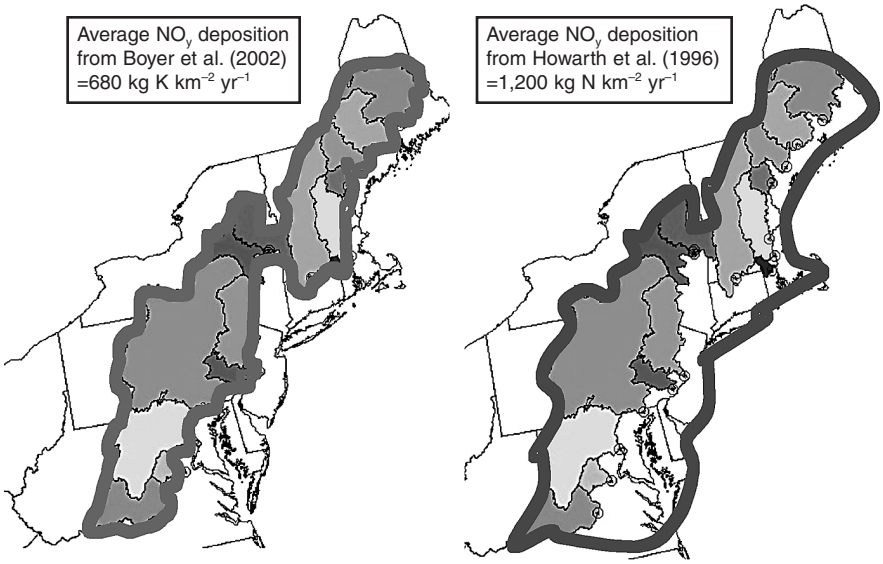

Figure 6.4. The geographic area considered by Boyer et al. (2002) was the area of 16 watersheds in the northeastern United States upriver from the lowest gaging station of the USGS (left). The area considered by Howarth et al. (1996) is somewhat larger, and includes the area on the coastal plain (right). Note that the average estimates for deposition of oxidized nitrogen pollution originating from fossil fuel combustion is ~ 80% greater in the Howarth et al. (1996) analysis, probably due to different approaches used for the estimation and/or the different area considered

the United States (Holland et al. 1999). The difference between the Howarth et al. (1996) and Boyer et al. (2002) estimates highlighted in this chapter is for deposition of oxidized nitrogen. Alternatively, the emission-based modeling may not accurately capture the spatial pattern of the deposition. These models rely on a mass balance of nitrogen in the atmosphere, so global deposition estimates are as accurate as the emissions data that feed them. However, deposition may be underestimated in some regions and correspondingly overestimated elsewhere.

Obviously, significant uncertainty exists in the overall magnitude of total nitrogen deposition in an area such as the northeastern United States. When considering points #1 and #2 above, it is important to note that extrapolations based on wet-depositional monitoring (Ollinger et al. 1993; Grimm and Lynch 2005) do not appear to capture any evidence of higher deposition near urban centers and transportation corridors. For reasons discussed further below, I believe it likely that these traditional approaches that use NADP wet deposition monitoring data to estimate total nitrogen deposition result in substantial underestimates, especially for total nitrogen deposition in the urbanized portions of the northeastern United States.

4. Throughfall as an Approach for Estimating Total Nitrogen Deposition

The difficulty with measuring dry deposition of N (particularly of gaseous forms such as NO, NO_2, and NH_3) has led some investigators to use tree-canopy throughfall as a surrogate for total N deposition (Lajtha et al. 1995; Lovett et al. 2000; Weathers et al. 2005; Schmitt et al. 2005). Throughfall is the material that falls through the canopy of a forest, and so includes whatever is deposited on the canopy in both wet and dry deposition, plus the net exchange of material with the vegetation. Most studies have found that the assimilation of nitrogen from deposition into leaves of the canopy is generally as great as or greater than the leaching of nitrogen out of leaves (Lindberg et al. 1990; Johnson 1992; Lovett and Lindberg 1993; Dise and Wright 1995; Lajtha et al. 1995). Consequently, many experts on atmospheric deposition have argued that throughfall measurements provide a minimum estimate of total nitrogen deposition (Lindberg et al. 1990; Johnson 1992; Lovett and Lindberg 1993; Dise and Wright 1995; Lajtha et al. 1995; Lovett et al. 2000; Schmitt et al. 2005).

The estimation of total nitrogen deposition from throughfall measurements often yields much higher rates than those inferred from wet deposi-

tion data. For example, in a forest in Falmouth, MA, on Cape Cod, Lajtha et al. (1995) measured wet deposition of 420 kg N km^{-2} yr^{-1} and estimated a total deposition rate of 840 kg N km^{-2} yr^{-1} by assuming that dry deposition equaled wet deposition. This estimate is quite similar to the deposition predicted for that location by the spatial extrapolation of Ollinger et al. (1993). However, from their throughfall data, Lajtha et al. (1995) estimated that actual total nitrogen deposition at the site was 1,310 kg N km^{-2} yr^{-1}, or more than 50% greater. In a more recent study, Weathers et al. (2005) compared throughfall data with more traditional approaches for estimating nitrogen deposition in the Acadia National Park in Maine and in the Great Smoky Mountains National Park in North Carolina. In both locations, they found that total nitrogen deposition rates estimated from their throughfall data were 70% greater than those estimated from NADP and CASTNet wet and dry monitoring data. These throughfall estimates lend strength to the argument that the traditional approaches for estimating total deposition—such as we used in Boyer et al. (2002)—yield low values.

5. What is the Fate of Nitrogen Deposited onto the Landscape?

Forests are the dominant land cover in the northeastern United States (Boyer et al. 2002), and so much of the nitrogen deposition on the landscape falls on forests. Only a portion of the nitrogen is exported downstream, and much is retained in the forests or denitrified (converted to non-reactive, molecular N_2). Productivity of most forests in the United States is limited by the supply of nitrogen (Vitousek and Howarth 1991), so as forests receive more nitrogen from atmospheric deposition, production and storage of nitrogen in organic matter can be expected to increase. On average for the northeastern United States, approximately 60% to 65% of the nitrogen inputs to forests through natural nitrogen fixation as well as atmospheric deposition are retained in the forest (primarily accreted in woody biomass) or harvested from the forests in wood (Goodale et al. 2002; van Breemen et al. 2002). A little over 20% is exported from the forest in streams (primarily nitrate, but also dissolved organic nitrogen), with the rest denitrified (van Breemen et al. 2002). The ability of forests to store nitrogen, however, is limited, and forests can become nitrogen saturated when inputs exceed the needs of trees and the ability for soils to assimilate nitrogen (Aber et al. 1989; Gundersen and Bashkin 1994; Emmett et al. 1998). Nitrogen export downstream can then increase dramatically (Emmet et al. 1998; Howarth et al. 2002b; Aber et al. 2003).

A recent comparative study suggests that for the forests of northern New England, the nitrate concentrations in streams and small lakes just downstream increase dramatically as total nitrogen deposition increases above 600 to 800 kg N km^{-2} yr^{-1} (Figure 6.5; Aber et al. 2003), indicating a substantial increase in nitrogen export from the forests receiving the higher deposition. Figure 6.5 indicates the estimated average NOy deposition for the northeastern United States in the Boyer et al. (2002) and Howarth et al. (1996) studies. Note that total deposition, including ammonia, ammonium, and organic nitrogen, would be greater by 20 to 40% (Boyer et al. 2002; Howarth et al. 1996), but is also much more uncertain (Holland et al. 1999; Howarth et al. 2006), so I have chosen to illustrate just the NOy component. Note also that the deposition estimates used in the Aber et al. (2003) analysis may also be low, since these are based on extrapolation of wet-monitoring data. On the other hand, all of the data in the analysis of Aber et al. (2003) are from fairly rural sites, relatively far from emission sources;

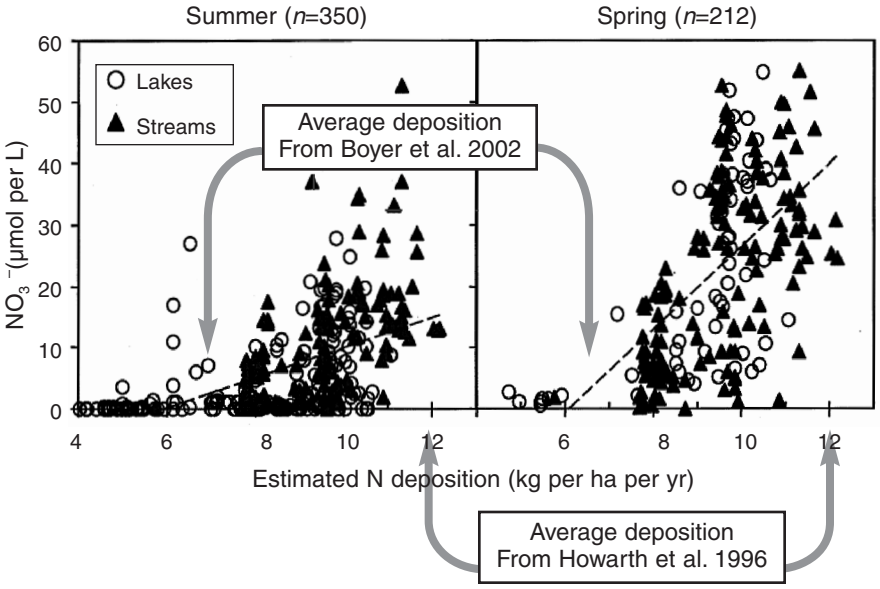

Figure 6.5. Concentrations of nitrate in small streams and lakes in forested catchments in northern New England in the spring (right) and summer (left) as a function of NOy deposition onto the landscape. Note the non-linear response, with nitrate concentrations tending to increase as deposition exceeds 6 to 8 kg N per hectare per year (600 to 800 kg N km^{-2} yr^{-1}). The arrows indicate the average deposition rates for oxidized nitrogen compounds (NOy) estimated for the northeastern United States in Boyer et al. (2002) and Howarth et al. (1996). Modified from Aber et al. (2003)

their deposition estimates may therefore be fairly reliable. In any case, Figure 6.5 suggests that nitrogen deposition onto the landscape on average in the northeastern United States is likely high enough to result in elevated losses of nitrogen from forests, particularly if the higher emission-based estimates used by Howarth et al. (1996) are valid.

While forests are often retentive of nitrogen, impermeable surfaces such as roads and parking lots are far less so. While not often studied, nitrogen runoff from these surfaces can be substantial. For example, runoff from highways near Providence, RI, is reported to be 1,700 kg N km^{-2} yr^{-1} (Nixon et al. 1995). Most if not all of this nitrogen likely originated from atmospheric deposition.

6. Case Study: How Important is Deposition to the Nitrogen Budget for Chesapeake Bay?

Chesapeake Bay is the largest estuary in the United States, and one of the most sensitive to nutrient inputs (Bricker et al. 1999; National Resource Council 2000). Nitrogen inputs to Chesapeake Bay have caused widespread loss of seagrasses (as they are shaded out by blooms of phytoplankton) and have greatly increased the volume of anoxic bottom waters. The role of atmospheric deposition as a source of nitrogen to the Bay apparently was not considered until Fisher and Oppenheimer (1991) suggested that it might contribute 40% of the total inputs. Their analysis was simple and preliminary, and was not believed by many scientists who worked on water quality in this estuary system. The most recent analyses by the Chesapeake Bay Program, while giving lower percentages, also suggest that deposition is important, contributing ~ 25% of the total nitrogen inputs to Chesapeake Bay (7% from direct deposition onto the surface waters of Chesapeake Bay, and 19% from deposition onto the landscape with subsequent export to Chesapeake Bay, using 2003 values; http://www.chesapeakebay.net/status.cfm?SID=126; see also http://www.chesapeakebay.net/nutr1.htm).

Two lines of evidence suggest that the Bay Program model may be underestimating the inputs of nitrogen from atmospheric deposition: 1) the model may be underestimating the magnitude of deposition onto the landscape; and 2) the model may be underestimating the percentage of deposition onto the landscape that is subsequently exported downstream. Each of these is discussed below.

The Chesapeake Bay Program model relies on an estimate of total nitrogen deposition of 1,210 kg N km^{-2} yr^{-1} (calculated from Figure A-4 Environmental Protection Agency 2003). The approach to derive this estimate is

very similar to that used by Boyer et al. (2002): extrapolation from wet-deposition monitoring data for the 15 NADP sites in the watersheds of the Chesapeake (Lewis Linker, Bay Program modeling coordinator, PowerPoint presentation by conference call, January 9, 2006), although the Boyer et al. (2002) estimate is in fact somewhat lower (1,010 kg N km^{-2} yr^{-1} for the area-weighted mean for the watersheds of the Susquehanna, Potomac, Rappannnock, and James Rivers up river of the USGS gaging stations). If we assume that the Boyer et al. (2002) estimate underestimates by 80% (based on comparison with the global-scale emission-based model used by Howarth et al. 1996), then actual deposition on the Chesapeake watersheds may be as great as 1,550 kg N km^{-2} yr^{-1} (28% greater than assumed for the Chesapeake Bay Program model).

Perhaps of greater significance is the treatment of nitrogen retention in the landscape by the Bay model. The model treats retention as a function of land use, and on average assumes that 86% to 89% of total nitrogen deposition onto the landscape is retained, and only 11% to 14% is exported downstream to the Bay (calculated from Figure A-4, Environmental Protection Agency 2003). Most of this retention is assumed to occur in the 57% of the area of the watershed that is forested, with the export coming from deposition onto agricultural lands and urban and suburban areas with impermeable surfaces. The model assumes that most of the forests in the Chesapeake Bay basin are not nitrogen saturated, and therefore leak little if any nitrogen (Environmental Protection Agency 2003).

The average export of nitrogen deposition from all land uses (12%) seems low in comparison with the estimate that average forests in the northeastern United States export over 20% of nitrogen deposition (Goodale et al. 2002; van Breemen et al. 2002). If the deposition in the Chesapeake basin is evenly distributed over land uses, then 43% falls on other land uses where much higher rates of export would be expected. If much of the deposition from nitrogen pollution that originates from vehicles falls near these emission sources (either onto impermeable surfaces or onto vegetation where the rate of deposition would be very high), then very high rates of export might be expected. Obtaining better data on nitrogen retention in the landscape has been identified as a high national research need in a multi-agency federal planning document (Howarth et al. 2003). But given current knowledge, it is probably as reasonable to assume that the percent export from atmospheric deposition onto the landscape of the Chesapeake Bay basin—including all land uses—is 30% as to assume the 12% used by the Bay model. Ranges from 20% to 40% and even higher can be reasonably inferred from studies of large watersheds (National Research Council 2000; Howarth et al. 2002b, 2006; Boyer et al. 2002).

Table 6.1 illustrates the sensitivity of nitrogen loading to Chesapeake Bay to various assumptions on the rate of deposition and on nitrogen retention in the landscape. Within this range of reasonable assumptions, the total input of nitrogen to the Bay from atmospheric deposition (both directly onto the surface waters of the Bay and indirectly from deposition onto the landscape and subsequent export downstream) ranges from 34 to 92 thousand metric tons of nitrogen per year, and comprises from 25% to 50% of the total nitrogen load to Chesapeake Bay from all sources. Under the assumptions of greater deposition and lower retention in the landscape, the estimate for total nitrogen load to Chesapeake Bay increases substantially – from 130 to 188 thousand metric tons per year, or 45% greater total nitrogen load. Perhaps surprisingly, monitoring of the load of nitrogen to Chesapeake Bay is not adequate to constrain this total load estimate within this range of uncertainty. As with many other large coastal marine ecosystems, significant portions of the watersheds of Chesapeake Bay are not gaged because of the difficulty in gaging tidal streams and rivers (Valigura et al. 2000;

Table 6.1. Importance of atmospheric deposition as a source of nitrogen pollution to Chesapeake Bay under various assumptions. Fluxes are thousands of metric tons of nitrogen per year. Percentage values given in parentheses are percentages of total nitrogen load. The baseline run assumptions are from EPA (2003)

	Total Load to Bay	Input to Bay from Direct Deposition onto Bay Water Surface	Input to Bay from Deposition onto Watersheds	Total Input to Bay from Deposition
Chesapeake Bay model (2000 conditions)	130	9 (7%)	25 (19%)	34 (26%)
Deposition increased to 1,550 kg N km^{-2} yr^{-1}; no change in retention assumptions	140	12 (9%)	32 (23%)	44 (32%)
Chesapeake Bay model assumptions on deposition rate; assume 70% retention in landscape	168	9 (5%)	63 (38%)	72 (43%)
Deposition increased to 1,550 kg N km^{-2} yr^{-1}; assume 70% retention in landscape	188	12 (6%)	80 (43%)	92 (49%)

National Research Council 2000; Howarth et al. 2002b). These areas of the watershed are therefore not monitored for their nutrient inputs to the Bay. While the fluxes of nitrogen from the watersheds above gaging stations in the Chesapeake Basin are reasonably well known, the fluxes from the watershed in the more urbanized areas on the coastal plain near Chesapeake Bay—where nitrogen deposition may be much greater, and retention of nitrogen in the landscape much less—are estimated only from models and not from empirical monitoring data.

7. Steps Towards Effective Management

Despite the widespread damage to coastal waters from nitrogen pollution, for the most part governments have been slow to systematically apply effective policies for controlling this problem, in the United States or elsewhere (NRC 2000; Howarth et al. 2005). The reasons for this policy failure are many, but one major reason is that management of eutrophication or nutrient pollution generally has focused on phosphorus rather than nitrogen since the early 1970s (Howarth and Marino 2006; Howarth et al. 2005). While this is appropriate for freshwater lakes, nitrogen is the larger problem in most coastal marine ecosystems (National Research Council 2000; Howarth and Marino 2006). Although some local or regional agencies have addressed nitrogen pollution in coastal waters over the past two decades, even today no national standards for coastal nitrogen pollution exist (National Research Council 2000; Howarth et al. 2005). Scientific evidence for the necessity of phosphorus control on eutrophication in freshwater lakes and nitrogen control in coastal marine ecosystems has steadily accumulated for many decades, but only in the past 5–10 years has this evidence begun to be fully accepted by water quality managers. Even when managers have recognized that nitrogen is the prime cause of eutrophication in coastal rivers and bays, management practices for non-point sources of nitrogen often have remained focused on those proven effective for managing phosphorus pollution, with insufficient recognition that other practices may be needed for nitrogen because of its much greater mobility in groundwater and through the atmosphere (National Research Council 2000; Howarth et al. 2005; Howarth 2005).

Both fossil fuel combustion and agricultural practices contribute significantly to atmospheric fluxes of nitrogen but not phosphorus. The magnitude of the contribution of these atmospheric fluxes to coastal nutrient pollution remains uncertain, and understudied. Nonetheless, atmospheric deposition is clearly an important contributor to coastal nutrient pollution. This source

demands more attention by water quality managers if the goal of reducing coastal nutrient pollution is to be met (National Research Council 2000).

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