4
Metrics and Implications for Risk Reduction Strategies for Reactive Nitrogen

It is important to develop risk reduction strategies for reactive nitrogen that take into consideration the ways in which Nr is introduced and transformed in the environment. This chapter reviews current and historical measurement and risk reduction activities for Nr and provides specific Committee findings and recommendations.

4.1. Measurement of Nr in the Environment

Although nitrogen is among the most abundant elements on earth, only a small fraction, Nr, is responsible for impacts on the environment. Most regulations focus narrowly on specific chemical forms of nitrogen as they affect media- or site-specific problems, setting limits or specifying control technologies without regard to the ways in which N is transformed once introduced into the environment. Measurement methods are typically expressed in terms of mass loadings or concentrations of a particular form of N (e.g., ppm NO_x, mg/L total NH_x, or kg/ha of NO_3^-).

Finding 13: The Committee finds that there is a need to measure, compute, and report the total amount of Nr present in impacted systems in appropriate units. What is measured influences what we are able to perceive and respond to; in the case of Nr, it is especially critical to measure total amounts and different chemical forms, at regular intervals over time.

Recommendation 13: *The Committee recommends that EPA routinely and consistently account for the presence of Nr in the environment in forms appropriate to the medium in which they occur (air, land, and water) and that accounting documents be produced and published periodically (for example, in a fashion similar to National Atmospheric Deposition Program [NADP] summary reports). The Committee understands that such an undertaking will require substantial resources, and encourages the Agency to develop and strengthen partnerships with appropriate federal and state agencies and private-sector organizations having parallel interests in advancing the necessary underlying science of Nr creation, transport and transformation, impacts, and management.*

4.2. Consideration of Nr Impacts in Risk Reduction Strategies

Historical measurement and impact categories

The types of impacts of Nr in the environment are dependent on three general factors: the sources of Nr, the types of media impacted, and Nr chemical forms. The magnitude of effects depends on loading and the nature of the system impacted. As illustrated in Figure 1, the impacts of a given source of Nr can be multiple as N is transformed in the environment and transported among ecosystem components. The nitrogen cascade provides a comprehensive framework for understanding the role of Nr in the earth's ecosystems and establishes a framework for developing and implementing management methods through which beneficial effects can be enhanced while minimizing detrimental impacts.

A management paradigm in which various approaches are used to limit environmental impacts to "acceptable" levels of risk is a useful concept for understanding the environmental impacts that Nr can have. For this purpose, impacts are divided into several general categories within which various contaminants have a direct correlation with damage. Once the nature and type of impacts are recognized, the risks should be characterized quantitatively, if possible. This information would then be used, along with other considerations such as economic, social and legal factors, to reach decisions regarding risk reduction strategies and the need for and practicability of implementing various risk reduction activities. The regulation of Nr in the environment by EPA follows an impact-by-impact approach which, with few exceptions, examines specific N forms in either aquatic, atmospheric, or terrestrial systems. As previously discussed, the principal regulatory authority pertaining to nitrogen is derived from the CWA and the CAA, although other legislation such as the Energy Independence and Security Act (EISA), and the Endangered Species Act (ESA) contain provisions that could result in regulatory actions that affect nitrogen management.

Historically, EPA environmental protection programs have addressed impacts of Nr such as climate change, eutrophication, ecotoxicity, human health (cancer and non-cancer), acidification, smog formation, and stratospheric ozone depletion, among others (Bare et al., 2003). Within these categories it is sometimes possible to express end points in terms of collective metrics, such as is done with greenhouse gases in the form of carbon dioxide equivalents, or acidification as H^+ equivalents. This approach has the considerable advantage of defining a straightforward framework within which environmental standards can be derived that are protective of human health and the environment – EPA's principal mission. This approach also encourages evaluation of damage

from multiple sources, as long as the characterization metric used is genuinely representative of the impact of a given contaminant. Thus, for example, the total impact of acidic gases such as SO_2 and NO_x on the acidification of watersheds can be expressed as a common metric. However, metrics for human health are generally not as simple to characterize nor are the appropriate end points; thus, the mechanism of toxicity, number of individuals affected, value of lost workdays, medical treatment costs, and value of human lives lost may all be used.

Ecosystem functions and services

A complementary approach to classical impact characterizations is the use of ecosystem "service" and "function" categories, in which the impairment of a specific service provided by one or more ecosystems or impairment of an ecological function by causative contaminant emissions is assessed (Costanza, 1997; Millennium Ecosystem Assessment, 2003). Such an approach is inherently attractive because of its basis in scientific reality, i.e., the health of humans is inextricably linked to the health of the environment. Less clear, in some cases, are ways in which to measure and monitor such impacts and account for the effects of a complex array of factors and stressors that contribute to or damage ecosystem service, function, and health. Table 11 provides examples of ecosystem services and corresponding functions.

The use of ecosystem services in a regulatory context would be a different approach for the EPA, one with considerable potential, but one for which experience is currently lacking. In comparison to the available data on reactive nitrogen usage, little is known about the response of ecosystems and ecosystem services to reactive nitrogen loads. This is discussed more fully in Sections 4.5, 4.6 and Appendix D on critical loads. In this context the Committee

Table 11: Ecosystem service and corresponding function categories

Ecosystem Service	Ecosystem Function
Gas regulation	Regulation of atmospheric chemical composition
Climate regulation	Regulation of global temperature, precipitation, and other biologically mediated climatic processes at global, regional, and local levels
Disturbance regulation	Capacitance, damping, and integrity of ecosystem response to environmental fluctuations
Water regulation	Regulation of hydrologic flows
Water supply	Storage and retention of water
Erosion control and sediment retention	Retention of soil within an ecosystem
Soil formation	Soil formation processes
Nutrient cycling	Storage, internal cycling, processing, and acquisition of nutrients
Waste treatment	Recovery of mobile nutrients, and removal or breakdown of toxic compounds
Pollination	Movement of floral gametes
Biological control	Trophic dynamic regulation of populations
Refugia	Habitat for resident and transient populations
Food production	That portion of gross primary production extractable as food
Raw materials	That portion of gross primary production extractable as raw materials
Genetic resources	Sources of unique biological materials and products
Recreation	Providing opportunities for recreational activities
Cultural	Providing opportunities for noncommercial uses

Source: Costanza et al., 1997 (Table 1, p. 254). Reprinted with permission; copyright 1997, Nature Publishing Group.

supports plans by the EPA to incorporate research on the services concept, focusing on Nr as the suite of contaminants of interest, into its future ecological research plan (U.S. EPA, 2009a). EPA's Ecological Research Plan was reviewed by the Science Advisory Board (U.S. EPA SAB, 2008). More recently, the Science Advisory Board completed a self-initiated study on "Valuing the Protection of Ecological Systems and Services" (U.S. EPA SAB, 2009). This report explores the concept of ecosystem services as a basis for regulatory action and presents a roadmap for implementing this approach.

Economic measures and impacts

It is also possible to translate the effects of Nr into economic terms. Two economic measures that are often used are the dollar costs of damages and the cost of remediation or substitution. Another important economic metric is the cost/ton of remediation for each form of Nr. Damage costs do not always scale as tons of Nr released into the environment. If damage costs rather than tons of nitrogen were utilized as a metric, the full implications of the cascade and the setting of priorities for intervention might differ.

It is important to note that the choice of metric used in assessing impacts may play an influential role in what and how one manages. Air and water protection laws state that the goal is "to protect human health and the environment." Yet, there is no generally agreed-upon common metric for measuring the full range of effects (which are complex and often unknown) or for setting priorities in the establishment or implementation of policies.

As noted above, there are multiple metrics for measuring Nr or any other agent in the environment. The most common metric utilizes quantitative measures of the total amount of Nr (and any of its specific chemical forms) in different environmental reservoirs and the mass flux between them. But while providing common units, typically mass or concentration, these measures do not distinguish the relative societal costs of health or environmental consequences of reactive nitrogen of different forms or places in the cascade. While not all damages can be turned into economic costs, and the costs of some damages have not been quantified, enough of the major damages can be quantified economically to provide a useful complementary metric for decision-making. (See the Chesapeake Bay example in Box 2.)

The advantage of monetizing damages is that it reflects an integrated value that human society places on lost ecosystem goods and services in common currency and illustrates the cascading costs of damages as Nr changes form and moves between different parts of the ecosystem. In addition, human health implications can also be included as the cost of health care treatment, lost work days and other aspects of morbidity and mortality (e.g., economic value of lives lost). A third metric is to look at morbidity and/or mortality separately and not monetize

them with a cost value. Of course a concern, particularly with respect to the economic metric, is that there are a number of ecosystem services that arguably cannot be easily monetized, for example the loss of biodiversity and those ecosystem functions that are affected by climate change or other stressors. Ecosystem services considered to be regulating and supporting are particularly difficult to fit into an economic metric. It is thus essential that a variety of complementary metrics be used to assess the impact of anthropogenic Nr on the environment and human well being.

There is value in each of the ways that N metrics are expressed. Traditional categories provide a readily adaptable framework for regulation, while ecosystem service and function-based categories provide a richer context for stating the complex connections among Nr inputs and transformations and their impacts on ecosystem health and human well-being. Dollar-based metrics provide a means of identifying those effects that have the greatest impacts and costs to society.

Finding 14: The Committee finds that reliance on only one approach for categorizing the measurement of Nr is unlikely to result in the desired outcome of translating N-induced degradation into the level of understanding needed to develop support for implementing effective Nr management strategies.

Recommendation 14: *It is, therefore, recommended that the EPA consider the impact of different metrics and examine the full range of traditional and ecosystem response categories, including economic and ecosystem services, as a basis for expressing Nr impacts in the environment, and for building better understanding and support for integrated management efforts.*

4.3. Water Quality Regulation and Management

Aquatic thresholds

In aquatic ecosystems, thresholds at which excess Nr becomes a problem can be expressed as a management goal such as a total maximum daily load (TMDL) or as a critical load (CL). Under the authority of the CWA, EPA has developed guidance for establishing numeric nutrient criteria on an eco-regional basis for lakes and reservoirs, streams and rivers, estuaries and coastal waters, and wetlands. EPA has proposed specific numbers for lakes and reservoirs and rivers and streams and protocols for developing criteria for estuaries and wetlands. Each state is advised to go through an assessment to determine the best methodology for implementing numeric criteria (U.S EPA, 2000c, 2000c, 2001b, 2007e). These criteria will identify impaired waterbodies for which TMDLs may be required.

The second type of threshold available for aquatic ecosystems is the critical load (CL). Unlike the TMDL, the CL (in the U.S.) has no regulatory framework but rather sets the threshold of Nr loading at which negative

Recently, the N cycle and the implications of the reactive nitrogen cascade were translated into economic terms for the case of Chesapeake Bay (Moomaw and Birch, 2005). This approach has recently been updated with more recent data, and the economic and health impacts of different forms of Nr in multiple ecosystems and media have been estimated using better modeling methods (Birch et al., 2011). As an illustration, each of these metrics is shown as a percentage of Nr fluxes in the Chesapeake Bay water and air shed in Figure 15. Abatement costs are summarized in Table 12. Atmospheric emissions account for 37% of Nr entering the watershed, but they account for 75% of the dollar damages and 76% of the mortality (U.S. EPA, 2005c). Mitigation costs per tonne of atmospherically released Nr are the lowest among the three sources. Additions of Nr to terrestrial ecosystems add 60% to the system, but contribute only 24% of the damage costs and 24% of the mortality, and have the highest mitigation costs. Freshwater releases, the second most expensive to mitigate, account for the smallest portion of Nr contributions to the system by any of the metrics considered: only 4% of the Nr, 2% of the cost damages, and none of the mortality losses (Birch et al., 2011). Costs of Nr damage and health metrics provide additional economic measures of the cost effectiveness of actions to reduce a metric ton of Nr.

The metrics of damage cost and mortality (morbidity shows a similar pattern to mortality, but is only one-tenth the damage cost) indicate that controlling emissions of NO_x from combustion and industrial processes produces greater gains in protecting human health and the environment than does reducing other Nr releases, though the two sources are comparable in terms of reactive forms of Nr released to the watershed. This difference occurs because emissions to the air cascade through more parts of the watershed ecosystem than releases directly to the Bay. If human health effects are monetized, then the economic gains are even greater from reducing atmospheric emissions (see Figure 16).

Figure 16 is a scatter plot of all quantifiable damage costs (including health impacts) relative to metric tons of Nr showing the significant difference in emphasis of the two metrics. Note that direct additions to the environment from agriculture are about 370,000 tonnes Nr/year, and cause $1.7 billion worth of damage. Emissions of NO_x from mobile sources represent only 180,000 metric tons Nr/year but cause nearly $4.4 billion in damages each year, of which $108 million is attributable to nitrate loading of the Chesapeake Bay, $3.9 billion to human morbidity and mortality, and the remainder to other forms of damage, such as crop and commercial forest damage. Hence the releases of Nr into the airshed from mobile sources, which are only half the amount of agricultural releases to the watershed, cause more than 2.5 times the economic damage of environmental additions from agriculture. This integrated inclusion of atmospheric, terrestrial, and aquatic additions of Nr is not reflected in today's regulations

Marginal abatement costs per metric ton Nr by source into each of the three media are provided in Table 12, and demonstrate that the least costly abatement cost per metric ton of Nr also comes from atmospheric emission controls. While most legislation constrains how cost for remediation can be considered, it is useful to know where the lowest cost options lie in setting priorities.
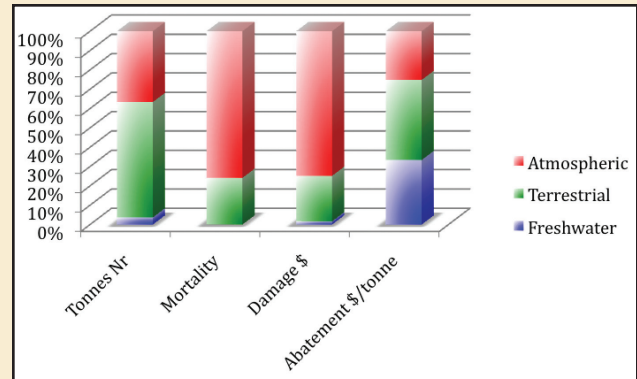

Figure 15: Relative importance of all reactive nitrogen sources released into atmospheric, terrestrial, and freshwater media within the Chesapeake Bay Watershed utilizing four different metrics

Source: Birch et al., 2011 (Figure 3, p. 173). Reprinted with permission; copyright 2011, American Chemical Society.
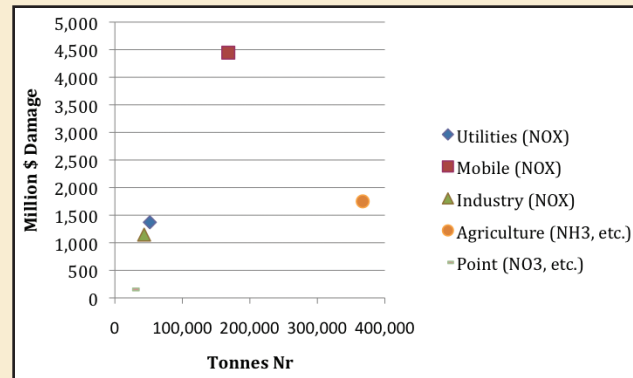

Figure 16: Quantified damage costs (including health impacts) relative to metric tons of reactive nitrogen in Chesapeake Bay Watershed

Source: Birch et al., 2011 (Figure 4, p. 173). Reprinted with permission; copyright 2011, American Chemical Society.

These multiple metrics provide several ways of looking at the nitrogen cascade and its impact on human health and the environment. However, there are many impacts that remain unaccounted for in any of these metrics. Some impacts might be quantified, but the necessary data have yet to be collected. Economic losses due to damage to commercial fisheries in the Bay are an example that is likely to be significant but has not yet been quantified. Similarly, economic losses due to climate change and ozone depletion from N_2O emissions have not been fully evaluated. Impacts such as loss of biodiversity cannot be readily quantified at all, so it is desirable to consider a set of qualitative and non-quantified metrics in addition to the quantitative ones.

Other parts of the country such as the Mississippi valley or the Central Valley of California are expected to show very different patterns of cost damages, with terrestrial and freshwater emissions causing proportionally higher damage costs, and emissions to the atmosphere causing a lower percentage of damages. But those very differences would assist EPA and the generators of those emissions in setting priorities for mitigation.

It is important to recognize that Nr is not the only stressor that can affect both human and environmental health. Researchers are challenged to comprehensively understand cause-and-effect relationships in a complex environment and to balance management actions and costs to ensure that risk-minimizing management strategies are effectively implemented.

As these multiple metrics indicate, decisions about which fluxes of Nr to mitigate depend upon which metric is utilized. The cascading economic costs of damage highlight the importance of regulating air emissions because of their impacts on human heath as well as their large contribution to the degradation of Chesapeake Bay water quality. Hence, if one is interested in reducing water impacts of Nr, the total reduction of damage may rely nearly as much on stricter enforcement of the Clean Air Act as the Clean Water Act. This challenges our traditional approach to regulation, but that is a consequence of comprehensively examining Nr guided by the nitrogen cascade.

Table 12: Marginal abatement cost per tonne of Nr by source

Location in the N cascade where emitted	Source/pollutant	Abatement cost per tonne of Nr
Air	Electric utilities/NO_x [14]	$4,800
	Industrial/NO_x [15]	$22,000
	Mobile sources/ NO_x [16]	$14,000
	Non-agricultural/NH_3	No estimate
Land	Agriculture/nitrate [17]	$10,000
	Urban and mixed open land uses/nitrate [17]	$96,000
Fresh water	Point sources/nitrates [17]	$18,000

impacts have been documented. Based extensively on European work, CLs for aquatic ecosystems are Nr inputs on the order of 2-15 kg N/ha/yr (Bobbink et al., 2010). There are numerous locations within the U.S. where deposition to surface waters falls within this range.

Water quality standards

Section 303 of the CWA requires states to adopt water quality standards and criteria that meet the state-identified designated uses (e.g., uses related to "fishable" and "swimmable") for each waterbody. Specifically, "a water quality standard defines the water quality goals of a water body, or portion thereof, by designating the use or uses to be made of the water and by setting criteria necessary to protect the uses" (40 CFR § 131.2). Further, "such standards serve the dual purposes of establishing the water quality goals for a specific water body and serve

[14] See U.S. EPA, 2005c
[15] See U.S. EPA, 1998
[16] See Krupnick et al., 1998
[17] See Chesapeake Bay Program, 2003a,b

as the regulatory basis for the establishment of water quality-based treatment controls and strategies beyond the technology-based levels of treatment required by sections 301(b) and 306 of the Act" (40 CFR § 131.2).

The EPA sets minimum requirements for approvable standards and criteria including: use designations; water quality criteria sufficient to protect the designated uses; and an antidigradation policy (40 CFR § 131.6). Traditionally, Nr and other land, air, and water pollutants are measured in terms of quantity (mass) released per unit time (e.g., kg/day) or as a concentration (e.g., milligrams per liter, ml/L). Therefore, regulations often specify mass loading limits or maximum concentrations in permits.

In the mid-to-late 1990s, EPA began to emphasize the development of numeric nutrient criteria for both P and N through the state standards-setting process because, according to the 1996 Water Quality Report to Congress (U.S.EPA, 1997), 40% of the rivers, 51% of the lakes and ponds, and 57% of the estuaries assessed for the report were exhibiting a nutrient-related impairment. Few states had adopted numeric nutrient criteria for all affected waterbodies, especially for N, often relying on narrative criteria or secondary effects such as chlorophyll-*a* concentration, dissolved O_2, or water clarity. EPA's strategy, driven by President Clinton's Clean Water Action Plan (U.S. EPA and USDA, 1998) mandated numeric nutrient criteria to begin to address the problem (U.S. EPA, 1999). To move the objectives of the Clean Water Action Plan forward, EPA published national nutrient criteria guidance for lakes and reservoirs (U.S. EPA, 2000d), rivers and streams (U.S. EPA, 2000b), estuaries and coastal waters (U.S. EPA, 2001b), and wetlands (U.S. EPA, 2007c), based upon ecoregional guidance for lakes and reservoirs and rivers and streams. To date, relatively few states have adopted new numeric criteria into their water quality standards. While some successes are evident in promulgating P criteria for freshwater systems, which has a richer history of numeric criteria incorporation into state water quality standards, development of numeric nitrogen criteria has been elusive for a variety of reasons.

Nr management in multiple media and across jurisdictions can be complicated because the CWA has little authority over atmospheric sources, and individual states explicitly lack authority to control upstream sources. For example, extensive monitoring and analysis of the sources of reactive nitrogen in the Raccoon River of western Iowa have shown that point sources from municipal treatment plants and residential septic tanks account for less than 8% of the total nitrogen load to the system, with agricultural runoff being the overwhelming source (Jha et al., 2010). This disparity is similar statewide (Libra et al., 2004). As a result, nutrient management strategies that are focused on the control of point sources can often result in inefficient allocation of resources if non-point sources are not also addressed.

In addition it is often the case for estuaries such as the Gulf of Mexico or Chesapeake Bay, that management goals that meet water quality standards cannot be attained without interstate compacts or a strong federal role. This may be resisted by upstream states that may have to bear the cost but do not necessarily reap the benefits of the water quality improvement. Such a dilemma underscores the need for an integrated approach to Nr management. The Committee notes that a State-EPA Nutrient Innovations Task Group has considered some options for improving control of nutrient pollution sources (State-EPA Nutrient Innovations Task Group, 2009).

Populated (urban/suburban/developed) land areas provide significant loads of Nr to the environment, both by generation (e.g., deposition of NO_x emissions) and by transfer (e.g., domestic sewage from food imported into the watershed). Categorical sources include sewage treatment plants (STPs), industries, subsurface (septic) systems, atmospheric deposition, domestic animal and wildlife waste, and fertilizers used on lawns, gardens and landscapes. Infrastructure (e.g., storm sewers) and landscape conditions (e.g., increased impervious cover) more efficiently move Nr associated with surface runoff to receiving waters and may also inject or infiltrate Nr into ground water. Landscape changes, primarily increases in impervious cover, soil disturbance and compaction, and wetland/hydric soil losses, have also reduced the capacity for natural systems to treat Nr inputs by recycling or denitrification. Other disruptions in chemical condition (e.g., acidification), biology (e.g., vegetative cover), and physical character (e.g., temperature increase) alter the nitrogen cascade, which may have both negative and positive consequences for Nr amelioration on the populated landscape and in air and water. Populated lands are estimated to export as much as 10 times the total nitrogen that was exported under pre-development conditions.

Finding 15: Intervention to control Nr under most water management programs generally occurs in three ways:

- Prevention or source controls.

- Physical, chemical, or biological "dead ending" or storage within landscape compartments where it is rendered less harmful (e.g., long-term storage in soils or vegetation; denitrification, primarily in wetlands; reuse).

- Treatment using engineered systems such as wastewater treatment plants or BMPs for stormwater and nonpoint source runoff.

While most management programs focus on the third (treatment) approach, there are opportunities for combining the three that can be more effective and cost less. Furthermore, it is important to recognize that in some cases total reduction of water impacts of Nr may rely nearly as much on stricter enforcement of the Clean Air Act as the Clean Water Act.

Recommendation 15: *To better address Nr runoff and discharges from the peopled landscape the Committee recommends that EPA:*

15a.

- *Evaluate the suite of regulatory and non-regulatory tools used to manage Nr in populated areas from nonpoint sources, stormwater and domestic sewage, and industrial wastewater treatment facilities, including goal-setting through water quality standards and criteria.*
- *Determine the most effective regulatory and voluntary mechanisms to apply to each source type (recognizing that in some cases total reduction of the impacts of Nr may rely nearly as much on stricter enforcement of the Clean Air Act as the Clean Water Act) with special attention to the need to regulate nonpoint source and related land use practices.*

15b.

- *Review current regulatory practices for point sources, including both wastewater treatment plants and stormwater, to determine adequacy and capacity towards meeting national Nr management goals.*
- *Consider technology limitations, multiple pollutant benefits, and funding mechanisms as well as potential impacts on climate change from energy use and greenhouse gas emissions, including nitrous oxide.*

15c.

- *Set Nr management goals on a regional/local basis, as appropriate, to ensure most effective use of limited management dollars.*
- *Fully consider "green" management practices such as low- impact development and conservation measures that preserve or re-establish Nr-removing features to the landscape as part of an integrated management strategy, along with traditional engineered best management practices.*

15d.

- *Research best management practices that are effective in controlling Nr, especially for nonpoint and stormwater sources, including land and landscape feature preservation and set Nr management targets that realistically reflect these management and preservation capacities.*
- *Construct a decision framework to assess and determine implementation actions consistent with management goals.*

15e.

- *In cooperation with the Departments of Agriculture and Army, the Fish and Wildlife Service and the Federal Emergency Management Agency, the EPA should develop programs to encourage wetland restoration and creation with strategic placement of these wetlands where reactive nitrogen is highest in ditches, streams, and rivers. The Agency should also address the means of financing, governance, monitoring, and verification. Such programs might be modeled on the Conservation Reserve Program or extant water quality and*

environmental trading programs, but need not be limited to current practices (as discussed in section 5.3.4).

4.4. Water Quality Monitoring and Assessment

Under Section 106 of the CWA, the EPA provides funds to assist state and interstate agencies and tribes to conduct monitoring of the nation's waters to ensure adopted water quality criteria and designated uses are met. Further, primarily under Section 305(b) of the CWA, those entities are required to report, on a biennial basis, on the health and status of their jurisdictional waters. These assessments are presented by the states to the EPA to categorize attainment of designated uses. EPA published these reports up until 1998 (U.S. EPA, 2000a), after which it transitioned into a Water Quality Report in 2000 (U.S. EPA, 2002) and a National Assessment Database in 2002 (U.S. EPA, 2010c). States also prepare a list of "impaired" waters under Section 303(d) of the CWA and EPA develops a synthesis of the CWA Section 305(b) and 303(d) reporting under a Consolidated Assessment and Listing Methodology (CALM) approach.

As discussed above, the EPA compiles the approved state 303(d) lists into a national listing (U.S. EPA, 2010e). The list provides information by state as well as by impairment cause, and identifies the TMDLs completed to date. The most current data available on the EPA Web site includes reporting from most entities through 2008. The report identifies 6,816 impairments related to "nutrients" (almost 9% of all identified impairments), although other impairments may ultimately have a nutrient enrichment cause. For example, organic enrichment/oxygen depletion (6,410), turbidity (3,046), noxious aquatic plants (981), algal growth (539), and ammonia (general toxicity 356), can all have a common cause such as N or P enrichment. It should also be clear that impairments may have multiple causes so, for example, waters identified as impaired by O_2 depletion may also be impaired by nutrients.

There are other initiatives promoted by EPA to monitor and assess the nation's waters, generally implemented in collaboration with, or by, the state and interstate agencies and tribes having jurisdiction over the waters. These include the Wadeable Stream Assessment (U.S. EPA, 2006c), the National Coastal Assessment and its National Coastal Condition Reports (U.S. EPA, 2001a, 2004a, 2006b), the Survey of the Nation's Lakes and Survey of the Nation's Rivers and Streams, and more recently, probabilistic monitoring efforts in lakes, streams, and estuaries (U.S. EPA, 2010d). Many of these are aimed at including a biological assessment component that is often lacking in water pollutant and chemistry efforts described above.

The USGS collects data on surface and underground waters and disseminates these data to the public, state and local governments, public and private utilities, and other

federal agencies involved with managing water resources. The Committee encourages EPA to work closely with USGS on monitoring and assessment activities.

The National Oceanic and Atmospheric Administration has periodically produced estuarine assessments under the National Estuarine Eutrophication Assessment (NEEA) program. The most recent report was released in 2007 (Bricker et al., 2007). The report has a focus on nutrient enrichment and its manifestations in the estuarine environment and relies on participation and interviews of local experts to provide data for the assessment. Among the key findings were:

- Eutrophication is a widespread problem, with the majority of assessed estuaries showing signs of eutrophication – 65% of the assessed systems, representing 78% of assessed estuarine area, had moderate to high overall eutrophic conditions.

- The most common symptoms of eutrophication were high spatial coverage and frequency of elevated chlorophyll-*a* (phytoplankton) – 50% of the assessed estuaries, representing 72% of assessed area, had excessive chlorophyll-a ratings.

4.5. Clean Air Act and Air Quality Regulation and Management

The modern history of American air pollution control legislation begins with the 1963 Clean Air Act (CAA)

which, along with its amendments, requires the EPA to establish and revise National Ambient Air Quality Standards (NAAQS) and to prepare state of the science reviews such as the Criteria Documents and more recently the Integrated Science Assessments (ISA) (U.S. EPA, 2005a, 2006a, 2007a). There are six criteria pollutants: carbon monoxide, lead, NO_2, ozone, SO_2, and PM. These have been determined to endanger public health or welfare. The CAA as currently written requires a review of the scientific criteria for these standards at five-year intervals. Although NO_2 is the only Nr compound specified as a criteria pollutant, NH_x and NO_y play a major role in formation of the secondary pollutants ozone and particulate matter.

The CAA has been amended several times since its inception. In 1970, the CAA was amended "to provide for a more effective program to improve the quality of the nation's air." The CAA was amended again in 1977, primarily to mandate reductions of emissions from automobiles. Despite evidence that NO_x is the central pollutant in photochemical smog formation (Chameides and Walker, 1973; Crutzen, 1973, 1974; Fishman and Crutzen, 1978; Fishman, et al., 1979), federal regulations did not require automobiles to control NO_x emissions to below 1 g/mi (0.14 g N per km) until 1981. Few locales violate the standards for NO_2,[18] but the secondary effects of several of these gases also pose health and welfare concerns. If a city had an annual average NO_2 level

Table 13: Federal primary ambient air quality standards that involve Nr, effective February 2010.

Pollutant	Federal Primary Standard (NAAQS)
Ozone (O$_3$) 1-hr average 8-hr average	0.12 ppmv 0.08 ppmv
Nitrogen Dioxide (NO$_2$) 1-hr average Annual average	100 ppb 0.053 ppmv (100 µg/m^3)
Particulate Matter, coarse (PM$_{10}$) Diameter ≤ 10 µm, 24-hr average Annual average	150 µg/m^3 50 µg/m$^-$
Particulate Matter, fine (PM$_{2.5}$) Diameter ≤ 2.5 µm, 24-hr average Annual average	35 µg/m^3 15 µg/m^3

Note: Secondary standards are currently identical to the primary standards. Source: www.epa.gov/air/criteria.html

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[18] In 2010, EPA promulgated a new 1-hour standard of 100 ppb for NO_2 [Primary National Ambient Air Quality Standards for Nitrogen Dioxide; Final Rule, *Federal Register* 75 (26): 6474-6537]. Monitoring for compliance with this new standard is required, but it will not be known for several years which if any locales violate this standard.

anywhere near the NAAQS for NO_2, it would risk severe photochemical smog – the summertime efficiency for ozone production ranges from 4 to 10 ppb O_3 per ppb NO_x.

As previously discussed, the focus on compliance monitoring for NO_2 ignores the other, equally important members of the NO_y family such as HNO_3 that deposits quickly onto the earth's surface. It is clear that a causal relationship exists between current levels of N and S deposition and numerous biologically adverse effects on ecosystems across the U.S. (U.S. EPA, 2008d). Conversion of the existing network of NO_x monitors to NO_y monitors with a detection limit of 0.1 ppb would still demonstrate compliance with the NO_2 standard but greatly increase the utility of the measurements for model evaluation as well as for understanding nitrate deposition and formation of photochemical smog, and haze.

Air pollution, especially ozone and PM, continued to be a problem in many American cities and the CAA was again amended in 1990. The Nr-relevant aspects were aimed at controlling urban smog and acid deposition. States were required to develop emissions inventories for reactive organic compounds, carbon monoxide, and NO_x, but not NH_3 or N_2O. Over the U.S., sulfate and nitrate are responsible for about two-thirds and one-third, respectively, of the direct deposition of acids. The CAA Amendment of 1990 required emissions decreases of 10 million tons of SO_2 and 2 million tons of NO_x relative to 1980 levels. Ammonia and ammonium, although they contribute to acidity after entering terrestrial ecosystems (Galloway et al., 2003; NRC, 2003) and are expected to play an increasing role (Pinder et al., 2008), were not regulated by this legislation.

The 1997 revision of the CAA and related regulations changed the standards for ozone and PM (see Table 13). A sizable fraction of the mass of PM less than 2.5 microns, $PM_{2.5}$, is condensed Nr. As stated above, these particles have adverse health consequences. PM is also controlled by the Regional Haze Regulations (40 CFR 51). These regulations require that by the year 2064, states must restore Class I areas defined in the regulations to their natural levels of atmospheric clarity.

Ozone and PM, the two most recalcitrant of the criteria pollutants, cover large spatial scales. All of the ozone and much of PM are secondary pollutants in that they are not released at the tailpipe but form in the atmosphere. Ample evidence shows that much or most of the PM in American cities is secondary (e.g., Donahue et al., 2009). Violations are declared on urban scales, responsibility for their control was assigned to states, but the physics and chemistry of smog and haze are regional. In the eastern U.S., ozone episodes often cover several states and involve pollutants emitted in upwind states that do not themselves experience violations (Husar et al., 1977; Logan 1989; Moy et al., 1994; Ryan et al. 1998). The 1990 amendments to the Clean Air Act authorized, in part as a response to this scaling problem, the Ozone Transport Assessment Group (OTAG)

and the Ozone Transport Commission (OTC). These have jurisdiction extending from Washington, D.C. to Maine. Progress has been made on regional control of emissions; the NO_x State Implementation Plan (SIP) call, implemented in 2003 and 2004, has led to measurable improvements in ambient ozone and nitrate levels (Gego et al., 2007; Sickles and Shadwick, 2007a). Experiences with ozone and PM provide a useful demonstration of why it is necessary to develop an integrated approach to management of Nr.

Atmospheric thresholds for Nr

As shown in Table 13 the metric used for safe, upper limits in the atmospheric environment is concentration (in mass per unit volume of air or volume mixing ratios) averaged for a given time period, usually 1 hour, 8 hours, 24 hours, or annually. The thresholds for excess Nr in the atmosphere remain an area of active research. The only Nr compound for which there is currently a NAAQS is NO_2, which may not exceed 0.053 ppm (100 μg/m^3) for the annual arithmetic mean and 100 ppb for the one-hour average. This standard, based on the direct health effects, is certainly inadequate because NO_2 concentrations well below 0.053 ppm lead to concentrations of secondary pollutants well above acceptable levels (i.e., $PM_{2.5}$ and O_3). The NO_2 concentration required to achieve the current 75 ppb ozone standard has not been rigorously established, but it must be well below 0.053 ppm, because information provided by EPA indicates that areas currently in violation of the ozone standard typically have NO_2 concentrations below 0.020 ppm (U.S.EPA, 2010a). The NO_2 concentration required to achieve the current 15 μg/m^3 $PM_{2.5}$ standard is probably also below the 100 μg/m^3 standard for NO_2 because of the role of NO_2 in secondary particulate formation. States in the eastern U.S. are considering substantial additional NO_x emissions reductions in order to comply with the new 8-hour 75 ppb ozone standard. One scenario being tested (G. Aburn, Maryland Department of Environment, personal communication) involves the following reductions: (1) reducing NO_x emissions for point sources by 65%, (2) reducing NO_x emissions for on-road sources by 75 percent, (3) reducing NO_x emissions for nonroad sources by 35%, and (4) reducing VOC emissions by 30% for all source groups.

As further discussed in Section 6.2, it is the opinion of the Committee that a decrease in NO_x emissions of 2 Tg N/yr relative to the 2002 baseline level can be achieved in the near term. Emissions decreases implemented since 2002 have already substantially improved ozone concentrations (Gégo et al., 2007). The absolute amount of decrease and the positive impact it would have on human health is region dependent, but further decreases will result in further beneficial decreases in $PM_{2.5}$ and O_3 concentrations.

The threshold for total Nr in the atmosphere is yet to be fixed, but depends on its rate of deposition to the surface and the sensitivity of the receptor(s). The immediate need for determining thresholds for atmospheric Nr is monitoring of NO_y and NH_x.

4.6. Thresholds for Excess Nr Effects on Terrestrial Ecosystems

In parallel with the original concept of critical loads developed by Nilsson and Grennfelt in 1988 and now widely used for air quality management in Europe (Appendix D), thresholds in general and critical loads specifically for Nr effects on terrestrial ecosystems in the U.S. should be understood to be "quantitative estimates of exposure to air concentrations of Nr compounds below which harmful effects on specified sensitive elements within ecosystem of concern do not occur according to present knowledge" (Nilsson and Grennfelt, 1988; Heittelingh et al., 2001).

In developing these quantitative estimates of thresholds and/or critical loads for terrestrial ecosystems in the U.S. (e.g., Fenn et al., 2003), it is imperative to understand the extraordinarily wide diversity and Nr-sensitivity of various components of terrestrial ecosystems in different parts of the U.S., as well as the huge differences in purposes and intensity of management and public perceptions of the value of these ecosystem components to various sectors of American society. Thus, the critical loads appropriate for maintaining species diversity in a natural grassland in northern Minnesota or a wilderness area in the Mediterranean climate of southern California are likely to be very different from those associated with direct effects on similar systems in other regions of the U.S. – or even for beneficial and/or adverse effects on other components of the same terrestrial ecosystem. For example, the threshold or critical load for adverse effects of excess Nr on understory vegetation, beneficial mycorrhizae, or lichen communities in a forest ecosystem is likely to be very different from the threshold for adverse effects on the dominant forest trees in that same ecosystem. Thus, public perceptions of "specified sensitive elements within the ecosystem" may be important in determining what specific thresholds or critical loads should be considered in order to minimize or avoid specific adverse effects of concern.

At present, the sum total of directly measured wet plus dry-deposited chemically oxidized (NO_y) and chemically reduced (NH_x) inorganic Nr loads in various states within the contiguous U.S. are on the order of 3 to 15 kg N/ha/year (NADP, 2010; CASTNET, 2010). As shown in Appendix A, a three-year run of the Community Multiscale Air Quality (CMAQ) model also provided estimates of the average annual total Nr loads (including organic forms as well as inorganic NO_y and NH_x forms of Nr) in the contiguous U.S. These model estimates varied from minimal deposition values of about 3 kg N/ha/year to maximum estimated values of about 17 kg N/ha/year. This range agrees well with the range of the measurements.

These directly measured and modeled estimates of total (wet plus dry) deposition of organic and inorganic forms of Nr indicate that there are several areas, especially in the eastern U.S., and a few areas of the western U.S.,

where current total Nr loads are already very close to, or will very likely soon exceed, the recommended threshold and critical load estimates provided by Bobbink et al. (2010) in their review of scientific evidence regarding the impacts of atmospheric nitrogen deposition on plant diversity in terrestrial ecosystems.

4.7. Comments on Nr Critical Loads

In recent years, the Acid Rain Action Plan developed by New England governors and eastern Canadian Premiers has led to evaluations of critical loads to surface waters and forests in that region. Those studies identified many waters and forest lands that met or exceeded critical load capacity for combined sulfur and nitrogen deposition both in the New England States and in the eastern Canadian provinces. The plan set target decreases of 20 to 30% for nitrogen oxide emissions by 2007 and a 50% decrease in sulfur dioxide emissions by 2010. These targets are intended to decrease long-range transport of air pollutants, acid deposition, and nutrient enrichment of marine waters in this region.

In May 2006, a Multi-Agency Critical Loads Workshop was held, which led to the formation of a Critical Loads Ad-Hoc Committee (CLAD) within the National Atmospheric Deposition Program (NADP). A goal of the program is to "provide consistency in development and use of critical loads in the U.S." One outcome is a project undertaken by the Northeast States for Coordinated Air Use Management (NESCAUM) to: "estimate critical loads of sulfur and nitrogen in atmospheric deposition for areas where sufficient knowledge, data, and methods exist" and "to demonstrate the use of critical loads as a tool for assessing environmental policies and programs and managing natural resources."

A February 2007 Workshop sponsored by EPA on "The Assessment of Health Science for the Review of the National Ambient Air Quality Standards (NAAQS) for Nitrogen (NO_x) and Sulfur Oxides (SO_x)" expansively reviewed both ecosystem and human health effects toward revision of the NAAQS. Policy discussions at this workshop raised the questions of whether critical loads assessments were an effective means of improving ecosystem management, and whether the science was understood well enough to use critical loads as a management tool. The conclusion was that, although there was a substantial body of accumulated scientific evidence, there was only limited use of critical loads approaches for management of air quality in the U.S. The Multi-Agency Workshop on Critical Loads (mentioned above) was cited at EPA's 2007 workshop as an agenda-setting effort to resolve some of the science and policy issues that could help advance critical loads approaches in the U.S. The Integrated Nitrogen Committee believes that the primary reason critical loads are not now used in the U.S. is that policy makers in this country have so far not been willing to adopt unfamiliar air and water quality management approaches or approaches that have not

been evaluated directly in this country. Thus, the Committee recommends that EPA consider implementation of the critical loads concept for management of deleterious Nr effects in various parts of the U.S.

Finding 16: The Committee finds that there have been persistent increases in the amounts of Nr that have been emitted into and retained within various ecosystems, affecting their functioning. Unless this trend is reversed, it will become increasingly difficult for many of these ecosystems to provide the services upon which human well-being is dependent. The Committee believes that there is a need to regulate certain forms of Nr to address specific problems related to excess Nr, and we believe that the best approach for an overall management strategy is the concept of defining acceptable total Nr critical loads for a given environmental system.

Recommendation 16: *The Committee recommends that the Agency work toward adopting the critical loads approach concept in determining thresholds for effects of excess Nr on terrestrial and aquatic ecosystems. In carrying out this recommendation the Committee recognizes that it will in many cases be necessary for the Agency to enter into new types of research, policy, and regulatory agreements with other federal, state, and tribal units based on cooperative, adaptive, and systemic approaches that derive from a common understanding of the nitrogen cascade.*

4.8. Tradeoffs of Nr Impacts in Risk Reduction Strategies

Because nitrogen is such an abundant and widespread element, and Nr is such a critical component of the earth's biosphere, associated impacts are many and pervasive. In many cases, strategies to manage the impacts of Nr involve tradeoffs, i.e., mitigating one type of impact may exacerbate others. Given the interactions among oxidized and reduced N species, it is important to recognize the potential for unintended consequences to occur as a result of strategies that are aimed at limiting one form of Nr in air or water but lead to the increased production of other forms of Nr, or the formation and release of other contaminants of concern. For example, stringent control of point sources of Nr can be energy-intensive, requiring significant energy investments for chemicals, electricity, and other support, and this may in turn lead to the production of more reactive nitrogen and increased CO_2 emissions. Furthermore, there may be environmental impacts of these treatment processes, particularly in the production of solid wastes that can be significant environmental hazards. This is the main reason that a life cycle approach is necessary in evaluating any remediation or treatment scheme. In addition, as discussed in Section 3.1.2, numerous lakes, reservoirs, rivers, and fjords worldwide exhibit N and P co-limitation, either simultaneously or in seasonally-shifting patterns. Therefore, strategies are needed to reduce both P and N inputs. Not all control practices will be effective

for dual nutrient reduction and this must be taken into consideration. Four categories of tradeoffs examined below are: ammonia release from concentrated feed lot operations (CAFOs), concerns about human nutrition, nitrification and denitrification, and nitrogen-carbon related impacts.

Ammonia release from CAFOs

As a result of effluent guidelines for NH_3 in aquatic systems, state and federal regulations and programs under the CWA were developed to address water quality protection from CAFOs. The resulting manure management systems utilized NH_3 volatilization as a means to remove N and decrease the N in the manure when land applied. Only recently has the resulting increase in NH_3 emission into the air been viewed as a potential problem with respect to air quality concerns and N deposition.

Finding 17: Current EPA policy (40 CFR Part 51, Clean Air Fine Particle Implementation Rule) discourages states from controlling ammonia emissions as part of their plan for reducing $PM_{2.5}$ concentrations. In this rulemaking, EPA states that "ammonia reductions may be effective and appropriate for reducing $PM_{2.5}$ concentrations in selected locations, but in other locations such reductions may lead to minimal reductions in $PM_{2.5}$ concentrations and increased atmospheric acidity." Ammonia is a substantial component of $PM_{2.5}$ in most polluted areas of the United States at most times. While it is true that reducing NH_3 emissions might increase the acidity of aerosols and precipitation, the net effect of NH_3 on aquatic and terrestrial ecosystems is to increase acidity. After being deposited onto the earth's surface, NH_4^+ is under most circumstances quickly nitrified, increasing the acidity of soils and waters. The Committee is unaware of any evidence that NH_3 reduces the toxicity of atmospheric aerosols or that high concentrations of NH_3 occur naturally over any substantive area of the United States. It has not yet been established which components of PM have substantive impacts on human health, but the total concentration of $PM_{2.5}$ correlates with morbidity and mortality, and NH_3 contributes to $PM_{2.5}$. The visibility degradation and other adverse effects associated with $PM_{2.5}$ are related to aerosol surface area or mass where NH_4^+ certainly plays a role.

Recommendation 17: *The Committee recommends that the EPA presumption that NH_3 is not a $PM_{2.5}$ precursor should be reversed and states should be encouraged to address NH_3 as a harmful $PM_{2.5}$ precursor.*

Swapping N between environmental systems

Nitrous oxide is produced in "natural" and agricultural soils, and all aquatic systems, almost exclusively as a result of the microbial processes of nitrification and denitrification. As NH_4^+ ion is the initial mineral N product formed during organic matter mineralization and most of the fertilizer used worldwide is NH_4^+ based (e.g., urea, ammonium sulfate) (FAO, 2007), the suite

of microbiological reactions that result in the release of gaseous N products need to be considered.

Nitrification is the oxidation of NH_4^+ ion to NO_3^- (Figure 17). Most commonly, nitrification is a chemolithotropic process consisting of the conversion of ammonium to nitrite, which is then converted to NO_3^- by a second group of bacteria. The ammonium oxidizing bacteria (AOB) are obligate aerobes with some species that are tolerant of low-oxygen environments. The most common genera of autotrophic NH_4^+ oxidizers are *Nitrosospira* and *Nitrosomonas*. AOB are found in most aerobic environments where ammonium is available through the mineralization of organic matter or where N compounds are added.

Biological denitrification is the dissimilatory reduction of NO_3^- and nitrite to produce NO, N_2O, and N_2 by a taxonomically diverse group of bacteria. These bacteria synthesize a series of reductases that enable them to utilize successively more reduced N oxides as electron acceptors in the absence of oxygen. The general reductive sequence is shown in Figure 17. In addition to the free-living denitrifiers, symbiotically-living Rhizobia in root nodules of legumes are able to denitrify nitrate and produce nitrous oxide (Mosier and Parkin, 2007).

The abundant denitrifiers are heterotrophs, which require sources of electron-reducing equivalents contained in available organic matter. Factors that most strongly influence denitrification are oxygen, nitrate concentration, pH, temperature, and organic carbon. The reductive enzymes are repressed by oxygen but not by NH_4^+. Nitrous oxide reductase appears to be more sensitive to oxygen than either NO_3^- or nitrite reductase. Therefore N_2 production predominates in more anoxic sites and N_2O production may be greater in more aerobic conditions. However, the ratio of N_2 to N_2O emitted may also be affected by high NO_3^- concentrations and associated higher levels of electrical conductivity and osmotic stress and soil pH (low pH favors N_2O production).

Given these interactions among oxidized and reduced N species (discussed above), it is important to recognize the potential for unintended consequences to occur as a result of strategies that may be aimed at limiting one form of Nr in air or water but lead to the increased production of other forms of Nr. One such instance is the potential offsetting of the benefits of NO_3^- remediation at the expense of increasing input of N_2O to the atmosphere. An example of such a situation involves NO_3^- leached from agricultural fields, much of which could be removed from drainage water in natural or reconstructed wetlands. This process is ideal if the denitrification process goes to completion, i.e., only N_2 is produced. If, however, the process is incomplete, and NO and N_2O gases are emitted, then the end result may create a compensating risk that could be greater than that posed by the nitrate that is removed. This is
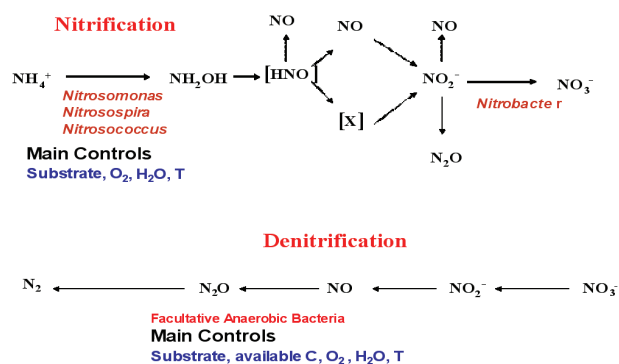

Figure 17: Diagram of the nitrification and denitrification processes

Source: Mosier and Parkin, 2007. Reprinted with permission; copyright 2007, Taylor & Francis Group LLC – Books.

because NO continues to be reactive in the atmosphere and is eventually redeposited in aquatic or terrestrial systems, and N_2O is a GHG that has an atmospheric life time of approximately 100 years and a radiative forcing of approximately 300 times that of CO_2 on a hundred-year time frame (IPCC, 2001). N_2O is also a major source of NO in the stratosphere and depletes stratospheric ozone (Crutzen, 1981). If more of the NO_3^- denitrified is converted to N_2O in wetlands than upstream or downstream, the environmental cost may be high. Hernandez and Mitsch (2007) found that permanently flooded wetlands had lower N_2O/N_2 ratios of emissions than did intermittently flooded wetlands. They also found that the ratio was higher in the cold months even though the flux rates are much lower then. A full risk assessment needs to be made to determine how much of such "pollutant swapping" is advisable.

A similar potential exists for Nr mediation in sewage treatment. The current practice is to convert ammonia/ammonium that mineralizes from excreted organic matter to nitrate through the nitrification process. As nitrate-containing effluent from sewage treatment flows into aquatic systems the nitrate may be denitrified, resulting in N_2O production if denitrification is not complete. The protein consumption by some 301 million humans in the U.S. results in the processing of ~ 2 Tg of N annually (~18.4 g N/ person/day), much of which flows through sewage treatment facilities and ultimately leads to the production of between 0.06 and 0.1 Tg of N_2O-N /yr in aquatic systems or soils to which sewage sludge is applied.

Tradeoffs among C and N-driven impacts

Reactive N also contributes to many impacts on the environment that are also impacted by other chemical species, notably carbon. As depicted in Figure 18, there are several points of tangency between the global C and N cycles. These are: combustion, agricultural production, industrial production, soil and sediment processes, and

end-of-life disposition of products. The implication of these interactions is that, in many instances, the perturbation of one cycle cannot be fully assessed without including effects on the other. For example, proposals to develop bio-based products (biofuels, but also other products) as the preferable alternative to fossil-based resources are not free from impacts. Such trade-offs may involve a single impact (e.g., global climate change to which both carbonaceous gases and N_2O contribute) but may also involve trade-offs between impacts that are not easily compared. Figure 19 shows the latter case in the form of climate change impacts (to which C is a principal contributor) versus eutrophication impacts (to which nitrogen is a principal contributor) for several different biofeedstock-product combinations which are evaluated relative to the substituted commercial product made from fossil C. A value of 100% on the y-axis would mean that the bio-based alternative is no better than the fossil-based counter-product, while the negative region of the y-axis in Figure 19 represents net C sequestration. It is difficult to make direct comparisons across disparate impact categories, however Figure 19 suggests that, in choosing among alternatives, policies that aim to minimize both sets of impacts would be preferred.

Finding 18: The Committee notes that the effective management of Nr in the environment must recognize the existence of tradeoffs across impact categories involving Nr transformations and the cycling of other elements.

Recommendation 18: *The Committee recommends that the integrated strategies for Nr management outlined in this report be developed in cognizance of the tradeoffs associated with reactive nitrogen in the environment (consistent with the systems approach of overarching recommendations 2 and 3 discussed in Section 6.2 of this report). Specific actions should include:*

- *Establishing a framework for the integrated management of carbon and reactive nitrogen;*

- *Implementing a research program that addresses the impacts of tradeoffs associated with management strategies for carbon, reactive nitrogen, phosphorus, and other contaminants of concern;*

- *Implementing a research and monitoring program aimed at developing an understanding of the combined impacts of different nitrogen management strategies on the interchange of reactive nitrogen across environmental media.*
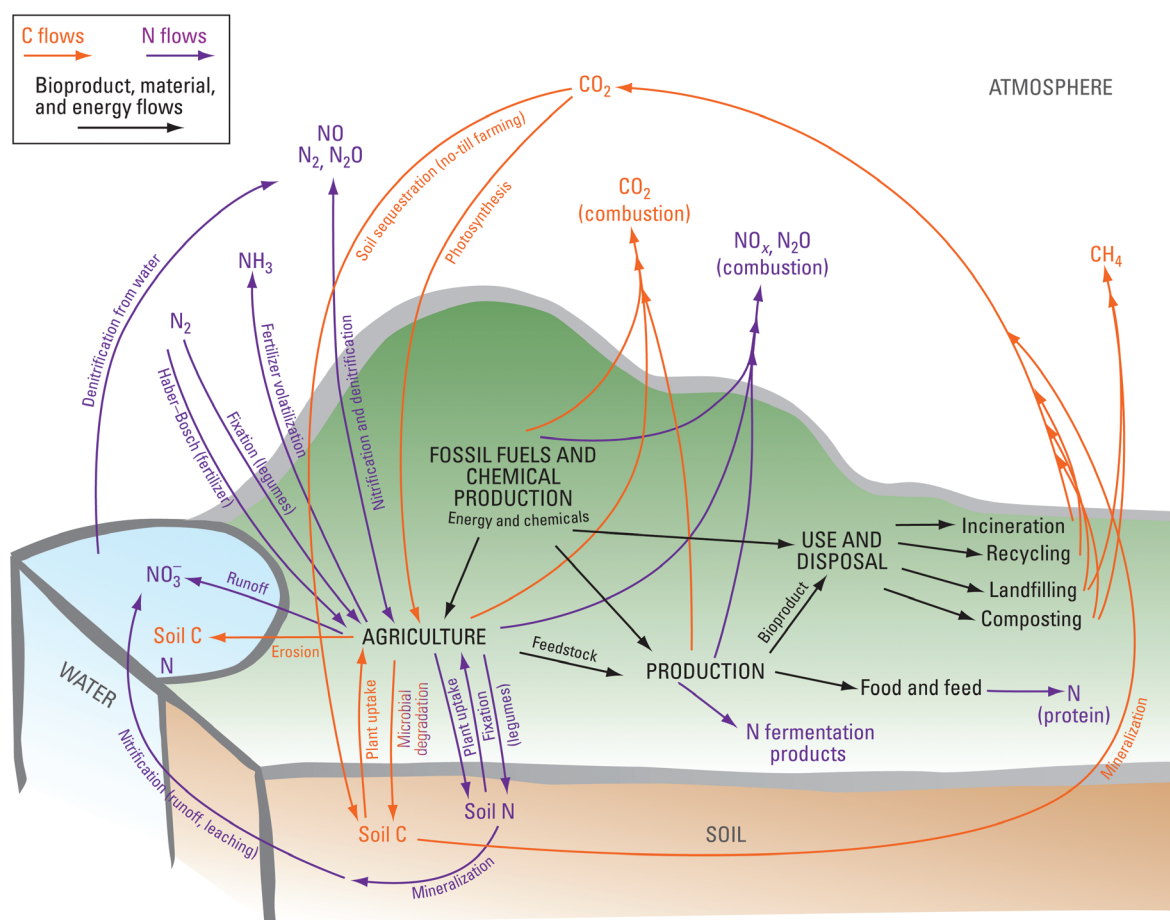

Figure 18: Combined carbon and nitrogen global cycles
Source: Miller et al., 2007 (Figure 1, p. 5178). Reprinted with permission; copyright 2007, American Chemical Society.

4.9. Interactions of the N Cascade and Climate

Weather and climate vary substantially on many time scales including the interannual. Long-term (decadal or more) changes in climate as have been predicted by IPCC (2007a,b) may have profound effects on the N cycle; conversely, changes in the biogeochemical cycle of Nr can induce climate forcing. While it is beyond the scope of
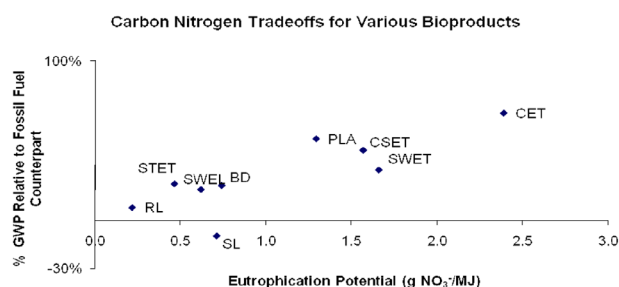
Carbon Nitrogen Tradeoffs for Various Bioproducts

Figure 19: Comparisons between Global Warming Potential (GWP) and eutrophication impact categories for various bioproducts

Abbreviations: BD=Biodiesel; CET=Corn Ethanol; CSET=Corn & Stover Ethanol; PLA=Polylactic Acid (Corn); RL=Rapeseed Lubricant; SL=Soybean Lubricant; STET=Stover ethanol; SWEL=Switchgrass Electricity; SWET=Switchgrass Ethanol.

Source: Adapted from Miller et al., 2007 (Figure 2, p. 5180). Adapted with permission; copyright 2007, American Chemical Society.

this report to fully address how cycles of C and N interact (see Figure 18 for a general treatment of the intersection points of C and N cycles), there are several ways in which climate impacts the biogeochemical cycle of Nr and vice versa (e.g., Yienger and Levy, 1995; Holland et al., 1997; Hungate et al., 2003; Hungate et al., 2004; Sutton et al., 2007; Thornton et al., 2007; Levy et al., 2008; Sokolov et al., 2008). These are highly interactive and nonlinear systems. The following important interactions are noted:

- Increased deposition of Nr into terrestrial and aquatic ecosystems can alter the sequestration of carbon, while increased ambient CO_2 can change the deposition and uptake of Nr.

- Nitrate flux from fields to surface waters increases with increasing rainfall (see Box 5:The Impact of Climate Change on Agricultural Discharge of Reactive Nitrogen).

- Increasing temperature can both increase and decrease atmospheric loading of particulate matter.

- Aerosols (PM) have direct and indirect (through cloud microphysics) effects on radiative forcing of climate and on the hydrological cycle.

- N_2O and O_3 are greenhouse gases.

- Soil Nr chemistry and emissions of N_2O, NH_3, and NO depend on environmental conditions such as temperature and soil moisture.

- The amount of Nr deposited and exported from the U.S. depends on meteorological variables including wind speeds and convection.

Numerical models, when verified against past climates, can provide insight into possible future climates and their impacts on the nitrogen cycle. For example, increasing temperatures increase the amount of NO_x control necessary to achieve the same amount of photochemical smog control (Bloomer et al., 2009; Jacob and Winner, 2009). The EPA program for studying the impact of climate change on photochemical smog (air pollution ozone) production offers a useful model; see Jacob and Winner (2009) for an overview.

Finding 19: The biogeochemical cycle of Nr is linked to climate in profound, but nonlinear ways that are, at present, difficult to predict. Nevertheless, the potential for significant amplification of Nr-related impacts is substantial, and should be examined in more complete detail.

Recommendation 19: *The EPA should support cross-disciplinary and multiagency research on the interactions of climate and Nr. To determine the interactions of global biogeochemical Nr cycles and climate, the Committee suggests that EPA follow a series of steps such as:*

1. *Select several likely scenarios for global climate from the IPCC report for the year 2050.*

2. *Down-scale statistics or nest regional climate models within each of these global scenarios to generate meteorological and chemical fields (e.g., temperature, relative humidity, winds, precipitation, CO_2) for a few years around 2050.*

3. *Run several independent biogeochemical Nr models (earth system models that include air/water/land) for North America for these years with current Nr and emissions and application rates.*

4. *Rerun models with decreased Nr emissions/application to evaluate strategies for controlling impacts such as those described in this report.*

The discharge of reactive nitrogen from intensively managed agroecosystems is characterized by a number of attributes that often exhibit a high degree of variability: fluctuating material flows associated with the degree of nitrogen fixation and the extent of denitrification, the interdependence of crops in rotation, and dependence on geography, weather patterns (particularly rainfall intensity, duration, and frequency), soil type, and agricultural practices.

One way to gauge the impact of climate change on such systems is to examine the ranges exhibited by historical data that collectively encompass the range of impacts that are anticipated. The assumption is that a changing climate will systematically alter governing attributes in plausibly predictable ways, for example increased annual rainfall and temperature over a large geographic region. The IPCC has provided general climate-induced impacts for world regions (IPCC, 2007a,b).
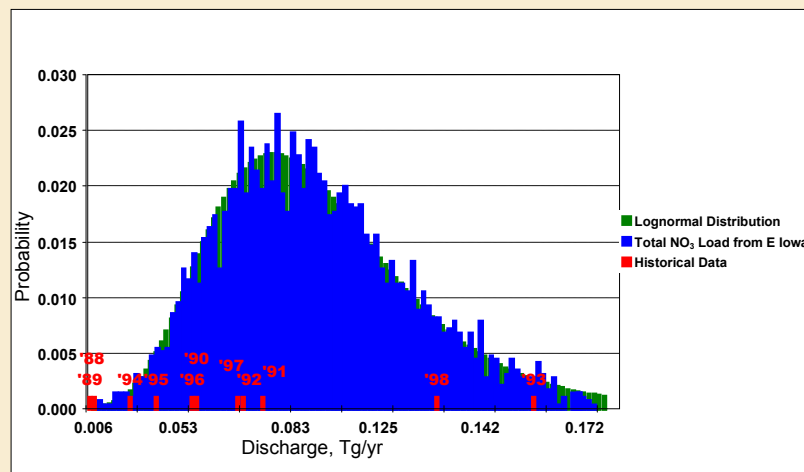

Figure 20: Probability of given discharge level for nitrate in the watersheds of eastern Iowa

Based on the simulation model of Miller et al. (2006). Red markers are historical data of discharges according to year as reported by Powers (2007). Green bars represent a log-normal distribution.

The general impact of climate change on the discharge of reactive nitrogen from agroecosystems can be discerned from the information in Figure 20. This figure shows a probability distribution for nitrate discharged from the watersheds of eastern Iowa (approximately 50,000 square km), which are dominated by corn-soybean agroecosystems (a general description of the region can be found in Kalkhoff et al., 2000). It is derived from information on the input of synthetic fertilizers in the region during the period 1989-1999, and includes factors that describe the transformation and transfer of Nr once applied. The distribution shown was generated using a Monte Carlo technique, details of which can be found in Miller et al. (2006). Also included in Figure 20 (in green) is a standard log-normal distribution, which the simulation most closely fits, and independently measured annual nitrate runoff data (an output of the system) over the same time period, as reported by Powers (2007). The simulation is not perfect, but it does capture the extremes of reactive nitrogen discharge, as represented by data for the years 1993 and 1998.

Figure 20 shows that the interannual variation in nitrate discharged is nearly 30-fold during the 11-year observation period. While the impact of climate change on such a system cannot be predicted for a given year, Figure 20 provides a basis for visualizing shifts in nitrate discharge due to changes in those factors that affect Nr transformation and transfer. For example a climate change scenario that predicts a general increase in precipitation amount and frequency, other factors being constant, will tend to shift the distribution of Figure 20 to the right, resulting in generally higher discharges of nitrate (see for example Vanni et al., 2001; the data point for 1993 in Figure 20 corresponds to precipitation in the region that was approximately 1.8 times the long-term annual average). Other factors, of course, may amplify or retard such impacts. Understanding whether or not implementation of best management practices and advanced technological methods can counteract climate change trends that favor increases in discharge would require a series of significant research studies and advances in modeling capabilities.