Reactive Nitrogen in the United States;

An Analysis of Inputs, Flows, Consequences, and Management Options

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Introduction

Reactive nitrogen (Nr) encompasses biologically and radiatively active, and chemically reactive nitrogen compounds[1]. Excess reactive nitrogen releases to the environment have been recognized as a major causative agent for air and water quality degradation causing major impacts to human and ecosystem health. Some sources and forms have been regulated through legislation such as the Clean Air and Clean Water Acts. Specific regulations have been introduced to decrease primary air pollutants such as nitrogen oxides (NO_x), eutrophication of waterways by nitrate ions (NO_3^-) and acid rain. While notable progress has been made, the fragmentation of this approach has proven a barrier to achieving additional improvements. This report has been developed to identify a more comprehensive analytical framework and alternative means by which to manage reactive nitrogen, and to advise EPA on strategies that might prove more effective. This requires an integrated systems approach that identifies the stocks and follows the flows and chemical transformations of reactive nitrogen through air, land and water. A more comprehensive integrated approach would also help to establish priorities for action.

At the global scale, human activities now create approximately twice as much Nr as natural continental ecosystems. In the United States (US), Nr creation by human activity is about five-times larger than natural processes. Human activities create Nr by: (1) the Haber-Bosch process to generate ammonia (NH_3) for synthetic nitrogen fertilizer and industrial feedstocks, (2) the enhancement of biological nitrogen fixation (BNF) in crop cultivation (e.g., legumes), and (3) the combustion of fossil fuels[2], and industrial process emissions. The first two anthropogenic activities form Nr on purpose; all three result in unwanted pollutants.

Anthropogenic creation of Nr provides essential benefits for humans--first and foremost in meeting human dietary needs. In fact, a large fraction of the human population of the earth could not be sustained if synthetic nitrogen fertilizers did not augment food production significantly all over the world. Essentially all of the Nr created by human activities, however, is released to the environment, often with unintended negative consequences. It circulates between, and accumulates within, the atmosphere, and aquatic and terrestrial ecosystems and contributes to a number of adverse public health and environmental effects, including photochemical smog, nitrogen-containing trace gases and aerosols, decreased atmospheric visibility, acidification of terrestrial and aquatic ecosystems, eutrophication of coastal waters (i.e., harmful algal blooms, hypoxia), drinking water concerns, freshwater Nr imbalances, greenhouse gas emissions and subsequent climate change, and stratospheric ozone depletion.

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[1] The term reactive nitrogen (Nr) is used in this paper to include all biologically active, chemically reactive, and radiatively active nitrogen (N) compounds in the atmosphere and biosphere of Earth. Thus, Nr includes inorganic chemically reduced forms of N (NHx) [e.g., ammonia (NH_3) and ammonium ion (NH_4^+)], inorganic chemically oxidized forms of N [e.g., nitrogen oxides (NOx), nitric acid (HNO_3), nitrous oxide (N_2O), N_2O_5, HONO, peroxy acetyl compounds such as PAN, and nitrate ion (NO_3^-), as well as organic compounds (e.g., urea, amines, amino acids, and proteins), in contrast to non-reactive gaseous N_2.

[2] Burning of biomass such as in forest fires or wood-burning stoves releases Nr into the atmosphere, but this is generally from fuel N; it represents an exchange of Nr between media and not the generation of new Nr. Burning of biofuels in high-temperature internal combustion engines such as ethanol from corn or biodiesel used to power vehicles does generate new Nr, and EPA emissions inventories account for these under transportation or "Highway Vehicles." In this report, emissions of new Nr from biofuels is grouped with Nr from fossil fuels.

Table A: Examples of Impacts of Excess Reactive Nitrogen on Human Health and Environment

Impact	Cause	Location	Metric	Source	Reference
Hypoxia of coastal waters	Excess nutrient loading, eutrophication, variable freshwater runoff	Gulf of Mexico, other estuarine and coastal waters	Benthic finfish/shellfish habitat loss, fish kills, sulfide toxicity, costs >$50 million annually	N, P from energy and food production	Bricker et al. 1999, Verity et al. 2006; USEPA SAB 2008; Rabalais et al., 1999; Mitsch et al., 2001
Acidification of surface waters; loss of biodiversity	Acidification of soils, streams and lakes is caused by atmospheric deposition of sulfur, HNO_3, NH_3 and ammonium compounds.	Primarily mountainous regions of the US	Out of 1,000 lakes and thousands of miles of streams surveyed, 75% of the lakes and 50% of the streams were acidified by acid deposition	Fossil fuel combustion and agriculture	http://www.epa.gov/acidrain
Harmful Algal Blooms	Excessive nutrient loading, climatic variability	Coastal waters inland and coastal waters	Fish kills, losses of drinking and recreational waters costs >$100 million annually	Excess nutrient (N & P) loading	Paerl 1988; ECOHAB 1995; NRC 2000
Human mortality	$PM_{2.5}$, O_3 and related toxins.	US urban and nearby areas.	Pollution related deaths estimated at 28,000-55,000 per year.	NO_y and NH_x from fossil fuels and agriculture	Mokdad et al., 2004]; Ezzati et al., 2004.
Visibility decrease	Fine particulate matter	National Parks and wilderness areas	visibility impairment	NO_y and NH_x from fossil fuels and agriculture	Malm et al., 2004 EPA-CASAC-09-010
Crop yield loss	Ozone	Eastern and Western US	$ 2-5 billion/year	Utilities & traffic	Heck et al, 1984
Forest decline	Ozone and acid deposition	Eastern and Western US	Decreased timber growth; increased susceptibility to disease and pests	Utilities, traffic, and animal agriculture	Johnson & Siccama, 1983; MacKenzie & El-Ashry, 1990
Biodiversity loss	Nitrogen deposition	Grasslands and forests in US receiving N deposition in excess of critical load	Decrease in species richness of grasslands and forests	Utilities, traffic, and animal agriculture	Bobbink et al., 2009; Fenn et al., 2003.

Nr effects are manifest as declines in both human health (e.g., respiratory and cardiac diseases) and ecosystem health (e.g., coastal eutrophication and loss in biodiversity). The effects are often magnified because any one atom of nitrogen in the environment can contribute to both beneficial and detrimental effects in sequence, as excess Nr moves through various environmental reservoirs. This feature of Nr is the conceptual foundation for the nitrogen cascade.

The nitrogen cascade provides a conceptual framework to assess and quantify the effects of Nr as it originates, flows and transitions through the atmosphere, land and water. The framework helps organize Nr sources within each environmental system, its transfer among the systems, and the benefits and impacts along the way. It further highlights potential decreases in emissions or management intervention within each system that integrates those actions among sources and media (air, land, water) to provide an efficient mechanism for regulation.

To assist EPA in its understanding and management of nitrogen-related air-, water-, and soil-pollution issues, this Integrated Nitrogen Committee (INC) was formed and charged by the Science Advisory Board (SAB) of the US Environmental Protection Agency to address the following objectives:

1. Identify and analyze, from a scientific perspective, the problems nitrogen presents in the environment and the links among them;
2. Evaluate the contribution an integrated nitrogen management strategy[3] could make to environmental protection;
3. Identify additional risk management options for EPA's consideration; and
4. Make recommendations to EPA concerning improvements in nitrogen research to support risk reduction.

This Executive Summary contains an overview of the analyses made by the INC to fulfill the four charges. The detailed analyses that support the conclusions and recommendations follow in the main body of the report. It summarizes the Nr inputs to the US, the fate of the Nr in the US and addresses how both public health and environmental impacts are, and could be, assessed. The recommendations are organized within three tiers. The first tier contains "overarching" recommendations for both research and management that the INC believes will help the EPA develop an integrated nitrogen management strategy. The second tier consists of four recommendations ("Target Goals") that are suggested for management actions that could achieve an initial 25% decrease in the amount of Nr lost to the US environment using existing technologies. The third tier offers more specific recommendations that support and clarify the intent of the overarching recommendations.

In recent years, other studies have examined some aspects of the biogeochemical flows of Nr in the US nitrogen cycle, the impacts of alterations in those flows, and suggested policies for addressing some specific consequences of those alterations. By contrast, this report of the INC committee seeks to provide an integrated and holistic approach to all aspects of the current Nr-

[3] An integrated nitrogen management strategy takes a holistic approach for managing Nr. In the context of the nitrogen cascade, all Nr anthropogenic creation and destruction mechanisms and all Nr uses are recognized. The strategy should take account of synergies and trade-offs, to ensure that decreasing one problem related to nitrogen does not result in other unintended adverse environmental, economic and societal consequences. By identifying relative priorities, assessing cost effectiveness and risks, the strategy should seek to maximize the benefits of Nr, while limiting overall adverse effects.

management problems and challenges in our country. The INC is convinced that the EPA has a potentially powerful lead role to play -- together with other federal, state, and local organizations -- in developing integrated strategies to maximize beneficial impacts and decrease detrimental impacts of Nr management in this country.

Overview

The INC addressed the four objectives in the following manner.

Objective 1: Identify and analyze, from a scientific perspective, the problems Nr presents in the environment and the links among them.

To address this objective, the INC used the nitrogen cascade framework to determine the major sources of newly created Nr in the US (see Figure 1). The flows of Nr within the food, fiber, feed and bioenergy production systems for the US were examined, paying special attention to the locations in each of these systems where Nr is lost to the environment. The same process was employed for energy production but, since all the Nr formed during energy production is lost to the environment, the committee identified the important energy producing sectors that contribute to Nr formation.

The committee next examined the fate of the Nr lost to the environment, estimated the amount stored in different systems (e.g., forest soils) and tracked Nr as it is transferred from one environmental system (e.g., the atmosphere) to another (e.g., terrestrial and aquatic ecosystems).

The introduction of Nr to the US environment, its flows between reservoirs, transfers of Nr out of the US environment and losses of Nr due to conversion to N_2 are summarized in Figure 2.

These two activities set the stage for addressing the environmental and human health problems Nr presents, and the links among them. Using the nitrogen cascade, the committee identified the impacts Nr has on people and ecosystem functions as it moves through different systems. The committee also addressed the metrics that could be used, i.e., impacts due to environmental changes (e.g., acid deposition), vs. impacts due to losses of ecosystem services (e.g., loss of biodiversity), and trade-offs among Nr Impacts.

Objective #2: Evaluate the contribution an integrated N management strategy could make to environmental protection.

An integrated management strategy should take into account the contributions of all Nr sources, and all chemical species of Nr that adversely impact both human health and environmental systems. Further, an integrated strategy should ensure that solving one problem related to Nr does not exacerbate another problem or diminish necessary human services to produce food, feed, fiber, or bioenergy. In short, the strategy should seek to achieve desirable benefits of Nr, while limiting adverse effects.

To address this challenge, the committee identified several actions that could be taken to better manage Nr in one environmental system that have caused unintended consequences in another. Examples of management actions that could be taken that would ‘integrative’ in nature are highlighted.

Objective #3: Identify additional risk management options for EPA's consideration.

The INC has identified four major Target Goals for actions that collectively will decrease Nr losses to the environment by about 25%, recognizing that decreasing Nr emissions by these actions will result in further decreases in Nr-related impacts throughout the nitrogen cascade. INC has suggested several ways in which each of these Target Goals could be attained including conservation measures, additional regulatory steps, application of modern technologies, and end-of-pipe approaches. These are initial actions; others should be taken once the recommended actions are completed. Thus the last sections of this report focus on a better understanding of N dynamics and impacts in the US.

Objective #4: Make recommendations to EPA concerning improvements in Nr research to support risk reduction.

Throughout the report, there are summary statements, labeled "Findings." Attached to these findings are one or more specific "Recommendation" for actions that could be taken by EPA or other management authorities. In each case, the intent is to provide the scientific foundation regarding a specific Nr-relevant environmental issue and one or more recommendations by which EPA acting alone or in cooperation with other organizations could use currently available technology to decrease the amount of Nr lost to the US environment.

The remaining sections of this Executive Summary cover the points made above in greater detail.

Nr Sources and Transfers

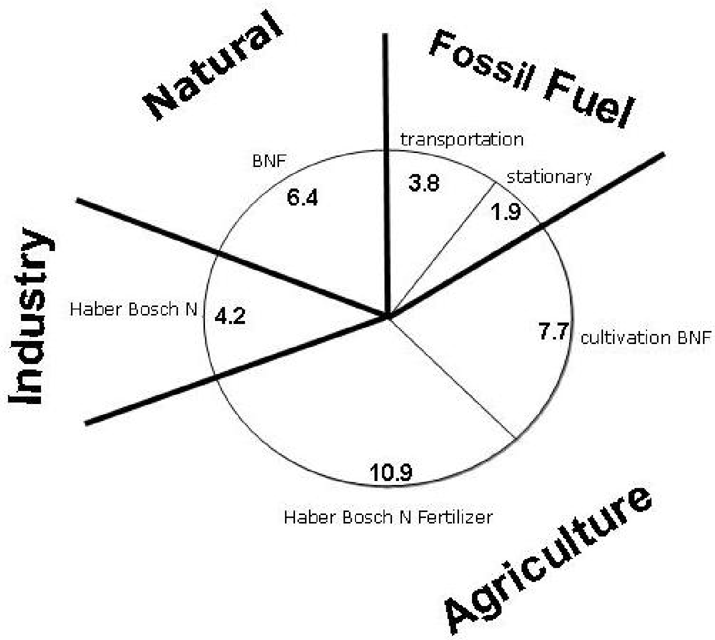
Sources of Nr

At the global scale, human activities produce approximately twice as much reactive nitrogen as do natural processes; in the US, however, the amount of Nr produced by human activities is approximately five-times larger than natural processes. As shown in Figure 1, natural ecosystems in the US introduce about 6.4 teragrams (Tg) of reactive nitrogen per year (Tg N/yr). In contrast, human activities introduce about 28.5 Tg N/yr.

The largest single source of Nr in the US is the Haber-Bosch process, which introduces about 15.2 Tg N/yr -- 9.4 Tg N/yr from domestic US Nr production and 5.8 Tg N/yr from imports of synthetic Nr fertilizers, feed grains and food. This total amount is used in three ways -- 9.9 Tg N/yr is used to produce agricultural crops; 1.1 Tg N/yr is used to produce turf grasses; and 4.2 Tg N/yr is used as industrial feed stocks for production of nylon, refrigerants, explosives and other commercial products. Cultivation-induced BNF introduces 7.7 Tg N/yr into Agroecosystems.

Fossil fuel combustion is the second largest source of Nr, relative to agricultural sources. It introduces approximately 5.7 Tg N/yr into the environment (almost entirely as NO_x) -- 3.8 Tg N/yr from transportation sources and 1.9 Tg N/yr from stationary sources such as electric utilities and industrial boilers.

Figure 1: Sources of reactive nitrogen (Nr) introduced into the US in 2002 (Tg N/yr)



Explanatory notes:

- Numerical units = teragram of reactive nitrogen (Nr) per year (Tg N/yr)
- Natural BNF = biological nitrogen fixation in natural grasslands, rangelands, and forests,
- Fossil Fuel-Transportation = combustion of fossil fuels in transportation vehicles.
- Fossil Fuel-Stationary = combustion of fossil fuels in power plants and industrial boilers.
- Agriculture-cultivation BNF = agricultural augmentation of biological nitrogen fixation -- for example by planting of nitrogen fixing legumes.
- Agriculture-Haber Bosch N fertilizer = agricultural use of synthetic nitrogen fertilizers produced by the Haber Bosch process for converting gaseous N_2 to Nr.
- Industry-Haber Bosch N = Industrial sources of Nr produced by the Haber-Bosch process.

The third largest source of Nr introduced into the US is enhancement of biological nitrogen fixation (BNF) by cultivation of legumes like soybeans and alfalfa that have nitrogen-fixing symbionts, or by crops like rice that have nitrogen-fixing bacteria in their rhizosphere. These Nr fixing crops introduce about 7.7 Tg N/yr. A small amount of additional Nr is also imported in grain and meat products; in 2002 this source of added Nr was approximately 0.2 Tg N/yr (not shown in Figure 1).

In summary, agricultural production of food, feed, fiber, and bioenergy and combustion of fossil fuels are the largest sources of Nr released into the environment in the United States. The percentage distribution of Nr released to the US environment from human activities in 2002 was: about 65% from agricultural sources, about 20% from fossil fuel sources, and about 15% from industrial sources (Figure 1).

Although fossil fuel combustion is widely recognized within EPA and society in general to be a major source of nitrogen pollutants and resulting environmental quality concerns in the US, in fact, feed and food production and subsequent consumption by animals and humans are much larger (about 3.3 times larger!) sources of reactive nitrogen than fossil fuel combustion.

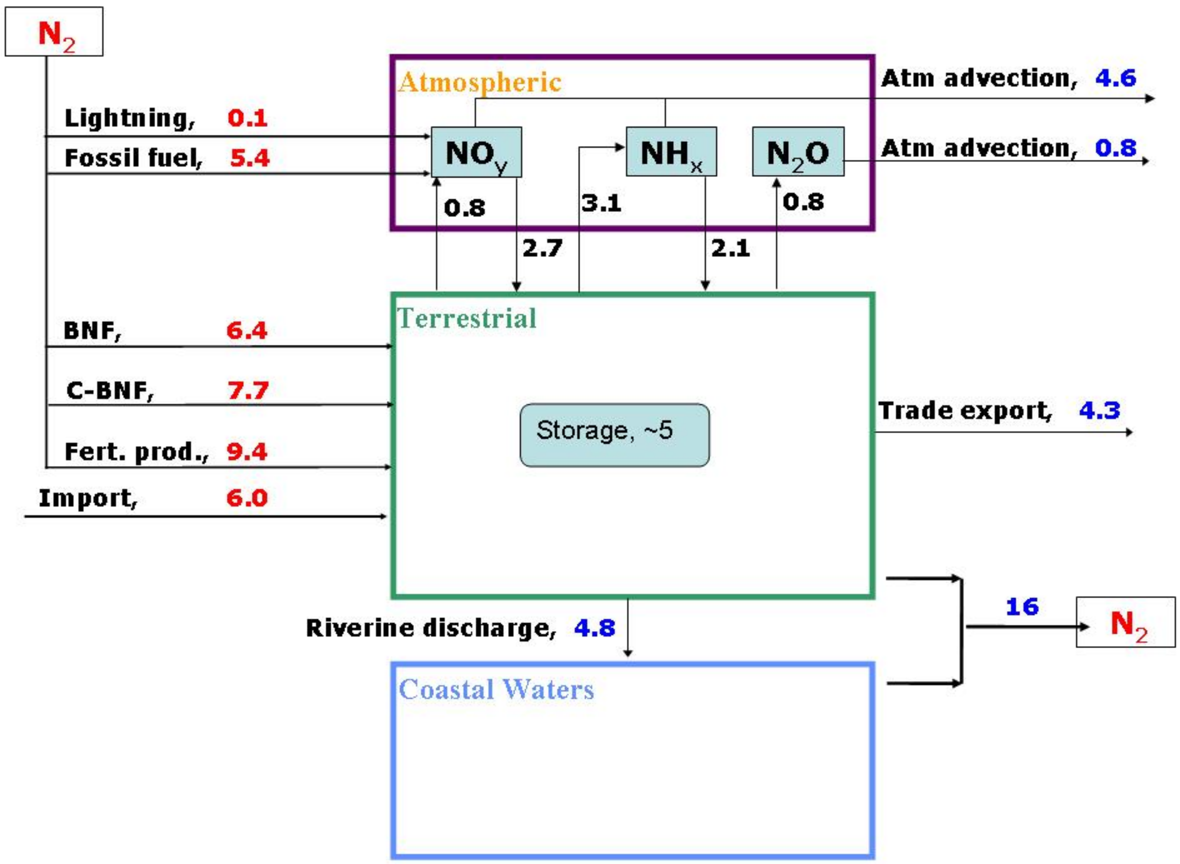
Transfers of Nr among environmental systems

There are several possible fates for the approximately 35 Tg N/yr introduced into the US environment each year from natural sources and human activities. Figure 2 illustrates these fates by showing the cascading flow of reactive nitrogen once it enters the atmospheric, terrestrial, and aquatic environments. The different environments have special characteristics. Nr in terrestrial and aquatic environments can be converted to non-reactive N_2 while much of the human morbidity and mortality attributed to Nr relates to the atmospheric environment. Given the flows of Nr and changes in its form within and between the different environments, decisions about where one intervenes to control or limit the impact of Nr become critical to both cost effectiveness and the extent of negative impact. For example, insofar as Nr's entry into terrestrial systems can be limited, its flow and transfer into aquatic and atmospheric systems may be mitigated and potential negative impacts avoided.

Emissions of N_2O discharge about 0.8 Tg N/yr into the global atmosphere. Of the 6.3 Tg N/yr of US NOx emissions, 2.7 Tg N/yr are deposited back onto the land and surface waters of the US. Thus, by difference we estimate that as much as 3.6 Tg N/yr per year of the NOx emissions are advected out of the US atmosphere. Similarly, of the 3.1 Tg N/yr of NH_3 that are emitted into the US atmosphere each year, about 2.1 Tg N/yr are deposited onto the land and surface waters of the US, and about 1 Tg N/yr is advected out of the US via the atmosphere. In sum, 5.4 Tg N are advected out of the US from all sources.

Riverine discharges of Nr to the coastal zone account for 4.8 Tg N/yr, while export of N-containing commodities (e.g., grain) removes another 4.3 Tg N/yr from the US. Altogether, along with 5.4 Tg N/yr of atmospheric advection, these total transfers to the environment add up to about 14 Tg N/yr, leaving about 21 Tg N/yr unaccounted for. Of this amount, we estimate that 5 Tg N/yr year are stored in soils, vegetation, and groundwater (as shown by the ‘Storage’ box in Figure 2), and, by difference, we estimate that about 16 Tg N/yr are denitrified to N_2 (Figure 2). Denitrification requires both a carbon source and anaerobic conditions, a situation that is found in wetlands and oxygen-depleted streams, rivers, soils, and other engineered denitrification systems. There are substantial uncertainties (+/- 50%) for some of these rough estimates -- especially those that involve NHx emission and deposition and terms that are arrived at by difference (e.g., atmospheric advection and denitrification). These significant uncertainties drive the three “Overarching Recommendations” of this report.

Figure 2: Nr inputs and outputs for the US in 2002 (Tg N/yr)



Explanatory notes:

- The left side of this figure shows the inputs into the US atmosphere from lightning and fossil fuel combustion, and into the US terrestrial system from biological nitrogen fixation in unmanaged landscapes (BNF), biological nitrogen fixation in cultivated landscapes (C-BNF), fertilizer production within the US (Fert prod), and imports of nitrogen-containing commodities. Not shown because the number is so small, is 0.2 Tg N/yr of NH_3 of Nr that is formed during fossil fuel combustion.

- The middle and right-hand parts of this figure show emissions of NO_y, NH_x and N_2O, and deposition of NO_y and NH_x to the US landscape. Transfers out of the US are shown as atmospheric advection of NO_y and NH_x (by difference), and of N_2O. The best estimate of advection of NOy plus NHx from the continent (export) are smaller than shown here. Nevertheless, these values are used for internal consistency among all media. See Section 2.3.

- NO_y, NH_x and N_2O are all components of Nr, but a fundamental difference is that the NO_y and NH_x are rapidly transferred from the atmosphere to receiving ecosystems due to a short atmospheric residence time (≤ 10 days) where they continue to contribute to the N cascade. Because of its longer residence time (~100 years) however, N_2O remains in the troposphere where it contributes to climate change, until it is transferred to the stratosphere, where it contributes to ozone depletion.

- The sum of the inputs is 35 Tg N/yr, the sum of the outputs is 14 Tg N/yr. The difference (missing Nr) is 21 Tg N/yr. As discussed in the text, we estimate that storage in soils, vegetation and groundwater is ~5 Tg N/yr. By difference, we estimate that formation of N_2 by denitrification is ~16 Tg N/yr.

- In this figure, freshwater wetlands, lakes, and rivers are included in the terrestrial box while coastal wetlands, lagoons, and other similar ecosystems are included in the coastal box.

Consequences, impacts, and metrics for Nr Use

Because nitrogen is both a critically important natural resource and also a contributor to a number of adverse environmental problems, it is imperative to understand how to decrease the risks to society while also providing the food, energy, and materials required by society. And, because agriculture depends on use of N fertilizers to support current levels of productivity on existing farmland, which reduces pressure to expand agriculture at the expense of rain forests and wetlands, efforts to mitigate the negative consequences of Nr cycling in crop and livestock systems must also consider potential tradeoffs on food production and land use. Fortunately there are opportunities for reducing the negative environmental impact from Nr use in agriculture while also sustaining the capacity to increase food production to meet increased demand expected from a larger and wealthier human population.

The most important beneficial consequence of Nr use in the United States (and other parts of the world) is providing adequate supplies of food, feed, fiber, and fuel crops to meet dietary and other needs of people in this country and abroad – an issue of global food security. In many ecosystems the supply of biologically available Nr is a key factor controlling adequacy of food, feed, and fiber supplies, the profitability of crop and animal agricultural, the nature and diversity of plant life, and vital ecological processes such as the cycling of carbon and soil minerals.

In addition to these important human-beneficial consequences, there are also numerous and important negative consequences from anthropogenic Nr. These negative consequences include formation of photochemical smog, exposure to toxic gases and aerosols in the air, acidification and eutrophication of terrestrial and aquatic ecosystems, climate change and other greenhouse effects, as well as stratospheric ozone depletion. Human activities have not only increased the supply but enhanced the global movement of various forms of Nr through air and water. Mitigating risk from these factors is difficult because one molecule containing Nr can contribute to all of these effects as a consequence of the nitrogen cascade. Nitrogen is a dynamic element easily transformed from one chemical form to another and is transported rapidly through and among ecosystem reservoirs. These characteristics make nitrogen an especially challenging element to control.

Reactive nitrogen plays a major role in climate forcing, but weather and climate also exert a profound influence on the public health and public welfare effects of Nr. Some climate predictions call for higher temperatures and increased intensity of rainstorms over North America – riverine discharge of nitrate increases with storm runoff and annual precipitation. Increasing the maximum summer temperatures can increase photochemical smog. Nitrogen deposition and carbon sequestration are linked, but the interactions are nonlinear and difficult to predict. This report calls for further investigation of the impact of climate on Nr, because it is probable that climate change will accentuate the need to control excess nitrogen.

Various approaches can be used to prevent, eliminate, decrease, or otherwise manage Nr risks. Understanding the environmental impacts of Nr can inform decisions on how best to manage nitrogen risks. There are two main approaches to characterizing the adverse public health and environmental impacts of Nr: traditional damage estimates and decreases in ecosystem services. Historically, EPA's environmental protection programs have addressed the adverse public health and public welfare impacts of Nr through use of such common metrics as National Ambient Air Quality Standards (NAAQS) and, in the case of water resources, Total Maximum Daily Loads (TMDLs) built upon attainment of water quality standards and criteria. These common metrics have had the considerable advantage of providing frameworks within which air and water quality standards could be derived that are protective of specific human health and environmental risks – the principal missions of EPA. The ecosystem services approach complements these traditionally used common metrics by considering how specific ecosystem services provided by one or more ecosystems are impaired by excess Nr. The ecosystem services approach can also recognize the important functions that wetlands and similar ecosystems can and do provide in decreasing Nr transfers to the environment. The attractiveness of this approach is its recognition that the health of humans and the health of ecosystems are inextricably linked. Less clear, in some cases, however, are practical ways in which to measure and monitor these adverse impacts.

Ecosystem-service-based measurements provide a richer context for the complex connections among Nr inputs and transformations. Furthermore, impacts on human well-being can help identify those adverse effects of Nr that impose the greatest damage costs to society.

The INC believes that using both traditional metrics and ecosystem service-based metrics will:

- Provide a clearer picture of priorities for action,
- Help identify effective control points for decreasing Nr impacts, and
- Provide insights into more efficient and cost-effective regulatory and non-regulatory strategies for decreasing Nr negative impacts.

Tradeoffs Among Nr Risk Management Options

Once the foreseeable impacts are understood and the suite of benefits associated with various risk reduction options is described, then managers can consider trade-offs. Risk reduction integration provides an intellectual framework that allows managers to make informed decisions about which benefits may need to be relinquished for other benefits when not all the desired benefits can be achieved. For example, limiting nitrogen fertilizer application to decrease risk from Nr applied to agro-ecosystems may decrease crop yields and increase food and feed commodity prices, which in turn may result in expansion of crop production area at the expense of natural wetlands, grasslands, and forests—a process called indirect land-use change.

Measurement of Reactive Nitrogen in the Environment

What air and water quality managers measure determines not only what they focus on but also how they gauge the success or failure of their environmental management strategies and tactics. Most regulations set limits or specify control technologies for specific forms of Nr without regard to the ways in which Nr may be transformed once it is introduced into the environment. Normally, regulations also require some form of monitoring to document compliance. However, monitoring of the specific chemical forms of Nr is not enough. There is a need to measure, compute, and report the total amount of Nr present in impacted systems in appropriate units because one chemical form of Nr can be quickly converted to other forms.

The impacts of reactive nitrogen often can be expressed as the economic costs of damages, the cost of remediation or substitution, or the cost/ton of remediation for each form of reactive nitrogen. Damage costs do not always scale as tons of Nr released into the environment. If damage costs rather than tons of Nr were utilized as a metric, the full implications of the cascade, and the setting of priorities for intervention might differ. Similarly, if human mortality and morbidity are the metrics used, priorities for decreases in Nr emissions could be very different.

To determine the extent of damage caused by excess nitrogen in environmental reservoirs, one needs to know both the present Nr concentration or loading within a reservoir and the threshold at which negative impacts are manifested. This threshold then provides a target load that can be used to guide strategies to decrease the amount of Nr in the reservoir. The thresholds for impacts are better known for some adverse impacts than others. For example, the impacts of ozone on human health are known well enough so that EPA has set standards for both ozone and for NO_2, an ozone precursor. The same can be said for the impacts of Nr discharge to coastal waters. Total Maximum Daily Loads (TMDLs) are used to link Nr loading to impact. On the other hand, the impact of Nr deposition on ecosystems is less well known. There is strong scientific evidence to show that Nr deposition rates of 10 – 20 kg N per hectare per year can cause negative impacts on a variety of ecosystems. Since a large part of the land surface in the northern hemisphere receives Nr deposition in that range, it is necessary to better define the link between Nr deposition and ecosystem response. Further, and related to the previous section, our knowledge of Nr deposition is uncertain, especially for the chemically reduced inorganic and organic forms of Nr. Our knowledge needs to be improved to better link deposition to ecosystem response and critical threshold.

Integrated Risk Reduction Strategies for Nr

Typically, quantitative risk assessment, technical feasibility, economic, social and legal factors, and additional benefits of various air and water management strategies contribute to the development of a suite of risk-reduction strategies from which managers can select an optimal approach.

Management Strategies for Nr

There are several ways in which the release and control of Nr in the environment can be approached. In general these can be classified as follows:

1. Improved practices and conservation—in which the flux of Nr that creates a detrimental impact is decreased through better management practices (e.g. on-field agricultural practices, control of urban runoff, controlled combustion conditions)

2. Product substitution—in which a product is developed or promoted which has a smaller dependency on Nr (e.g. use of switchgrass instead of corn grain as a feedstock for biofuel ethanol production).

3. Transformation—in which one form of nitrogen is converted to another less damaging form of nitrogen (e.g. nitrification of municipal wastewaters, denitrification of Nr by converting it back to non-reactive gaseous N_2 with created and restored wetlands).

4. Source limitation—in which the amount of Nr introduced into the environment is decreased (e.g., lower fertilizer application rates, use of catalytic converters and low-NOx burners in power plants).

5. Removal—in which particulate forms of Nr are captured in a more readily managed physical form such as sewage sludge which can be disposed of by land application or incineration.

6. Improved use or reuse efficiency—in which the efficiency of production that is dependent on Nr is improved (e.g. increased grain yields per unit of Nr fertilizer applied, decreased NOx emissions from improved diesel engines in trucks and off-road construction equipment, reuse of Nr-laden runoff to grow algae for other uses, such as bioenergy or animal feeds).

Efficient and cost-effective management of Nr often requires combinations of these six Nr management strategies; no one approach is a perfect alternative for decreasing excess Nr in the environment.

Policy Mechanisms for Management of Nr in the Environment

Generally speaking, US environmental policies employ one or more of the following four mechanisms for management of pollutants in the environment:

1. Command-and-Control—in which permitted limitations on pollutant or chemical-precursor emissions are issued under various regulatory statues. Violations may result in the assessment of penalties.

2. Government-based programs affecting the desirability of an environmental management mechanism, such as directed taxes, price supports for a given commodity, subsidies to bring about a particular end-result, and grants for capital expansion or improvement of pollution-abatement technologies.

3. Market-based instruments for pollution control in which cap and trade markets are used to bring about a desired policy end-result -- often at decreased overall cost to society.

4. Voluntary programs in which desired environmental outcomes are achieved using private or government-initiated agreements or through targeted outreach and education programs.

An integrated approach to the management of Nr will likely use a combination of these four implementation mechanisms. Each mechanism must be appropriate to the nature of the problem at hand, supported by critical research on decreasing the risks of excess Nr, and reflect an integrated policy that recognizes the complexities and tradeoffs associated with the nitrogen cascade. Management efforts at one point in the cascade may be more efficient and cost effective than control or intervention at another point. This is why understanding the nature and dynamics of the N cascade is critically important.

Major Findings and Recommendations

The ultimate goal of this report is to aid EPA in the development of an integrated N management strategy. To accomplish this, the committee recommends that EPA and other research organizations strengthen the science related to flows and impacts of Nr, that EPA use current knowledge to identify management actions that can be taken now, and that EPA join with other organizations to implement management actions within a framework that does not exacerbate one Nr problem when addressing another.

Recommendation A

An integrated approach to the management of Nr will likely use a combination of these four implementation mechanisms. Each mechanism must be appropriate to the nature of the problem at hand, supported by critical research on decreasing the risks of excess Nr, and reflect an integrated policy that recognizes the complexities and tradeoffs associated with the nitrogen cascade. Management efforts at one point in the cascade may be more efficient and cost effective than control or intervention at another point. This is why understanding the nature and dynamics of the N cascade is critically important.

Recommendation B

EPA should form an Intra-agency Nr Management Task Force that will build on existing Nr research and management capabilities within the Agency. This Intra-Agency Task Force should be aimed at increasing scientific understanding of: 1) Nr impacts on terrestrial and aquatic ecosystems, human health, and climate, 2) Nr-relevant monitoring requirements, and 3) the most efficient and cost-effective means by which to decrease various adverse impacts of Nr loads as they cascade through the environment.

Recommendation C

EPA should join with other agencies within the US government in establishing an Inter-agency Nr Management Task Force. The members of this Inter-Agency Task Force should include at least the following federal agencies: US Department of Agriculture (USDA), US Department of Energy (DOE), US Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), US Geological Survey (USGS), US Forest Service (USFS,) and Federal Emergency Management Agency (FEMA)). This Task Force should coordinate federal programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and managing Nr in the environment.

The intra- and inter-agency Nr-Management Task Forces should take a systems approach to research, monitoring, and evaluation in the following areas to inform public policy related to Nr management:

- Development of methods to help implement a systems approach
  - developing and evaluating proposed Nr budgets
  - developing appropriate life cycle accounting methods
  - developing monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads
  - evaluating the critical loads approach to air and water quality management
  - developing Nr indicators for excess Nr effects on human health and environment
  - developing new systems-based approaches for controlling Nr releases to the environment
- Enhancing ecosystem services that lead to the denitrification of Nr in the landscape including reconnecting rivers and streams to their floodplain, creation and restoration of wetlands in agricultural landscapes, and stream and ditch enhancements that increase the surface area of potential denitrification.
- Best management practices (BMPs)
  - developing the scientific understanding required for identifying best management practices (BMPs) for specific application, including:

        - Nr applications in agriculture to ensure adequate food, feed, fiber, and bioenergy feedstock supply while also avoiding negative impacts on the environment and human health;
        - Nr applications for developed (e.g., residential and commercial) runoff mitigation and landscape maintenance;
        - planning and pollution prevention including low impact development and natural ecosystem service preservation;
        - Enhancing the appropriate matching of crops, cropping systems, and land types and capabilities for the most productive use of Nr and the reduction of excess Nr
        - development and natural ecosystem service preservation;
        - primary use of natural land features and attributes, such as wetland preservation and enhancement, natural soil profiles and buffer strips;
        - improved removal of Nr from sewage waste streams at both large-scale wastewater treatment facilities and individual subsurface (septic) systems
    - establishing proactive extension and technology transfer approaches to facilitate adoption of BMPs
- Assessment activities
    - assessing combined carbon (C) and Nr effects on terrestrial and aquatic ecosystems
    - assessing indicators/endpoints, costs, benefits and risks associated with impairment of human health and decline and restoration of ecosystem services
    - reviewing existing and proposed legislation for purposes of extending Nr regulatory authority or streamlining procedures for enacting Nr risk reduction strategies
    - evaluating economic incentives, particularly those that integrate air, aquatic, and land sources of excess Nr
- Developing new education, outreach, and communication initiatives

In addition, INC makes four recommendations that set near-term targets for the decrease of Nr entering the environment from various sources.

(1) INC recommends that the EPA expand its NO_x control efforts from the current decreases of emissions of light duty vehicles (including passenger cars) and power plants to include other important unregulated mobile and stationary sources (e.g., off road vehicles) sufficient to achieve a **2.0 Tg N/yr** decrease in the generation of reactive nitrogen.

(2) INC recommends a goal of decreasing livestock-derived NH_3 emissions by 30% (a decrease of **0.5 Tg N/yr**) by a combination of BMPs and engineered solutions. This is expected to decrease $PM_{2.5}$ by ~0.3 μg/m^3 (2.5%), and improve health of ecosystems by achieving progress towards critical load recommendations. Additionally we recommend decreasing NH_3 emissions derived from fertilizer applications by 20% (decrease by **~0.2 Tg N/yr**), through the use of NH_3 treatment systems and BMPs.

(3) INC recommends that excess flows of Nr into streams, rivers, and coastal systems be decreased by approximately 20% **(~1 Tg N/yr)** through improved landscape management and without undue disruption to agricultural production. This would include activities such as using large-scale wetland creation and restoration to provide needed ecosystem services of Nr retention and conversion as well as matching cropping systems and intensity of Nr use to land characteristics.. Improved tile-drainage systems and riparian buffers on crop land, and implementing storm water and non-point source management practices (e.g., EPA permitting and funding programs) are other alternatives that are less proven. In addition, the committee recommends that crop N-uptake efficiencies be increased by up to 25% over current practices through a combination of knowledge-based practices and advances in fertilizer technology (such as controlled release and inhibition of nitrification). Crop output can be increased while decreasing total Nr by up to 20% of applied artificial Nr, amounting to ~**2.4 Tg N/yr** below current amounts of Nr additions to the environment. These are appropriate targets with today's available technologies and further progress is possible.

(4) INC recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA. This will decrease Nr emissions by between **0.5 and 0.8 Tg N/yr**.

INC is confident that implementing these recommendations will decrease the amount of Nr introduced into the US by about 25%, which will similarly decrease the amount of Nr lost to the atmosphere, soils and waters.

Epilogue

Human activities have significantly increased the introduction of Nr loads into the US environment. While there have been significant benefits resulting from increased food production, there also have been, and will continue to be, major risks to the health of both ecosystems and people due to the introduction of Nr into the nitrogen cascade. Regulations to set limits, require monitoring, and specify control technologies for nitrogen compounds should give careful consideration to the ways in which Nr may be transformed once it is introduced into the environment. To maintain the benefits and limit the negative impacts of excess of Nr, EPA should join with other federal agencies, States, universities, and private sector organizations in developing both an integrated total Nr management strategy, and a strong Nr public outreach and education program. The committee understands that there are real economic costs to the recommendations contained in this report. For each recommendation there will of necessity be tradeoffs derived from the varying cost-effectiveness of different strategies. It is vitally important that the recommendations, and the analysis of the associated trade-offs, be implemented and addressed in an integrated manner.

Chapter 1: Introduction

1.1 General background about environmental impacts of N loading

Water, water everywhere, and all the boards did shrink;

Water, water everywhere, nor any drop to drink.

This couplet from the *Rime of the Ancient Mariner* (Samuel Taylor Coleridge, 1772–1834) is an observation that, although sailors were surrounded by water, they were dying of thirst because of its form. Just as water is a critical substance for life, so is N. And just as most of the water on the planet is not useable by most organisms, most of the N is also unavailable.

Approximately 78% of the atmosphere is diatomic nitrogen (N_2), which is unavailable to most organisms because of the strength of the triple bond that holds the two N atoms together. Over evolutionary history, only a limited number of species of bacteria and archaea have evolved the ability to convert N_2 to Nr via biological N fixation. However, even with adaptations to use N efficiently, many ecosystems of the world are limited by N.

This limitation has driven humans to use increasingly sophisticated and energy-intensive measures to obtain Nr to sustain food production and to produce other commodities (e.g., nylon, explosives). In the beginning, hunters and gatherers harvested food from natural stocks. With the advent of agriculture, local sources of Nr were used (soil stocks, crop residue, manures) to increase productivity of landscapes. In the 19th century, long range transport of Nr to sustain food production increased by shipping bird guano from the Pacific Islands and nitrates from South America to Europe and other locations. By the beginning of the 20th century, these sources were not sufficient to sustain the growing global population requirements for food.

This deficiency led to what has been called one of the world's most important discoveries—how to extract N_2 from the atmosphere and convert it to NH_3—the Haber-Bosch process (Smil, 2001; Erisman et al., 2008). Today this process and cultivation-induced biological N fixation (C-BNF) introduce over 140 teragrams (Tg) of N per year (hereafter expressed as Tg N/yr) into the global environment to sustain food production. Another 23 Tg N/yr are introduced by the Haber-Bosch process for the chemical industry, and 25 Tg N/yr are introduced via the combustion of fossil fuels.

The total global anthropogenic Nr creation rate is ~190 Tg N/yr (2005), substantially larger than the median of estimates for Nr creation by natural terrestrial processes (~100 Tg N/yr) (Galloway et al., 2008). The fact that humans are more effective than nature in Nr creation means that on average, humans are less reliant on natural sources of Nr. However, with global commodity stocks running at a 58 day supply and food prices increasing dramatically, the challenge is to increase the nutrient use efficiency of Nr in agricultural systems while maintaining or increasing yields (USDA ERS/World Agricultural Outlook Board, July 11, 2008. World Agricultural Supply and Demand Estimates).

There are large regional disparities in Nr creation rates on both absolute and per capita bases. Total Nr creation is larger in Asia than in any other region. Per capita Nr creation is largest in North America and Europe. Humans also redistribute large amounts of Nr among countries or regions of the world through exports of fertilizers, feed grains, and fossil fuels. Nevertheless, there are large regions of the world with populations approaching one billion, where there is malnutrition due to a lack of adequate supply of available Nr to sustain crop production, among other reasons.

The introduction of Nr into most regions of the United States (US) by humans has greatly increased food availability. However, since essentially all the Nr created for food production and by fossil fuel combustion is lost to the environment, it has also greatly increased Nr's contribution to a wide variety of environmental problems. Most plants, animals, and microorganisms are adapted to efficiently use and retain small increments of additional Nr. Addition of Nr to most ecosystems may first lead to increased uptake, growth, storage, and hence to increased biomass, including food or fiber production. However, further addition of Nr in excessive amounts often leads to imbalances in the movement of Nr among reservoirs and potential transfers to the environment in the form of air emission or water discharges into other ecosystems where it may disrupt ecosystem functions and have a negative impact on resources. In essence, the assimilative capacity of the ecosystem may be insufficient to benefit from increases in Nr without disruptive change. While there will always be Nr transfers to the environment during food production, the challenge is how to minimize those transfers while meeting the demand for food production.

Negative consequences of Nr flux in the US environment include increases in photochemical smog and $PM_{2.5}$, decreases in atmospheric visibility, both increases and decrease in productivity of grasslands and forests, acidification of soils and freshwaters, accelerating estuarine and coastal eutrophication, increases in the emission of greenhouse gases to the atmosphere, and decreases in stratospheric ozone concentrations. Most of these changes in environmental conditions lead to a variety of negative impacts on both ecosystem and human health. These changes, which impact air, land, water and the balance of life in an interrelated fashion, are often referred to a cascade of effects from excess Nr[4] or the "nitrogen cascade" (Figure 3). Unlike other element-based pollution problems, the N cascade links the negative impacts, where one N-containing molecule can in sequence contribute to all the environmental issues mentioned above.

The nitrogen cascade has three dimensions:

- biogeochemical,
- environmental changes and
- human and ecosystem consequences (Figure 3).

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[4] Excess reactive nitrogen (Nr) is defined as the amount of Nr that is present in, or introduced into, an environmental system (e.g., Nr inputs to the atmosphere, Nr inputs to grasslands and forests, N_r inputs to estuaries) from anthropogenic sources that is not incorporated into agricultural and other biological products (e.g., food, feed, fuel and fiber), or stored in long-term storage pools (e.g., cropland soils).

Thresholds are used to determine at what amount excess Nr causes negative effects on ecosystem services and functions, and human health. Thresholds vary by metric (e.g., concentration, loading, etc) and depend on the environmental system (e.g., atmosphere, forest). Examples for specific thresholds are given later in the report in relevant sections.

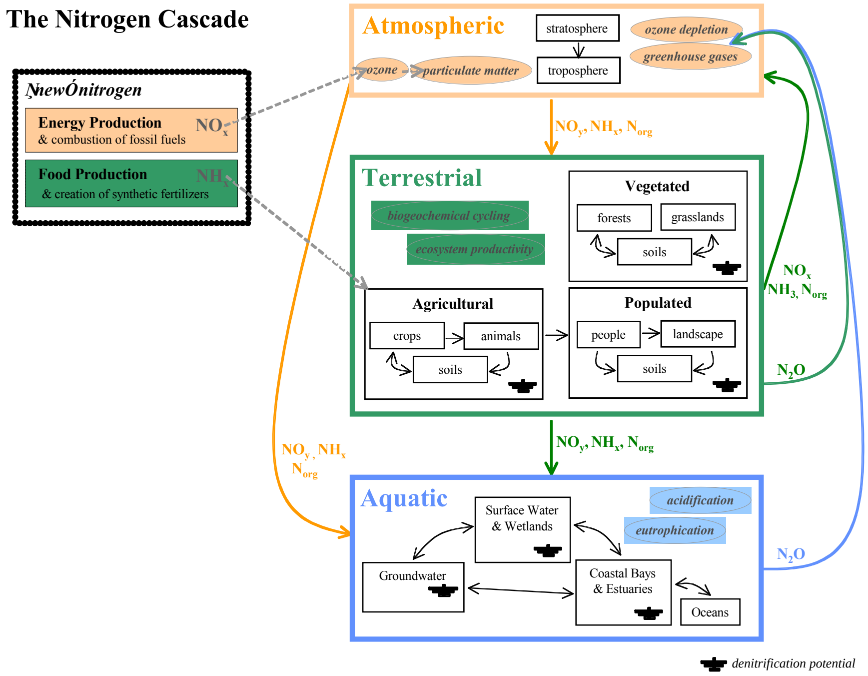
The "biogeochemical" dimension of the nitrogen cascade involves Nr creation from N_2 as a consequence of chemical, food and energy production, Nr use in food and chemical production, Nr transfers to the environment, changes in Nr species residence times in environmental reservoirs, Nr transfers among reservoirs and Nr conversion back to N_2. Environmental changes then result from increased Nr levels in the environment. These environmental changes have negative consequences for ecosystem and human health at local, regional, national and global scales. Because nitrogen is a critical resource and also a contributor to many of the environmental concerns facing the US today, it is imperative to understand how human action has altered N cycling in the US, and the consequences of those alterations on people and ecosystems. The over-arching question is how do we protect and sustain an ecosystem that provides multiple benefits to society while also providing the interconnected material, food and energy required by society.

In summary, Nr inputs to the nation and the world have been increasing, largely due to human activities associated with food production and fossil fuel combustion. Despite the obvious benefits of a plentiful supply of food and energy, the adverse consequences associated with the accumulation of Nr in the environment are large, with implications for human health and the environment.

The greater the inputs of Nr to the landscape, the greater the potential for negative effects, caused by greenhouse gas production, ground level ozone, acid deposition, and Nr overload that can contribute to climate change, degradation of soils and vegetation, acidification of streams, lakes and rivers, estuarine and coastal eutrophication, hypoxia and habitat loss.

The growing nature of the Nr problem, and the adverse and intertwined consequences associated with Nr inputs to air, land, and water as exhibited in the N cascade underscore the need for researchers and managers to explore integrated strategies that minimize N inputs, maximize its use efficiency, promote Nr removal processes and protect humans and natural resources.

Figure 3: The Nitrogen Cascade



The concept of the nitrogen cascade highlights that once a new Nr molecule is created, it can, in sequence, travel throughout the environment contributing to major environmental problems (Galloway et al., 2003). This adaptation of the cascade was developed by the Integrated Nitrogen Committee to provide a context for considering nitrogen-related issues and ecosystem effects in the US. To consider the cascading effects of Nr in the US, we examine the relative sizes of the various atmospheric, terrestrial, and aquatic environmental systems where Nr is stored, and the magnitudes of the various flows of N to, from, and within them. The nitrogen cascade concept implies the cycling of Nr among these systems. The process of denitrification is the only mechanism by which Nr is converted to chemically inert N_2, 'closing' the continuous cycle. Denitrification can occur in any of the indicated reservoir except the atmosphere.

The "new" N box depicts the two primary anthropogenic sources by which Nr originates, energy production and food production, and where they enter ecosystems. Energy production includes both fossil fuel and biofuel combustion. Food production includes N fertilizer produced in the US, cultivation-induced biological N (C-BNF) in the US, production of animals and crops in the US for human consumption, and imports of N-containing fertilizer, grain and meat to the US.

The atmospheric system indicates that tropospheric concentrations of both ozone and particulate matter are increased due to NO_x emissions to the atmosphere. The ovals illustrate that the increase in N_2O concentrations, in turn, contribute to the greenhouse effect in the troposphere and to ozone depletion in the stratosphere. Except for N_2O, there is limited Nr storage in the atmosphere. Transfers of Nr from the atmospheric system include NO_y, NH_x, and organic nitrogen (Norg) deposition to terrestrial and aquatic ecosystems of the earth's surface. There is little potential for *conversion of Nr to N_2 via denitrification in air. However, once airborne deposition of Nr occurs it will be subject to denitrification pathways via soil and water.*

The terrestrial system depicts that Nr enters agricultural lands via food production and is introduced to the entire terrestrial landscape via atmospheric deposition. Within agricultural regions there is cycling between soils, crops and animals, and then a transfer of Nr as food to populated regions, from which there are Nr transfers to the environment (e.g, sewage, landfills). The ovals showing ecosystem productivity and biogeochemical cycling reflect that Nr is actively transported and transformed within the terrestrial system, and that as a consequence there are significant impacts on ecosystem productivity due to fertilization and acidification, often with resulting losses of biodiversity. There is ample opportunity for Nr storage in both biomass and soils. Losses of Nr from this system occur by leaching and runoff of NOy, NHx and Norg to Aquatic ecosystems and by emissions to Atmospheric system as NO_x, NH_3, Norg, and N_2O. There is potential for conversion of Nr to N_2 via denitrification in the terrestrial system.

The aquatic system shows that Nr is introduced via leaching and runoff from terrestrial ecosystems and via deposition from atmospheric ecosystems. Connected with the hydrological cycle, there are Nr fluxes downstream with ultimate transport to coastal systems. Within the aquatic system, the ovals highlight two significant impacts of waterborne Nr—acidification of freshwaters and eutrophication of coastal waters. Except for Nr accumulation in groundwater reservoirs, there is limited Nr storage within the hydrosphere. Transfers of Nr from the aquatic system are primarily via N_2O emissions to the atmospheric system. There is a very large potential for conversion of Nr to N_2 via denitrification in water and wetlands.

NO_y, NH_x and N_2O are all components of Nr, but a fundamental difference is that the NO_y and NH_x are rapidly transferred from the atmosphere to receiving ecosystems due to a short atmospheric residence time ($\leq$10 days) where they continue to contribute to the N cascade. Because of its longer residence time (~100 years) however, N_2O remains in the troposphere where it contributes to climate change, until it is transferred to the stratosphere, where it contributes to ozone depletion.

1.2 Overview of EPA research and risk management programs in context of other environmental management and research programs

The mission of the Environmental Protection Agency is to protect human health and the environment. In achieving this mission, EPA is accountable for addressing five goals given in the *2006 – 2011 EPA Strategic Plan*:

1. Clean air and global climate change,
2. Clean and safe water,
3. Land preservation and restoration,
4. Healthy communities and ecosystems, and
5. Compliance and environmental stewardship.

The *Strategic Plan* includes targets for reducing risk from N. EPA's *Report on the Environment* (ROE), provides "data on environmental trends," to determine whether or not EPA is on track to meet its targets and goals. EPA is responsible and accountable for reducing at least some risks from reactive N.

The principal mechanisms for Nr removal from circulation in the environment are complete denitrification (re-conversion of Nr back to non-reactive gaseous N_2), and storage in long-term reservoirs (e.g., soils, sediments, and woody biomass). In some cases, it may be possible to capture Nr emissions or discharges and deliver them to food or fiber production areas where there are nitrogen deficiencies. However, major challenges in the management of the N cycle are how to decrease creation of Nr while still meeting societal needs, promote denitrification of excess Nr (without producing N_2O), and improve the efficiency of use and reuse of excess Nr in a cost-effective manner. Solving both these challenges will result in less Nr accumulation

The parts of EPA most directly concerned are the Office of Air and Radiation, the Office of Water, and the Office of Research and Development. Programs designed to save energy, such as Energy Star, tend to reduce emissions of Nr as well.

EPA's Office of Air and Radiation reduces risk from Nr in over a dozen programs including National Ambient Air Quality Standards (NAAQS) standard setting and implementation; emission standards for industrial stationary sources and area sources; the Acid Rain Program; the Clean Air Interstate Rule; and programs that focus on mobile source emissions. EPA's Office of Water addresses Nr under both the Clean Water Act and the Safe Drinking Water Act with activities such as; criteria development and standard setting; NPDES permits; watershed planning; wetlands preservation; and regulation of Concentrated Animal Feeding Operations (CAFOs).

EPA's Office of Research and Development's mission is to conduct leading-edge research and foster the sound use of science and technology in support of EPA's mission. ORD is well recognized for providing a scientific basis for the development of the NAAQS standards for NO_x and particulate matter (PM). ORD's revised Multi-Year Plan for Ecological Research will identify and quantify the positive and negative impacts on ecosystem services resulting from changes in nitrogen loadings from major source categories to support policy and management decisions in EPA's Offices of Air Resources and Water.

EPA has brought a great variety of risk reduction tools to bear on reactive N: conventional regulation and enforcement; cap and trade approaches; measurement, monitoring and place-based approaches; control technology development and verification; communication and education; intergovernmental and international cooperation, and voluntary approaches. The variety and breadth of EPA programs addressing Nr reflect the ubiquity of Nr in the environment, the historical single medium regulatory approach, and the lack of a silver bullet for reducing risks from N.

Recommendation #8 of the Science Advisory Board's 1990 report, *Reducing Risk*, was:

EPA should increase its efforts to integrate environmental considerations into broader aspects of public policy in as fundamental a manner as are economic concerns. Other Federal agencies often affect the quality of the environment, e.g., through the implementation of tax, energy, agricultural, and international policy, and EPA should work to ensure that environmental considerations are integrated, where appropriate, into the policy deliberations of such agencies.

In the current era of increasing responsibilities without commensurate budgets, intergovernmental cooperation, partnerships and voluntary programs have become vital tools for agencies needing to stretch their resources to fulfill their missions.

1.3. The need for integration

Some impacts of N on ecosystems and people have been known for centuries (e.g., impacts of atmospheric deposition on agricultural crops); others for only a few decades (e.g., impact of N_2O on stratospheric ozone). Notwithstanding current uncertainties, the EPA and its predecessor organizations have been active in the management of Nr for a variety of reasons, including decrease in the Nr amount in sewage, control of NO_x to decrease photochemical smog and acid rain, control of Nr inputs to coastal systems, controls on fine particulates in the atmosphere and decrease in Nr leaching and runoff from crop and animal production systems. As beneficial as those efforts have been, they focus on the specific problem without consideration of the interaction of their particular system with other systems downstream or downwind. Given the reality of the nitrogen cascade, this approach may result in short term benefits for a particular system but will also likely only temporarily delay larger scale impacts on other systems. Thus there is a need to integrate N management programs, to ensure that efforts to lessen the problems caused by N in one area of the environment do not result in unintended problems in other areas.

1.4 Charge and scope of this report

In 1973 the Science Advisory Board issued its first report, the 200-page *Nitrogenous Compounds in the Environment*. The report addressed sources and effects of nitrogenous compounds, including those from air emissions, animal wastes, crop agriculture, industrial processes, and solid wastes. After concluding that, "At present, all known trends appear to be ones that can be managed and kept within control, if appropriate steps are taken now," the SAB provided recommendations relating to Nr research on and control. Later, the SAB would consider the1970s to be the first step in environmental protection, characterized by broad agreement on environmental problems and their sources. The second step emerged in the 1980s when the risk assessment/risk management paradigm proposed by the National Research Council in 1983 achieved wide acceptance.

In *Toward Integrated Environmental Decision-Making*, published in 2000, the SAB articulated a third step in environmental protection -- the framework for integrated environmental decision-making. In this 2000 report, the SAB noted that the 3-phase structure (problem formulation, analysis & decision-making, followed by implementation and evaluation), "belies the complexities involved in putting the concept of integrated decision-making into practice."

The SAB's interests in N science and integrated environmental protection converged in 2003, when the SAB identified integrated N research and control strategies as an important issue facing the Agency and formed the Integrated Nitrogen Committee to undertake a study of this issue. The charge to the committee was to:

1. Identify and analyze, from a scientific perspective, the problems N presents in the environment and the links among them;

2. Evaluate the contribution an integrated N management strategy could make to environmental protection;
3. Identify additional risk management options for EPA's consideration; and
4. Make recommendations to EPA concerning improvements in N research to support risk reduction.

In the course of its study, the Integrated Nitrogen Committee held four public face-to-face meetings at which it invited briefings from EPA's Office of Air and Radiation, Office of International Affairs, Office of Research and Development, and Office of Research and Development; from the Department of Agriculture's Agricultural Research Service, Cooperative State Research, Extension and Education Service, and the Economic Research Service; and from external organizations such as the Energy Research Centre of the Netherlands, Environmental Defense Fund, International Plant Nutrition Institute, Iowa State University, LiveFuels, and the Soil and Water Conservation Society.

Additionally, the INC invited scientists and managers from EPA, other federal agencies, states and localities, academia, non-governmental organizations and the private sector to participate in a October 20-22, 2008 Workshop Meeting on Nitrogen Risk Management Integration.

Chapter 2: Behavior of reactive nitrogen in the environment

This chapter identifies and analyzes, from a scientific perspective, the problems Nr presents in the environment and the links among them.

2.1 Introduction

The aquatic system shows that Nr is introduced via leaching and runoff from terrestrial ecosystems and via deposition from atmospheric ecosystems. Connected with the hydrological cycle, there are Nr fluxes downstream with ultimate transport to coastal systems. Within the aquatic system, the ovals highlight two significant impacts of waterborne Nr—acidification of freshwaters and eutrophication of coastal waters. Except for Nr accumulation in groundwater reservoirs, there is limited Nr storage within the hydrosphere. Transfers of Nr from the aquatic system are primarily via N_2O emissions to the atmospheric system. There is a very large potential for conversion of Nr to N_2 via denitrification in water and wetlands.

This chapter of the report addresses three aspects of the committee's work. The first two are the introduction of Nr into US systems from fossil fuel combustion and from food production (Section 2.2) and the fate of Nr after it is emitted to the atmosphere by fossil fuel combustion or lost to the air, water and soils from agricultural production systems (Section 2.3). The third aspect is the impacts of Nr on humans and ecosystems (Section 2.4) from both a traditional view (i.e., specific effects such as impacts of smog on people and plants) and a more integrated view (i.e., the consequences of Nr on ecosystem services).

The issues of Nr in the US environment revolve around the introduction of new Nr by imports, fertilizer production, C-BNF, and fossil fuel combustion, and by its distribution within agricultural system and populated systems and redistribution through losses from those systems to the environment (Figure 3). National-level values for Nr fluxes are displayed in Table 1. Those fluxes that represent the introduction of new Nr into the US are marked with an asterisk. Specific sections of the report will use these values to more clearly determine the flux and fate of Nr in the US.

Table 1: Nr fluxes for the US, Tg N in 2002[a]

Nr inputs to the *Atmospheric* environmental systems	Tg N/yr	%
N_2O-N emissions	0.8	8
Agriculture - livestock (manure) N_2O-N	0.03	
Agriculture - Soil management N_2O-N	0.5	
Agriculture - field burning agricultural residues	0.001	
Fossil fuel combustion - transportation*	0.1	
Miscellaneous	0.1	
NH_x-N emissions	3.1	31
Agriculture: livestock NH_3-N	1.6	

Agriculture: fertilizer NH_3-N	0.9	
Agriculture: other NH_3-N	0.1	
Fossil fuel combustion - transportation*	0.2	
Fossil fuel combustion - utility & industry*	0.03	
Other combustion	0.2	
Miscellaneous	0.1	
NO_x-N emissions	6.2	61
Biogenic from soils	0.3	
Fossil fuel combustion - transportation*	3.5	
Fossil fuel combustion - utility & industry*	1.9	
Other combustion	0.4	
Miscellaneous	0.2	
Total *Atmospheric* inputs	10.0	100
Nr inputs to the *Terrestrial* environmental system		
Atmospheric N deposition[b]	6.9	19
Organic N	2.1	
Inorganic no_y-N	2.7	
Inorganic-nh_x-N	2.1	
*N fixation in cultivated croplands	7.7	21
Soybeans*	3.3	
Alfalfa*	2.1	
Other leguminous hay*	1.8	
Pasture*	0.5	
Dry beans, peas, lentils*	0.1	
N fixation in non-cultivated vegetation*	6.4	15
N import in commodities*	0.2	0.3
Synthetic N fertilizers*	15.1	41
(9.4 produced in US*, 5.8 net imports to US*		

Fertilizer use on farms & non-farms	10.9	
Non-fertilizer uses such as explosives	4.2	
Manure N production	6.0	16
Human waste N	1.3	3
Total *Terrestrial* inputs	43.5	100
Nr inputs to the *Aquatic* environmental system		
Surface water N flux	4.8	

Table 1 Notes

a. The Nr estimates in this table are shown with two significant digits or 0.1 million metric tons N per year (or Tg N/yr) to reflect their uncertainty; occasionally this report will show data to more significant digits, strictly for numerical accuracy. Obtaining quantitative estimates of each of the Nr terms and the associated uncertainty, remain a major scientific challenge.

b. Reducing the uncertainty in total deposition of atmospheric Nr to the surface of the 48 contiguous US remains a scientific and policy priority. Based on observations and models, we estimate 5.9 (range 4 ─ 9) Tg N/yr total anthropogenic Nr deposition to the entire 48 States (Section 2.3.1.10). The EPA sponsored Community Multiscale Air Quality (CMAQ) Model run yielded a value of 4.8 Tg N/yr. The value shown for the total (6.9 Tg N/yr) reflects the assumption that organo-nitrogen species should be added to the model estimate as 30% of the total.

* Terms with an asterisk indicate Nr that is created, highlighting where reactive nitrogen is introduced to the environment.

Table 1 Data Sources:

- Emissions, N_2O-N (EPA Inventory of US Greenhouse Gas Emissions and Sinks)
- Emissions, NH_x-N (EPA National Emissions Inventory)
- Emissions, NO_x-N (EPA National Emissions Inventory)
- Atmospheric deposition, organic N (30% of total atmospheric N deposition, Neff et al. 2002)
- Atmospheric deposition, inorganic NO_y-N & NH_x-N (EPA CMAQ model)
- N_2 fixation in cultivated croplands (USDA census of agriculture, literature coefficients)
- N_2 fixation in non-cultivated vegetation (Cleveland and Asner, unpublished data)
- Synthetic N fertilizers (FAO & AAPFCO)
- Non-fertilizer uses such as explosives (FAO)
- Manure N production (USDA census of agriculture, literature coefficients)
- Human waste N (US Census Bureau population census, literature coefficients)
- Surface water N flux (USGS SPARROW model; long-term flow conditions)

2.2 Sources of Nr new to the US environment

2.2.1 Introduction

Creation of "new" Nr in the environment refers to Nr that is either newly fixed within or transported into the US. This "new" Nr highlights where Nr is introduced into ecosystems. New Nr arises from fossil fuel combustion, food production and materials production (Table 1).

Fossil fuel combustion emits Nr (mostly NO_x) to the atmosphere[5]. Fossil fuel combustion introduces 3.5 Tg N/yr and 1.9 Tg N/yr of NO_x-N to the atmosphere from transportation, and utility/other industry sources, respectively (Table 1). Another 0.2 Tg N/yr of NH_3-N and 0.1 Tg N/yr of N_2O-N is emitted from the same sources (Table 1). Thus the total amount of Nr created by fossil fuel combustion is 5.7 Tg N/yr, of which > 90% is in the form of NO_x-N.

Synthetic Nr fertilizers are typically produced by the Haber-Bosch process and used primarily in agriculture to support food production. Production of fertilizers within the US introduces Nr into US terrestrial landscapes at the rate of 9.4 Tg N/yr, and net imports of fertilizer via world trade introduce 5.8 Tg N/yr. Of this total (15.2 Tg N/yr), 9.8 Tg N/yr is used as fertilizer on farms and 1.1 Tg N/yr is used on non-farms (i.e., residential and recreational turf-grass and gardens, and in explosives used by the mining industry), and 4.2 Tg N/yr is introduced for non-fertilizer uses, such as for production of plastics, fibers, resins, and for additives to animal feed (Table 1).

Additional Nr is introduced into the US from cultivation-induced biological nitrogen fixation (BNF) by agricultural legume crops such as soybean and alfalfa (7.7 Tg N/yr), and from imports of N contained in grain and meat (0.15 Tg N/yr) (Table 1).

Thus in 2002, anthropogenic activities introduced a total of 29 Tg N into the US, mostly in support of food production, although turf production, industrial uses and fossil fuel combustion were also important sources. Natural sources of Nr in the US are BNF in unmanaged landscapes, and lightning. The former contributes 6.4 Tg N/yr (Table 1) and the latter 0.1 Tg N/yr. Clearly, anthropogenic activities dominate the introduction of Nr into the US.

Transfers of Nr to the environment in the US occur during fossil fuel combustion and food production. The former occurs immediately, as Nr formation during combustion is inadvertent and the Nr, primarily as NO_x, is emitted directly into the atmosphere. The latter occurs through all stages of food production and consumption. The remaining sections of Section 3.2 document the magnitude of the transfers to the environment from the various components of both energy and food production.

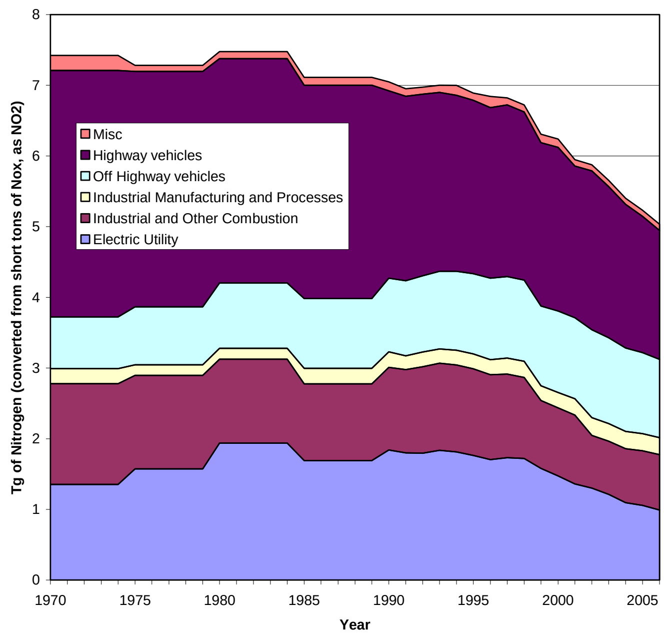
2.2.2 Nr formation and transfers to the environment from fossil fuel combustion

Fossil fuels such as coal, petroleum, and natural gas provide about 80% of all energy production (based on year 2000). When these fuels are burned at high temperatures, NO_x is formed. The source of N is either the N contained in the fossil fuel or the N_2 that comprises about 80% of atmosphere. Fuel-derived N is important in the case of burning coal (which contains N), while atmospheric-derived N_2 is formed during higher temperature processes that occur when gasoline or diesel fuel is burned in motor vehicles (Table 1). In the US, highway motor vehicles account

[5] Nr is generally not formed during combustion of wood and modern biomass because of lower combustion temperatures.

for the largest anthropogenic source of NO_x at 36% (Figure 4), while off-highway vehicles, electric utilities and industrial processes account for 22%, and 20%, respectively.

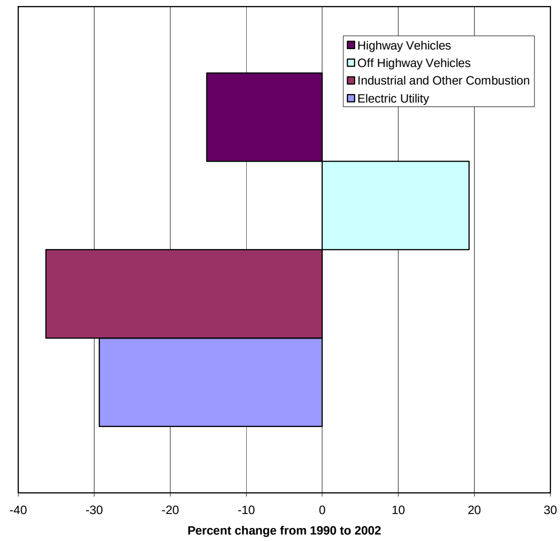
Figure 4: US NO_x emission trends, 1970-2006. Data are reported as thousand of metric tons of N converted from NO_x as NO_2



(Source: http://www.epa.gov/ttn/chief/trends/index.html)

Figure 4 also illustrates that the amount of NO_x (reported as metric tons of N) released from various fossil fuel sources has decreased dramatically from 1970. Total emissions were on the order of 7,400 metric tons in 1970, decreased to 5900 in 2002, with further decreases in 2006 to 5,030 metric tons. Overall this represents a decrease of over 30%. The top sources (highway vehicles, off-highway vehicles, electric utilities, and other industrial and combustion systems) show decreases between 15-30% from 1990 to 2002 (Figure 5). Reductions were the highest for "other" systems followed by electric utilities. These decreases are most likely the result of changes in regulations and control technologies for these stationary systems. To a lesser extent, changes in highway vehicle regulations and the removal of older fleets from the road has resulted in a decrease of approximately 15%. This decrease however, is accompanied by an increase in miles traveled, which suggests that the actual decrease in a single vehicle is larger. Off highway vehicles showed an increase in emissions, potentially due to better quantification of these sources. Sources here include locomotives, marine engines, etc. While some regulations are in place for some of these sources, such as locomotives, further control of these and other sources could decrease emissions. In fact, technological development in the locomotive industry shows that decreases of approximately 70% are possible. Further decreases would require more innovative, expensive methods such as Selective Catalytic Reduction (SCR) with urea injection. Engine manufacturers are also investigating using SCR systems for diesels. However, it must be noted that these systems emit small amounts of NH_3 and must be operated properly to avoid trading off NO_x emissions for NH_3.

Figure 5. Percent reductions in NO_x emissions, 1990-2002, from different sources (off-road, on-road vehicles, power generation, etc.)



It should be noted that it is difficult to control nitrogen emissions with regulations on one source solely. As seen in Table 2 (data taken from 2001 for illustrative purposes), Texas' fuel combustion sources are on the same order as highway vehicles; this is in comparison to California, where vehicles, highway and off-highway are the dominant source (over 75%) for this state. These results are attributed to industries and coal-fired power plants located in Texas. Almost 40% of the power generation in Texas is due to coal-fired plants. On the other hand, California imports most of its coal-fired power and generates its own power predominantly from other sources, such as natural gas (50%), hydro and nuclear (33%). Florida, Ohio, and Illinios are also shown. The emissions of NOx from highway vehicles is likely related to population. For example, the estimated population of California for 2006 is 36.4 million people versus Ohio and Illinois which are on the order of 11-12 million.

Table 2: Examples of multiple sources from states with high NO_x emissions (based on 2001 data; and tons of NO_x as NO_2)

(Source: These data were derived from the 2001 information obtained at: http://www.epa.gov/air/data/geosel.html)

	TX	CA	FL	OH	IL
Fuel Combustion-Electric Util.	91,441	8,441	87,489	93,792	59,124
Fuel Combustion - Industrial	98,978	31,237	11,792	17,300	26,481
Fuel Combustion - Other	9,222	21,407	5,707	12,974	10,894
Industrial Processes	25,584	13,786	5,933	8,123	7,122
Highway Vehicles	164,937	182,471	116,889	83,593	78,278
Off-Highway Vehicles	106,162	85,064	38,475	46,239	52,797
Miscellaneous Sources	4,807	7,882	13,110	1,526	999
TOTAL, metric tons	**501,151**	**350,301**	**279,778**	**263,561**	**235,817**

2.2.3 Nr inputs and transfers to the environment from crop agriculture

Agriculture uses more Nr and accounts for more Nr transfers to the environment than any other economic sector. Synthetic fertilizers are the largest sources of Nr input to agricultural systems. The next largest source is cultivation-induced BNF (Table 1). The major pathways by which Nr is lost from these systems include NO_3^- transfers from leaching, runoff and erosion and gaseous emissions via volatilization of NH_3 and NOx and nitrification/denitrification. Similar transfer pathways occur for Nr that cycles through livestock systems, which also account for a large portion of Nr flux (predominantly as NH_3) in animal agricultural systems (Aneja et al. 2006). Therefore, assessment of Nr impacts on the environment and development of strategies to minimize negative impact should be based on a thorough understanding and accurate accounting of Nr fluxes in both crop and livestock systems, and the trends in management practices that have greatest influence on Nr transfers to the environment from these systems (Aneja et al, 2008a,c).

In the past 60 years, N fertilizers have had a beneficial effect on agriculture both nationally and globally by increasing crop yields. However, the high loading of Nr from agricultural nutrient sources has lead to deleterious effects on the environment, such as decreased visibility from increased aerosol production and elevated N concentration in the atmosphere, ground, and surface waters (Galloway et al. 2003).

2.2.3.1 Nitrogen fertilizer use

Obtaining accurate data on fertilizer use is a critical first step in understanding Nr cycles in agriculture. There are several sources of data reporting fertilizer usage but it is not clear whether data quality is sufficient for assessing environmental impact. Although the Uniform Fertilizer Tonnage Reporting System (UFTRS) was developed to collect fees to fund the consumer protection mission of State Chemists and fertilizer regulatory control officials, it also provides data on fertilizer sales in many states, which in turn are used by many agencies and environmental scientists to estimate consumption and use of nitrogenous fertilizers in the US. The Association of American Plant Food Control Officials (AAPFCO) tallies and publishes the statewide fertilizer sales data annually (Terry et al. 2006), which is one of the most widely used sources of data on fertilizer use. It is typically assumed that fertilizers are used in the same region in which they were sold. The annual state-level data published by AAPFCO, which are based on commercial fertilizer sales and often taxed at the state level (but not in all states), are the only data source available. This state-level data source includes fertilizer sales for both agricultural and non-agricultural purposes. These state-level data must then be allocated to counties, regions, or watersheds in the states, and the algorithms used for this process are based on a number of assumptions that address dealer/farmer storage, inventories, and cross-state sales issues (personal communication, Stan Daberkow, USDA-ERS).

The USDA National Agricultural Statistics Service Information (NASS) fertilizer usage data represents another source of information derived from farmer "agricultural chemical use" surveys that provide information in six categories: field crops, fruits and vegetables, nurseries/floriculture, livestock use, and post-harvest application. For each group, NASS collects fertilizer, pesticide, and pest management data every year on a stratified random sample of farmers at the field level (http://usda.mannlib.cornell.edu/usda/current/AgriChemUsFC/AgriChemUsFC-05-16-2007_revision.pdf). The NASS report represents another useful data source but also would require extrapolation across reported crop acreage to represent a complete sample of application rates.

The UFTRS was not designed to track the source of inorganic nutrients applied to agricultural land on the geographic scale needed for watershed modeling. The system only tracks sales of synthetic fertilizers and not manure or biosolids applied to farmland. In addition, geographical data associated with each sale may or may not be near the actual point of application. However, given either regulatory or legislative changes (data reporting is mandated through each state's fertilizer law), it could be possible to refine the current system used by each state Department of Agriculture to generate more precise data for improved modeling of watershed-scale nutrient mass balances. Those changes would help target interventions and extension programs to improve nutrient management and reduce nutrient losses. The lack of potential funding and the necessity to coordinate all the states involved limit the practicality of such an approach.

The state Departments of Agriculture have already made recommendations to improve the reporting system. These include:

1. an assessment to determine the needs for fertilizer usage data, the accuracy of the current data collection methods, and whether methods require revision to meet highest priority needs,
2. improvements in the database format and web-based access,
3. The identification of funding sources to support development of a more accurate, accessible, and comprehensive database system, and
4. Education and outreach to improve precision of reported fertilizer tonnage including a clear distinction between nutrients used in crop, livestock, and non-agricultural operations.

In addition, the information could be refined to reflect site-specific data layers, although that would require development of a geospatial framework (and legal authority) to encourage reporting at the retail level where it is possible to collect geographic information.

The Chesapeake Bay watershed provides a good example of the fertilizer data dilemma. While the fertilizer tonnage that is currently being utilized to calibrate the Chesapeake Bay Program Watershed Model is relatively accurate, the county-specific tonnage may have an accuracy of only ±20 to 50%. For example, in a recent year, 17% of the reported tonnage had been reported without an identified use and there are indications some tonnage may have been reported more than once through the distribution chain (Chesapeake Bay Scientific Technical Advisory Committee report, Oct. 2007). It is also possible that fertilizer reported for crop agriculture may actually have been used for lawn and turf, forestry, or other non-agriculture applications.

Nitrogen fertilizer application data on a specific crop-by-crop basis that can be associated with crop yields and location are essential for assessing both use patterns and efficiency. The USDA-NASS maintains a database on N fertilizer rates applied to the major crops (corn, wheat, cotton, soybeans, and occasionally other crops) based on farmer surveys conducted every other year. These data represent another source of information ("Protocols for Farming Reporting" Mark R. Miller, USDA, NASS).

Data derived from NASS farmer surveys include six categories: field crops, fruits and vegetables, nurseries/floriculture, livestock use and post-harvest application. For each group, NASS collects fertilizer, pesticide, and pest management data every year on a stratified random sample of farmers at the field level. One field represents an entire farm for each sample in the field crops survey. Fruit and vegetable information are collected for the entire farm. If the field chosen for sampling has had manure applied in conjunction with inorganic fertilizer, only the inorganic portion will be reported because the survey does not ask about manure. Core crops are surveyed every other year on an even/odd basis for different crops and surveyed states are selected to cover at least 80% of planted acres. NASS is not currently scheduled to resume coverage of corn and other commodity crops until 2010, which is a five year gap. NASS will try to resume its coverage of corn management survey in 2008. This is a critical data gap and it is a problem given the large changes in corn price and production area during this period. Those data have to be available if there is to be progress in assessing fertilizer use and efficiency for major crops in the US USDA NASS must resume their yearly data collection for commodity crops (Chemical Use Survey). Potential environmental impacts of increased N inputs associated with expanded corn acreage for biofuel production cannot be properly evaluated in the absence of such critical nutrient management data.

Based on these data, USDA has recently released an updated report on fertilizer use that provides data on fertilizer consumption and type of fertilizer used from 1960-2006 (Figure 6) and types of fertilizers used (Table 3). (US Fertilizer Use and Price; Released Friday, October 5, 2007). Share of crop area receiving fertilizer and fertilizer use per receiving acre, by nutrient, are presented for the major producing states for corn, cotton, soybeans, and wheat. Additional data include fertilizer farm prices and indices of wholesale fertilizer price. See http://www.ers.usda.gov/Data/FertilizerUse/

Figure 6: Fertilizer consumption in the US 1960 to 2006

(Source: AAPFCO; 1960 - 2006. www.aapfco.org)

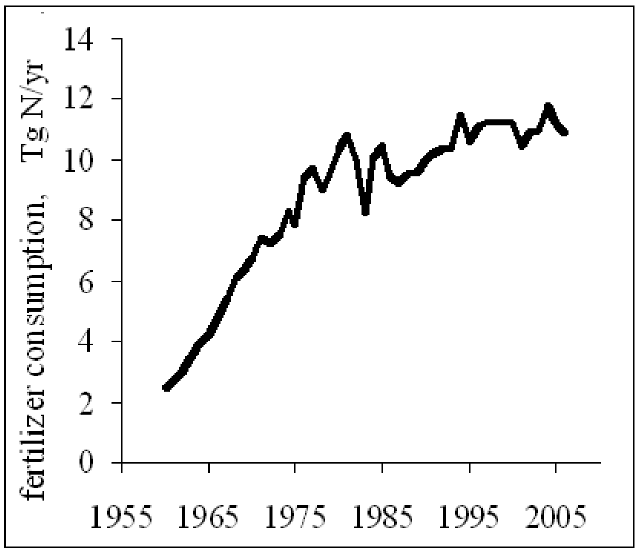


Table 3: Sources and amount of nitrogen fertilizers used in the US in 2002.

[Data from Terry et al. (2006)]

Synthetic Nitrogen Fertilizers	Tg N/year	% of total
Other	0.21	2
Urea	2.21	20
N Solutions	2.55	23
Anhydrous NH_3	2.88	26
DAP, MAP, and NPK blends	2.28	32
NH_4^+ SO_4^{2-}, NH_4^+, Thiosulfate, and Aqua NH_3 and NH_4^+ Nitrate	0.76	7
Total	**10.89**	**100**

Finding 1

Crop agriculture receives 63% of US annual new Nr inputs from anthropogenic sources (9.8 Tg from N fertilizer, 7.7 from crop BNF versus 29 Tg total) and accounts for 58% (7.6 Tg, see Table 1) of total US Nr transfers from terrestrial systems to air and aquatic ecosystems, yet current monitoring of fertilizer use statistics by federal agencies is inadequate to accurately track trends in quantities and fate of N applied to major crops and the geospatial pattern by major watersheds.

Recommendation 1: *Increase the specificity and regularity of data acquisition for fertilizer application to major agricultural crops in terms of timing and at a sufficiently small application scale (and also for urban residential and recreational turf) by county (or watershed) to better inform decision-making about policies and mitigation options for reducing Nr load in these systems, and to facilitate monitoring and evaluation of impact from implemented policies and mitigation efforts.*

2.2.3.2 Nitrogen fertilizer use efficiency

Nitrogen fertilizer use efficiency (NFUE) is critical because higher use efficiency leaves less N remaining to create potential environmental problems. Here and throughout this report we define NFUE as the grain yield per unit of applied N, which is the product of two parameters: (i) the proportion of applied N fertilizer that is taken up by the crop, or N fertilizer recovery efficiency [(RE) in kg N uptake per kg N applied)], and (ii) the physiological efficiency with which the N taken up by the crop is used to produce economic yield such as grain or fruit [(PE), kg yield per kg N uptake](Cassman et al., 2002)] All else equal, when higher NFUE is achieved without yield reduction, the crop takes up more of the applied N and incorporates it into its biomass, which leaves less of the applied Nr at risk for loses via leaching, volatilization, or denitrification. Fixen (2005) reports that there is substantial opportunity for increasing NFUE through development and adoption of more sophisticated nutrient management decision aids.

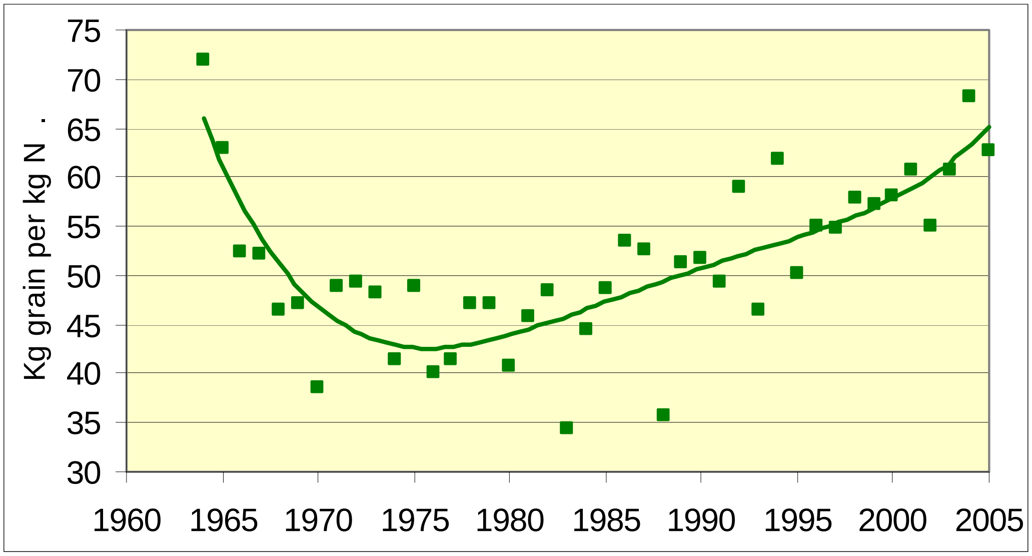
In most cropping systems, RE is the most important determinant of NFUE. A recent review of RE for cereals based on field studies around the world, mostly conducted on “small-plot” experiments at research stations, reported mean single year RE values for maize, wheat and rice of 65%, 57% and 46%, respectively (Ladha et al., 2005). However, crop RE values based on actual measurements in production-scale fields are seldom greater than 50% and often less than 33%. For example, a review of RE in different cropping systems, estimated average recoveries of 37% for maize in the north central US (Cassman et al., 2002). It is also important to note that soil N provides the majority of the N taken up by most crops grown on soils with moderate to good soil fertility. For maize in the U.S. corn belt, for example, 45-77% of total N uptake was estimated to come from soil N reserves based on experiments from research stations (Sawyer J, Nafziger E, Randall G, Bundy L, Rehm G, and Joern B. 2006. Concepts and Rationale for Regional Nitrogen Rate Guidelines for Corn. Iowa State Extension PM 2015, www.extension.iastate.edu/Publications/2015.pdf). Therefore highest N efficiency and economic return on N inputs are achieved when the amount and timing of applied N is synchronized with the availability of soil N throughout the growing season to minimize both the quantity of N input required and the N losses from soil and applied N sources.

However, there are relatively few data that provide direct measurement of N fertilizer recoveries by our major field crops under production-scale conditions and reducing the uncertainty in estimates of N fertilizer RE is fundamental for prioritization of research and education investments, both in the public and private sectors. While management can substantially improve RE on average, in any given year weather will always be an uncontrolled factor that can significantly influence system efficiency through effects on crop growth vigor and ability to acquire applied nutrients, and on losses of nutrients due to runoff, denitrification, and leaching that can occur in periods of excessive rainfall.

Although total N fertilizer use in the US has leveled off in the past two decades (Figure 6), yields of all major crops have continued to increase. Because crop yields are closely related to N uptake (Cassman et al., 2002), these trends imply a steady increase in NFUE and reduced N transfers to the environment because more of the applied N is held in crop biomass and harvested grain. Greater NFUE has resulted from two factors. The first factor is a steady improvement in the stress tolerance of corn hybrids (Duvick and Cassman, 1999) that increase crop growth rates and allows sowing at higher plant densities, which together accelerate the establishment of a vigorous root system to intercept and acquire available N in the soil profile. The second factor is the development and adoption of technologies that may improve the congruence between crop N demand and the N supply for indigenous soil resources and applied N. Examples of such technologies include soil testing for residual nitrate and adjusting N fertilizer rates accordingly, split N fertilizer applications, fertigation (the application of nutrients through irrigation systems), site-specific management, and new fertilizer formulations (e.g. controlled release, nitrification inhibitors). For maize, which receives the largest share of total N fertilizer in the US (44%, 2005), NFUE has increased by more than 50% from 1974-76 to 2002-05 (Figure 7).[6] Similar improvements have been documented for rice production in Japan and for overall crop production in Canada.

[6] N fertilizer use efficiency (NFUE) is calculated as the ratio of grain yield to the quantity of applied N fertilizer (kg grain/kg applied N).

Figure 7: Trends in Corn Grain Produced per Unit of Applied Fertilizer-N in the U.S. (USDA data)



Despite these steady improvements, current levels of N fertilizer uptake efficiency appear to be relatively low (Cassman et al., 2002), although data from production-scale studies are few (Cassman et al., 2002). Most farmers do not use best management practices (BMPs) with regard to nitrogen fertilizer management. For example, a recent USDA-ERS AREI report indicates that a majority of farmers still apply N in the fall, which gives the lowest fertilizer uptake efficiency and highest Nr losses compared to application in spring or during the crop growth period (USDA, ERS, 2006). This situation suggests substantial potential for improvement in NFUE and an associated reduction in Nr losses from crop agriculture, especially for maize in the warmer portions of the Corn Belt and other southern and southeast areas where maize is grown. One potential development is the use of controlled release fertilizers that emit N in congruence with crop demand during the growing season. Although such fertilizers are already in use on high value horticultural crops, they are currently too expensive for lower value commodity grains such as corn, rice, or wheat. Such enhanced efficiency fertilizers can increase NFUE where there is high risk for N losses in cereal systems that receive the total amount of applied N in one or two large doses.

As producers have increased yields in commodity crops significantly over the past 25 years, the question arises whether university recommendations for nutrient applications are still current. Many university recommendations are now 20 to 25 years old. As a corollary to this problem, numerous environmental models of nutrient pollution are still utilizing older yield estimates, which often underestimate crop nutrient uptake and overestimate nutrient losses (Robert Burgholzer, cited in Understanding Fertilizer Sales and Reporting Information, *Workshop Report, Oct. 2007).*

A systematic effort needs to be made to update those data. The concept of NFUE should be emphasized as a way to address the need to balance economic *and* environmental goals. In fact, the development and adoption of technologies that improve nitrogen fertilizer efficiency can contribute to more profitable cropping systems through a reduction in fertilizer costs. For example, average NFUE in the US required 1.0 kg of applied N to produce 43 kg of grain yield in the 1974-76 period, whereas that same amount of N produced 65 kg of grain in 2003-05 period (data taken from Figure 7). This gain in efficiency means that it is possible to achieve the 2004 US average corn yield of about 150 bushels per acre with 144 lbs of applied N fertilizer based on the most recent NFUE achieved by U.S. corn producers, versus about 200 lbs of N fertilizer at the 1980 efficiency level. At a cost of $0.40 per pound of applied N, this reduction in N fertilizer input requirements represents a saving of about $22 per acre.

Nitrogen costs have become extremely volatile, mirroring natural gas prices. In late 2008, Nitrogen fertilizer prices were more than double 2006/7 Nitrogen prices. More recently, Nitrogen fertilizer prices have fallen back to two thirds of the high following the decline of natural gas prices. If corn brings $4.00 per bushel (25.5 kg) and nitrogen costs $0.40 a pound (0.45 kg), this is a 10 to 1 price ratio – not different from the $2.00 corn and $0.20 nitrogen ratio that was typical from 2000-2005. There are also other critical factors in the farmer's nitrogen application decisions such as yield at the margin and weather. In the corn belt, one or two years in five may provide extremely favorable weather for corn production. A producer may view applying some extra nitrogen, hoping for good weather, as a reasonable economic gamble. If the yield is more than half a bushel (12.7 kg) of corn per pound (0.45 kg) of N at the margin or if there is more than one extremely good year in five, the farmer comes out ahead.

Realistically, few farmers calculate their marginal returns from additional N in good years versus average, but the high corn-to-fertilizer price ratio encourages some farmers to plan for a good year and consider a larger N application than might otherwise be appropriate for the N utilization in the four years of lower yield. This presents a real dilemma if the policy goal is to reduce N transfers to the environment, especially in the four years of average or lower yields. Meeting this challenge will require approaches such as the development of real-time, in-season, decision-making tools that allow crop producers to use N fertilizer rates for average yields at planting and during early vegetative growth, and a final top-dressing as required to meet any additional N demand above this amount due to favorable climate and soil conditions that support higher than average yields (Cassman, 1999; Cassman et al, 2002). Robust crop simulation models using real-time climate data at a relatively localized geographic scale will be required to develop such tools.

Another option is to develop new, alternative crop production systems that require less N fertilizer. Such systems may employ legume cover crops, more diverse crop rotations, and tighter integration between crop and livestock production to achieve greater reliance on N inputs from legume N fixation and recycling of N in manure and compost. At issue, however, is whether such systems actually reduce Nr transfers to the environment because the same loss mechanisms and pathways operate on N from both commercial fertilizer and organic sources. Also at issue is the indirect land use change impact from widespread adoption of these more diverse cropping systems because they have reduced crop yields per unit land area compared to more simplified crop rotations such as corn-soybeans that receive N fertilizer. Lower yields would require more land in production to meet food demand. Therefore, a key issue is whether the tradeoff in reduced N fertilizer inputs to more diverse crop rotations with organic N inputs would actually result in less Nr transfers to the environment compared to conventional cropping systems that require less land to produce the same amount of crop output.

<u>Finding 2</u>

Nr inputs to crop systems are critical to sustain crop productivity and soil quality. Moreover, given limited land and water resources, global population growth and rapid economic development in the world's most populous countries, the challenge is to accelerate increases in crop yields on existing farm land while also achieving a substantial increase in N fertilizer uptake efficiency. This process is called "ecological intensification" because it recognizes the need to meet future food, feed, fiber and energy demand of a growing human population while also protecting environmental quality and ecosystem services for future generations (Cassman, 1999). More diverse cropping systems with decreased Nr fertilizer input may also provide an option on a large scale if the decrease in Nr losses per unit of crop production in these diverse systems can be achieved without a decrease in total food production, which would trigger indirect land use change to replace the lost production and negate the benefits.

Recommendation 2:

a) *Data on NFUE and N mass balance, based on direct measurements from production-scale fields, are required for the major crops to identify which cropping systems and regions are of greatest concern with regard to mitigation of Nr load and to better focus research investments, policy development, and prioritization of risk mitigation strategies.*
b) *Promote efforts at USDA and land grant universities to: (i) investigate means to increase the rate of gain in crop yields on existing farm land while increasing N fertilizer uptake efficiency and (ii) explore the potential for more diverse cropping systems with lower N fertilizer input requirements so long as large-scale adoption of such systems would not cause indirect land use change.*
c) *EPA should work closely with the US Department of Agriculture (USDA), Department of Energy (DOE), and the National Science Foundation (NSF), and land grant universities to help identify research and education priorities efficient use and mitigation of Nr applied to agricultural systems.*

2.2.3.3. Biological fixation in cultivated croplands.

Reactive nitrogen is also introduced to the landscape in significant quantities via BNF in cultivated crop lands. Management of biologically fixed N, insofar as it is possible, is proportionally as critical a task as the management of synthetic N because Nr from BNF is prone to the same loss pathways as Nr from commercial fertilizers. To quantify BNF due to human cultivation of crops, the committee calculated the annual agricultural fixation for 2002 using crop areas and yields reported by the Census of Agriculture (2002). The committee multiplied the area planted in leguminous crop species by the rate of N fixation specific to each crop type, assigning rates based on a literature review, as summarized in Table 4 below and shown relative to other inputs in Table 1. Annual nitrogen inputs to cropping system from BNF by legume crops was 7.7 Tg N/yr in 2002, accounting for ~15% of the overall Nr inputs to the terrestrial landscape from all sources and 20% of the agricultural sources (Table 1). Soybean and alfalfa contributions are the most important agricultural legumes in terms of nitrogen input and contribute 69% of total BNF inputs in US agriculture.

Table 4: Estimates of nitrogen input from biological nitrogen fixation (from major legume crops, hay, and pasture)

Nr fixation in cultivated croplands

	production area, Mha	rate, kg/ha/yr	Tg N/yr	% of total
Soybeans	29.3	111	3.25	42
Alfalfa	9.16	224	2.05	27
Other leguminous hay	15.4	117	1.80	23
Western pasture	161	1	0.16	2
Eastern pasture	22.0	15	0.33	4
Dry beans, peas, lentils	0.88	90	0.08	1
Total			**7.67**	**100**

*Updated estimate for soybean based on a generalized relationship between soybean yield and the quantity of N fixation (Salvagiotti et al., 2008). Other values are from Boyer et al. (2002).

2.2.3.4. Emissions factors and transfers to the environment from fertilizers and organic nitrogen sources.

Agriculture is a significant contributor of Nr inputs into the atmosphere. Nitrogen fertilizer losses vary greatly due to differences in soil properties, climate, and the method, form, amount, timing and placement of applied nitrogen (Cassman et al., 2002). In addition, any factor that affects crop growth vigor and root system function also affects the ability of the plant to recover applied N efficiently. For example, denitrification can range from 0-70% of applied N (Aulakh et. al, 1992). This process is mediated by heterotrophic, facultative anaerobic soil bacteria that are most active under warm, wet soil conditions; they have low activity in dry sandy soils.

Despite this variation, watershed, regional and national assessments of carbon and N cycling often rely on average values for losses from each pathway. For example, the Intergovernmental Panel on Climate Change (IPCC) assumes that 1% of applied N fertilizer (uncertainty range of 0.3-3.0%) is lost from direct emissions of N_2O at the field level due to denitrification, based on analysis of all appropriate scientific publications that report these losses for specific crops and cropping systems (IPCC, 2007). The same 1% default emission factor for field-level N_2O emission is applied to other N inputs from crop residues, organic amendments such as manure, and from mineralization of native soil organic matter. Data from scores of field studies were used to obtain this average value. A number of recent studies confirm that N_2O transfers to the environment during the growing season at the field level represent <1% of the applied nitrogen—even in intensive, high-yield cropping systems (Adviento-Borbe et al., 2006). Despite these average values, it is also clear that N_2O transfers can vary widely even within the same field and from year to year due to normal variation in climate and crop management (Parkin and Kaspar, 2006; Snyder, 2007). Moreover, the loss of nitrogen from agricultural watersheds is strongly dependent on climate change (e.g. rainfall changes). Predicted increases and decreases in rainfall will likely have a dramatic impact on nitrogen export from agricultural fields. For example, precipitation is predicted to increase in the upper Mississippi watershed, and other factors being equal, N export should increase (e.g., Justic et al., 1995b).

Additional indirect N_2O emissions result from denitrification of volatilized NH_3 deposited elsewhere or from NO_3^- lost to leaching and runoff as the Nr cascades through other ecosystems after leaving the field to which it was applied. Here the IPCC assessment protocol assumes that volatilization losses represent 10% of applied N, and that N_2O emissions for these losses are 1% of this amount; leaching losses are assumed to be 30% of applied nitrogen and N_2O emissions are 0.75% of that amount (IPCC, 2007). Therefore, the IPCC default value for total direct and indirect N_2O emissions represents about 1.4% of the applied N from fertilizer. By the same calculations, 1.4% of the N in applied organic matter, either as manure or compost, or in recycled crop residues, is also assumed to be emitted as N_2O.

Others have estimated higher average N_2O losses of 3-5% of applied nitrogen fertilizer based on global estimates of N_2O emissions from recycling of Nr (Crutzen et al., 2008), as opposed to the field-based estimates that form the basis of IPCC estimates. Because N_2O is such a potent greenhouse gas, and given the more than 2-fold difference in estimates of N_2O losses, there is a critical need to improve understanding and prediction of N_2O losses from agricultural systems. N_2O emissions in the US are estimated to be 0.78 Tg N/yr (Table 5) (EPA, 2005).

Table 5: N_2O emissions in the US, 2002

	Tg N/yr	%
Agricultural Soil Management	0.54	69
Manure Management	0.03	4
Mobile Combustion	0.09	12
Stationary Combustion	0.03	4
Nitric & Adipic Acid Production	0.05	6
Wastewater Treatment	0.02	2
Other	0.02	2
Total	**0.78**	**100**

Biogenic NO_x emissions from croplands are on the order of 0.5% of fertilizer input—much more than this in sandy soils and less as clay content increases (Aneja et al. 1996; Sullivan et al. 1996;

Veldkamp and Keller. 1997; Civerolo and Dickerson, 1998). However, NO_x emissions by agricultural burning are relatively unimportant. Ammonia volatilization of N from applied fertilizer can be the dominant pathway of N loss in rice soils and can account for 0->50% of the applied N depending on water management, soil properties and method of application (citations within Peoples et al. 1995). Ammonia volatilization can be of the same range in upland cropping systems, with largest losses occurring typically on alkaline soils (Peoples et al. 1995). The IPCC (2007) uses a value of 10% of synthetic fertilizer N application and 20% of manure N as estimates of average NH_3 volatilization.

Taken together, N losses from all forms of direct gaseous emissions forms from crop production systems can represent a substantial portion of applied N fertilizer when soil conditions favor such emissions and there is a lack of synchrony between the amount of N applied and the immediate crop demand (Goulding, K., 2004). Therefore, achieving greater congruence between crop demand and the N supply from fertilizer is a key management tactic to reduce N losses from all sources. Success in reducing N losses and emissions from agriculture will depend on increased efforts in research and extension to close gaps in our understanding of N cycling and management in crop production, especially as systems further intensify to meet rapidly expanding demand for food, feed, fiber, and biofuel.

Finding 3

Nitrous oxide emissions from the Nr inputs to cropland from fertilizer, manure, and legume fixation represent a large proportion of agriculture's contribution to greenhouse gas emissions, and the importance of this source of anthropogenic greenhouse gas will likely increase unless NFUE is markedly improved in crop production systems. Despite its importance, there is considerable uncertainty in the estimates of nitrous oxide emissions from fertilizer and research should focus on reducing this uncertainty.

Recommendation 3: *The committee recommends that EPA ensure that the uncertainty in estimates of nitrous oxide emissions from crop agriculture be greatly reduced through the conduct of EPA research and through coordination of research efforts more generally with other agencies such as USDA, DOE, NSF and with research conducted at universities.*

2.2.3.5. Impact of biofuel production capacity on Nr flux in agriculture

The enormous use of liquid fuels in the US, the rising demand for petroleum-based liquid fuels from countries like China and India, and the decline in petroleum discovery all contributed to the recent record high petroleum prices. In addition most of the world's petroleum reserves are located in politically unstable regions. Together these factors have provided strong motivation for policies promoting investment in biofuels made from corn, oil crops, and ultimately from cellulosic materials. In the US, ethanol production capacity from corn has more than doubled to over 47 billion liters/year (January 2009) in just three years since 2006. The renewable fuels standard in the 2007 Energy Independence and Security Act (EISA) will support another 9.5 billion liters/year of corn based ethanol by 2015. An additional 79.5 billion liters is to come from cellulosic ethanol by 2022. Biodiesel from vegetable oils also is encouraged in EISA, but expansion has been slowed by the high food value of such oils. Brazil is rapidly expanding its production of relatively low cost sugarcane ethanol and US policies continue to be aimed at bringing about increased future biofuel production in the US.

In 2007 and 2008 petroleum prices pushed ethanol prices high enough to draw corn from food and feed uses into ethanol production and contribute to the increased price of corn. Because of the increase in petroleum/ethanol prices and the government subsidy for ethanol production, 30% of the corn crop ended up going to ethanol in 2008 (Abbott, et. al. 2008). With the subsequent collapse in petroleum and ethanol prices, followed by corn prices, we have had unused capacity in the US ethanol industry as the corn/ethanol price ratio made ethanol production uneconomic for some firms. However, EISA is likely to lead to the production of cellulosic materials and even some expanded corn production for biofuels once the US gets beyond the current blending limit for ethanol (Doering & Tyner, 2008).

The higher corn prices of 2007 and 2008 resulted in more land being planted to corn and higher total N fertilizer use because much of the expansion in corn area came at the expense of soybeans. As a legume, soybeans do not require N fertilizer because legume crops can biologically fix N from dinitrogen gas (N_2) in the atmosphere. Corn acreage went from 32 million ha in 2006/7 to 38 million ha in 2007/8. About 5 million ha of the expansion was from a reduction in soybeans, and the remaining new corn area came primarily from reduced cotton, hayland and pasture. This strong response to high demand for biofuel feedstock has led to concern about increased pressure on the environment from biofuels One important factor is the increased N input that is necessary for growing corn and cellulosic materials (Robertson, et. al. 2008). Expansion of corn or cellulosic materials production into marginal lands can be even more problematic with respect to nutrient leaching and soil erosion. Changes in N fertilizer prices add uncertainty to the additional amounts of N that may ultimately be used in biofuel feedstock production. Production of large amounts of distillers grains co-product is also changing the way in which livestock feed rations are formulated, which in turn could have a large influence on the cycling of N in cattle manure (Klopfenstein et al., 2008).

Finding 4

Rapid expansion of biofuel production is changing the cost-benefit ratio of N fertilizer use in crop production and also changing the nutrient profile of livestock diets with consequences for effective management of Nr.

Recommendation 4: Rapid expansion of biofuel production has the potential to increase N fertilizer use through expansion of corn production area and associated N fertilizer inputs, and from extending cultivation of cellulosic materials that will also need N inputs. Distiller's grains are changing animal diets and affecting N recycling in livestock. Both have important consequences for the effective future management of Nr.

2.2.4. Nr inputs and losses from animal agriculture

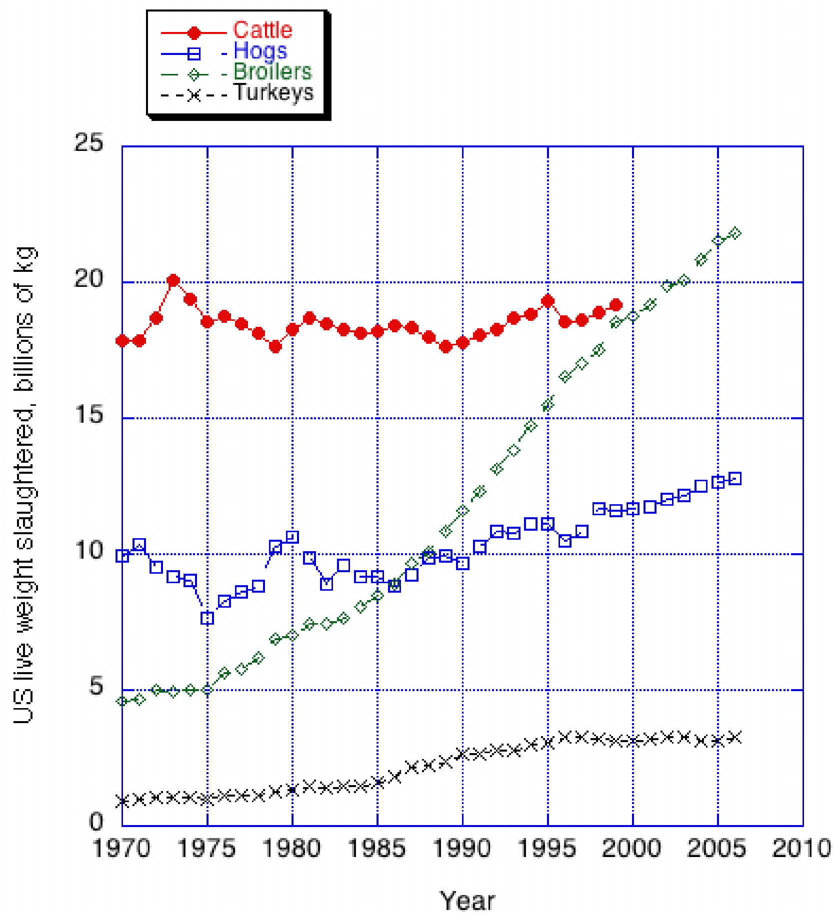
In the US, domestic animals produce 6.0 Tg N/yr in manure and are the largest source of atmospheric NH_3-N (1.6 Tg N/yr) (Table 1). Livestock also contribute to N_2O-N emissions, though in much smaller proportions (~4% of total US N_2O-N emissions).

2.2.4.1 Trends in Animal Agriculture

While animal production has been increasing since World War II, this report will emphasize the period from 1970 to 2006. The production of chicken broilers increased by more than four fold from 1970 to 2006 (Figure 8) and milk production increased by nearly 60% in this time period (Figure 9). Turkey production doubled and pork production increased about 25%, while meat from cattle (beef and dairy) remained constant (Figure 8).

<u>Figure 8: Meat production from 1970 to 2006. Source: USDA-NASS, Census Reports</u>

(Data on cattle not taken after 1999)



Another trend in animal production has been for fewer animals to produce more animal products. For example, the 60% greater amount of milk produced in 2006 compared to 1970 required 25% fewer cows (Figures 9 and 10). Animal inventories declined by 10% for beef brood cows from 36 million head in 1970 to 33 million head in 2006, and the inventory of breeder pigs and market hogs declined 8% from 673 million head to 625 million head in the same period, even with similar or greater annual meat production. This trend resulted from greater growth rates of animals producing more meat in a shorter amount of time. In 1970, broilers were slaughtered after 80 days on feed at 1.7 kg live weight, but by 2006 the average weight was 2.5 kg after only 44 days on feed (NASS-USDA, 2007).

Figure 9: Milk production from 1970 to 2006. Source: USDA-NASS, Census Reports.

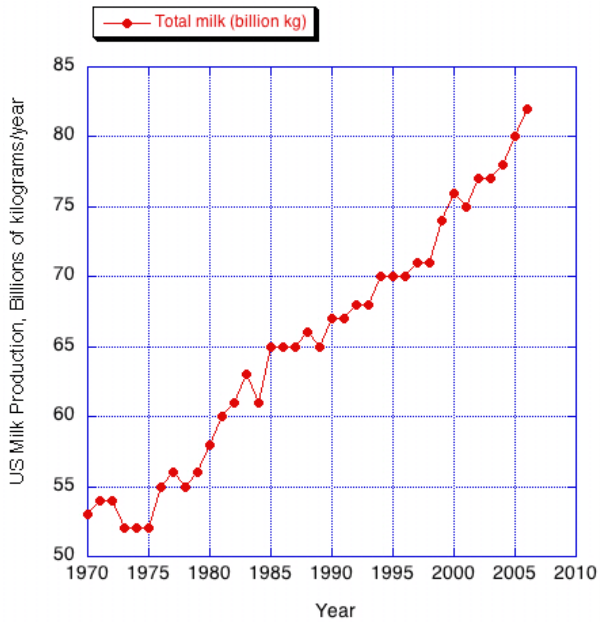
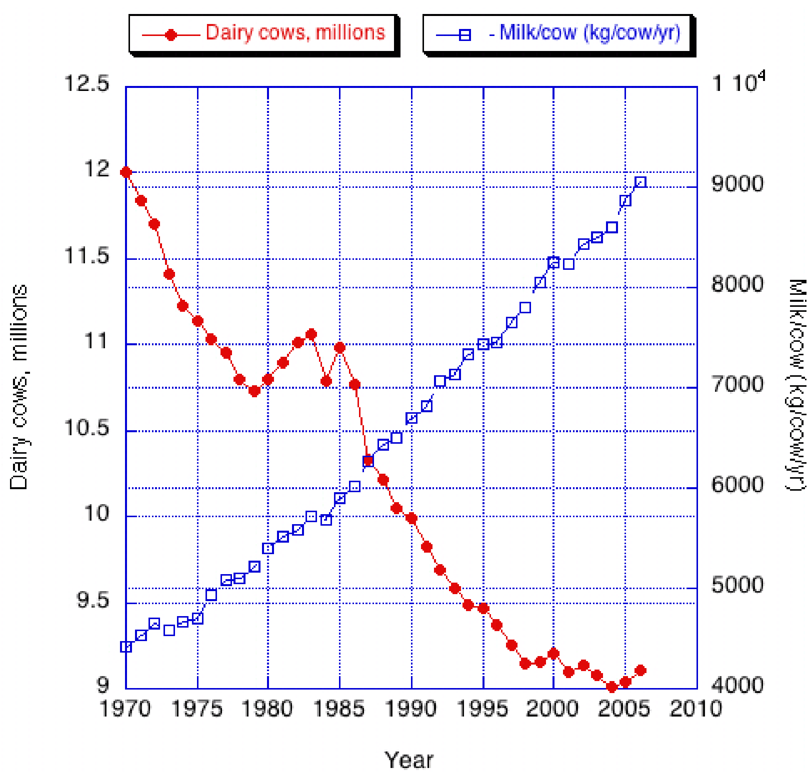
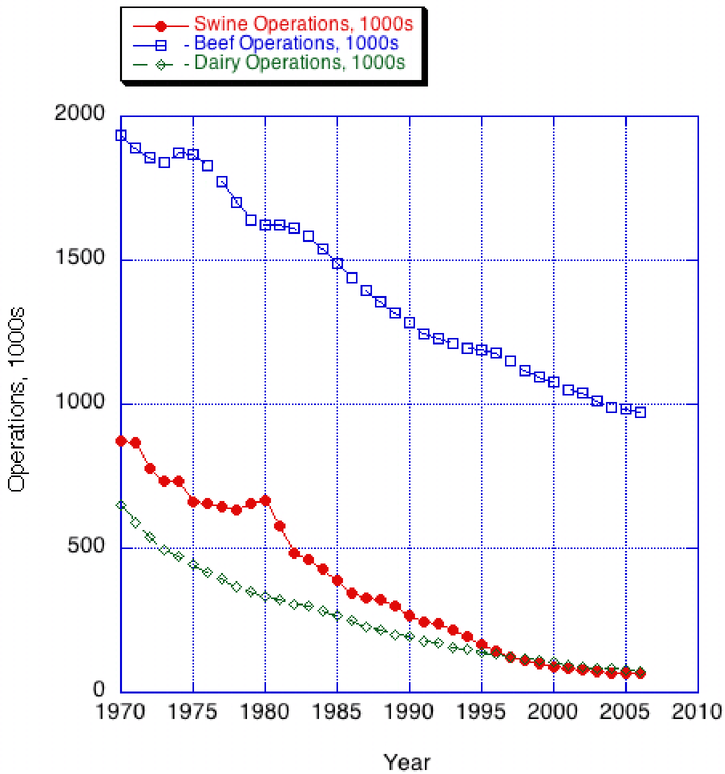


Figure 10: US Inventory of mature dairy cows and milk production per cow from 1970 to 2006. Source: USDA-NASS, Census Reports.



Another trend in animal agriculture has been the increased size and smaller number of animal operations, which results from the mechanization of agricultural practices and increased specialization. There were only 7% as many swine operations and 11% as many dairy operations in 2006 as there were in 1970 (Figure 11). There were half as many beef operations in 2006 as in 1970, but beef operations also expanded in size while smaller producers held jobs off the farm.

Figure 11: Number of animal operations in the US from 1970 to 2006. Source: USDA-NASS, Census Reports.



All of these trends show an increase in management and labor efficiency to produce a similar or greater amount of animal products. Also, because animal production is more concentrated on fewer farms with greater specialization, fewer crops are produced on those farms. As a result, it is increasingly common to have more manure nutrients produced on a livestock farm than can be used efficiently as fertilizer for crops on that farm. Therefore, unless the manure is applied over a larger crop area, the resulting over-application of manure on the livestock farm can reduce the subsequent efficiency of its utilization and result in greater nutrient losses.

2.2.4.2. Impact of livestock production trends on nitrogen use efficiency

The trends have both positive and negative environmental impacts. One of the significant positive impacts is that with smaller animal inventories producing greater quantities of animal products, there is an improved efficiency of nitrogen utilization per product produced. This effect is partly the result of effectively reducing maintenance requirements during production. The requirements for feeding animals can be divided into two components: maintenance and production. The maintenance component is that feed which is used to keep the animal alive and healthy so that production is possible. The production component includes feed that is converted to animal protein and waste due to the inefficiencies of these conversions. The maintenance component depends upon the number of animals, each animal’s mass, and the time the animal is on feed. Thus, the maintenance requirement is diluted by faster growth rates and greater body weight at slaughter. The increases in production rates over time have lead to greater efficiencies in N and P utilization for animal production and lower amounts of nutrients excreted per unit of animal protein produced.

Public concerns about the potential environmental and health effect of air emissions from CAFOs expand the impacts of food production beyond those associated with traditional agricultural practices (NRC, 2001, Aneja et al., 2009). Increased emissions of N compounds from animal agriculture into the atmosphere may lead to increased odor and interact in atmospheric reactions (e.g. gas-to-particle conversion) (Baek et al. 2004a; Baek and Aneja 2004b). These are then transported by wind and returned to the surface by wet and dry deposition processes, which may have adverse effects on human health and the environment (McMurry et. al, 2004; Aneja et. al, 2006, 2008a, b, c; Galloway et. al, 2008).

Adverse effects include eutrophication, soil acidification, loss of biodiversity, and reactions that increase the mass concentration of atmospheric aerosols ($PM_{2.5}$). Aerosol formation occurs when HNO_3 reacts with basic compounds, and NH_3 reacts with acidic compounds. Ecosystem acidification can occur when HNO_3 is deposited from the atmosphere. In addition, acidification can also occur when NH_x is deposited due to the production of HNO_3 from nitrification via soil microbes. Soil acidification occurs when HNO_3 or NH_4^+ deposits on soils with low buffering capacity, which can cause growth limitations to sensitive plant species. Deposition of NO_3^- or NH_4^+ also causes eutrophication (i.e. an over-abundance of nutrients), which can promote harmful algal growth leading to the decline of aquatic species. In fact, volatized NO_3^- can travel hundreds of miles from its source affecting local and regional biodiversity far from its origin (Aneja et al. 2008b; James, 2008).

The potential for reduced environmental impact from Nr in livestock systems depends on the proportion of the total intake attributable to maintenance costs. The commonly used tables for diet formulation published periodically by the National Research Council (NRC) for various animal commodities can be used to track diet formulation practices and assumptions regarding maintenance and production requirements. About one third of the energy intake recommended for growing broilers was assumed to be needed for maintenance (NRC, 1994) but protein requirements were not divided between maintenance and production. For example, a dairy cow producing 40 kg milk per annum would divert about 25% of its energy and 12% of its protein to maintenance (NRC, 1989).

In terms of nutritional efficiency of a herd or flock, maintenance of a productive phase (e.g. growth, lactation) also requires maintenance of a reproductive phase of the animals' life cycle. In other words, the actual nutritional maintenance cost of a herd or flock is greater than it is for productive individuals only. For example, milk production requires non-lactating cows and heifers in the herd which do not produce milk but which consume nutrients. These additional maintenance costs are lower for broiler flocks than for cattle.

Finding 5

There are no nationwide monitoring networks in the US to quantify agricultural emissions of greenhouse gases, NO, N_2O, reduced sulfur compounds, VOCs, and NH_3. In contrast there is a large network in place to assess the changes in the chemical climate of the US associated with fossil fuel energy production, ie the National Atmospheric Deposition Program/National Trends Network (NADP/NTN), which has been monitoring the wet deposition of sulfate (SO_4^{2-}), NO_3^-, and NH_4^+ since 1978.

Recommendation 5: *The status and trends of gases and particulate matter emitted from agricultural emissions, e.g., NO_3^- and NH_4^+ should be monitored and assessed utilizing a nationwide network of monitoring stations.*

2.2.4.3. Changes in feeding practices

From 1970 to 2006, several feeding practices were changed for diets fed to livestock. In 1989 and 1996, the NRC introduced the idea of dividing the form of protein fed ruminants into that which is degraded by rumen microorganisms and that which passes through the rumen to be digested directly in the stomach and small intestine. Feeding ruminants with attention to rumen degraded and rumen undegraded protein decreases the amount of protein fed by 10 to 15% for a given protein requirement. For poultry and swine, manufactured amino acids were added to diets, decreasing the need for protein by 30%. Today, two amino acids (lysine and methionine), coated in a way to prevent degradation in the rumen, are sometimes added to dairy cattle diets thereby decreasing protein intake by another 15% (NRC, 2001). Phytase added to swine and poultry diets in the past decade has decreased phosphorus feeding by 20 to 50% with some of the decrease attributed to simply better understanding phosphorus requirements

It is difficult to estimate the combined effects of changes in feeding practices, but for calculations on changes in manure N, we assume improvements in both production rates and ration formulation. In the case of beef cattle diet formulation, the changes in feeding practices were determined by comparing the NRC 1976 recommendations with the NRC 2006 recommendations. Surprisingly, NRC 1996 recommended greater total crude protein compared to NRC 1976 despite formulating for rumen degraded and un-degraded protein and considering amino acid content. Therefore, improved diet formulation did not decrease N intake for beef in this time range but the effect of reduced maintenance did improve efficiency of N utilization.

2.2.4.4. Reduced nitrogen excretion from increased efficiency

Nitrogen excretion as fraction of animal production decreased from 1970 to 2006 (Table 6). However, in cases where the total amount of animal production in the US increased substantially (e.g. broilers), total N excretion increased. The decrease in N excretion per unit of animal productivity was estimated by calculating the effects of changes in feeding practices and reduction of maintenance as described previously. The data on Table 6 indicate that there has been an increase in N utilization efficiency for livestock products.

Table 6: Livestock N excretion efficiency per kg production (g/kg) and per total US (Tg /yr)

Commodity*	1970		2006	
	g/kg product	Total US	g/kg product	Total US
Milk	17	0.89	11	0.92
Pork, live weight	57	0.56	42	0.54
Broilers, live weight	56	0.26	46	1.00
Beef, live weight	123	1.2	110	1.3

*Does not include manure produced for reproduction of stock (e.g. growing dairy heifers, breeder pigs).

For broilers, data are available to more accurately estimate the effect of changes in feeding and genetics on N excretion over time. However, these data do not represent the time period of interest in this report. Havenstein et al. (1994) compared a 1957 strain of broiler fed a 1957 diet to a 1991 strain fed a 1991 diet. Based on the reported N intake and production data, there was a 51% reduction in N excreted between these diets (Kohn, 2004).

Similarly, Kohn (2004) compared N excreted by US dairy cows in 1944 and 2001. In 1944, the historically largest herd of dairy cattle in the US (25 million cows) produced an average of 7 kg milk per cow per day (NASS-USDA, 2007). In 2001, nine million cows produced an average of 27 kg milk per cow per day. Assuming the cows in 1944 and 2001 were fed according to popular feeding recommendations of the time, the N intakes were 360 and 490 g/d per cow, and N excretion rates (N intake minus N in milk) were 326 and 364 g/d per cow. Multiplying by the number of cows in the US, shows that total milk production increased 40% from 52 billion kg to 73 billion kg, while N excretion decreased 60% from 3.0 Tg N to 1.2 Tg N, respectively.

For Table 7, manure N was calculated for all US animal agriculture using data on animal production from the 2002 Census of Agriculture (USDA 2002). For data on livestock production (cattle, calves, poultry, hogs, and pigs), manure was calculated by the methods of Moffit and Lander (1999), following the exact methods they had used to compute manure from the 1997 Census of Agriculture, but using the updated information from the 2002 Census of Agriculture. For data on production of manure from other animals (horses, goats, and sheep), the table uses coefficients for manure excretion as a function of average animal weights and animal inventory, taken from Battye et al. (1994).

Table 7: Manure production from animal husbandry in the continental US, Tg N per year 2002.

	Tg N/yr	%
Cattle & Calves	4.35	72
Poultry	0.94	16
Hogs & Pigs	0.53	9
Horses, Goats & Sheep	0.19	3
Continental US	6.02	100

2.2.4.5. Volatilization of animal waste

Ammonia volatilization is highly variable and is influenced by the amount of total ammonical nitrogen (TAN), temperature, wind speed, pH, chemical and microbiological activities, diffusive and convective transport in the manure, and gas phase resistance in the boundary layer above the source (Arogo et al., 2006). For example, greater TAN concentrations, wind speeds, temperatures, and pH levels increase NH_3 volatilization. Ammonia increases linearly with TAN concentration. Higher temperatures increase NH_3 volatilization rates due to decreased solubility in turn affecting NH_3/ NH_4^+ equilibrium which follows Henry's law for dilute systems:

$$NH_3(l) \leftrightarrow NH_3(g)$$

$$NH_3(g, manure) \leftrightarrow NH_3(g, air)$$

Ammonia-ammonium equilibrium [$NH_4^+(l) \leftrightarrow NH_3(l) + H^+$] is affected by temperature influencing the dissociation constant K_a [$K_a = (NH_3)(H_3O^+)/(NH_4^+)$] and pH (Arogo et al., 2006; James, 2008). At pH 9.2 a solution contains approximately equal amounts of solution NH_4^+ and solution NH_3. At pH 7.2 the solution contains approximately 99% solution NH_4^+ and 1% NH_3. Thus NH_3 emissions are typically higher in more basic soils. Chemical equilibria dictate that an aqueous solution will hold less NH_3 with increasing temperature so, temperature affects solution-atmosphere NH_3 exchange as well (Freney et al. 1983)

EPA estimates annual manure N excreted in livestock production in the US. for the "Inventory of US Greenhouse Gas Emissions and Sinks" (EPA, 2007). For the year 2002, these estimates (Appendix 3, Table A-174; EPA, 2007) indicate that a total of 6.8 Tg of N was excreted in livestock manure. Only a fraction of this N, ~1.24 Tg was recovered and applied directly as a nutrient source for crop production. Approximately 1.8 Tg N was transferred from the manure management systems, most likely by ammonia volatilization. Other loss vectors include leaching and runoff during treatment, and storage and transport before soil application. The remainder of the N was deposited in pastures and rangeland or in paddocks. This N is also susceptible to movement into the atmosphere and aquatic systems or incorporation into soil organic matter. By a combination of BMPs and engineered solutions it may be possible to reduce the emissions and discharge of odors, pathogens, and nitrogen compounds from agricultural operations (Aneja et al. 2008b,d).

Total manure reported in Table 7 in the contiguous US was estimated using USDA's method and yields an estimate of 6.0 Tg N/yr; while EPA's greenhouse gas inventory method in Table 8 yields a total for the US of 6.8 Tg N/yr in 2002. The 'greenhouse gas' method suggests 13% higher manure N production. This difference highlights uncertainty in the calculations. The values in Table 8 include Alaska and Hawaii whereas the values in Table 7 do not; though given the small relative amount of livestock production in those states that doesn't contribute substantially to the difference.

Table 8: Fate of Livestock Manure Nitrogen (Tg N) (EPA, 2007)

Activity	1990	1992	1994	1996	1998	2000	2002	2004
Managed manure N applied to major crops	1.1	1.2	1. 2	1.2	1.2	1.3	1.2	1.3
Manure N transferred from management systems	1.5	1.6	1.6	1.6	1.7	1.7	1.8	1.7
Pasture, range, & paddock manure N	3.0	4.0	4.1	4.2	3.9	3.8	3.8	3.7
Total	**6.6**	**6.7**	**6.9**	**7.0**	**6.9**	**6.8**	**6.8**	**6.7**

Finding 6

Farm-level improvements in manure management can substantially reduce Nr load and transfer. There are currently no incentives or regulations to decrease these transfer and loads despite the existence of management options to mitigate.

Recommendation 6: *Policy, regulatory, and incentive framework is needed to improve manure management to reduce Nr load and ammonia transfer, taking into account phosphorus load issues.*

2.2.5. Nr inputs to residential and recreational turf systems

Turf grasses cover 12.6-16.2 million ha across the continental US (Milesi et al. 2005). The area under turf grass is roughly the size of the New England states and occupies an area up to three times larger than that of *irrigated* corn (The Lawn Institute, 2007). The majority of this turf area (approximately 75%) is in residential lawns. About 80% of all US households have private lawns (Templeton et al. 1998) that average 0.08 ha in size (Vinlove and Torla, 1995). Another approximately15 % of total turf grass area is in low maintenance parks and approximately 10% is in athletic fields and golf courses, which often receive higher levels of N application due to hard use conditions.

Supplemental N fertilization is often necessary to maintain healthy and aesthetically pleasing turf color, high shoot density and the ability to resist and recover from stress and damage. Nitrogen also may be derived from atmospheric deposition or recycled decomposition of soil and grass clipping organic matter. Whether these inputs are sufficient to maintain lawns of adequate quality depends on many factors including age of the turf, uses, and expectations or goals of the homeowner or field manager. Also, turf grasses are used to stabilize soil, often with an erosion prevention matrix such as organic mats or with hydroseeding. Depending on circumstances, these turf uses may be temporary until natural vegetation succeeds the turf, or may be low maintenance turfs that are seldom fertilized such as highway medians and shoulders, grassy swales and buffers.

Turf grass is maintained under a variety of conditions. Approximately 50% of all turf grass is not fertilized, while the remainder is fertilized at varied intensities (Petrovic, personal communication—June 5 2007). We have arrayed the different turf managements into three groups according to the estimated amount of N-fertilizer applied annually (Table 9), residential lawns maintained by homeowners (0.73 kg/100 m^2), residential lawns cared for by professional lawn care companies (2.92 (range, 1.95-7.3) kg/100 m^2), and athletic fields and golf courses (3.89 (range, 2.64-6.64) kg/100 m^2). The estimate of total N-fertilizer used on turf grass in the US is 1.1 Tg N/year, or 9% of the total average annual N-fertilizer used between 1999 and 2005. Depending on land use patterns, certain areas of the country, particularly coastal areas where residential and urban properties prevail, turf fertilizer can be an important or even dominant source of nitrogen to surface waters.

Turf fertilizer N is susceptible to transfers to the atmosphere, and surface and ground water when it is not properly managed. Research on lawns has shown that leaching of NO_3^- can range between 0 and 50% of N applied (Petrovic, 1990). Nitrogen leaching losses can be greatly decreased by irrigating lightly and frequently, using multiple and light applications of fertilizers, fertilizing at the appropriate times, especially not too late in the growing season, and using soil tests to ensure proper balance of non-N soil condition and pH. In a soil column experiment with turf coverage, the percentage of N leached (as percentage of nitrogen applied) varied from 8 to 14% using light irrigation and from 2 to 37% with heavy irrigation.

Applying fertilizer in appropriate amounts, avoiding periods when grass is dormant, and not fertilizing too soon before irrigation or large rainfall events can all help ensure leaching and runoff will be minimal without affecting turfgrass color and growth (Mangiafico and Guillard, 2006).

Nitrogen runoff losses are poorly quantified but a range similar to leaching is probable (Petrovic, personnel communication). The chemical form of fertilizer N does not impact leaching/runoff unless it is applied in late autumn (Petrovic, 2004), although use of slow release or organic fertilizers can help reduce runoff and leaching. Shuman (2002) notes that runoff can be limited by applying minimum amounts of irrigation following fertilizer application and avoiding application before intense rain or when soil is wet. Transfers of Nr to the atmosphere can be significant when urea is applied. Measured denitrification losses are usually small, but depend upon timing of N application relative to soil water status, irrigation and temperature. Typically 25% of N applied is not accounted for in runoff, leaching, and uptake/removal, or soil sequestration (Petrovic, personnel communication), which suggests that volatilization and denitrification are important loss vectors. Nitrogen volatilization (Kenna, 2008, CAST Book) rates ranged from 0.9% under light irrigation to 2.3% under heavy irrigation.

While under-fertilization can lead to reduced grass stand and weed encroachment which results in more leaching and runoff N losses than from well managed lawns (Petrovic, 2004; Petrovic and Larsson-Kovach, 1996), Guillard (2006) recommends not fertilizing lawns of acceptable appearance. Further, prudent fertilization practices may include using one-third to one-half (or less) of the recommended application rate, i.e., application rates below 0.5 kg/100m^2, and monitoring response (Guillard, 2006). Less or no fertilizer may produce acceptable lawns, especially once the lawn has matured, provided clippings are returned and mowing length is left high.

As noted above, according to Petrovic (personal communication) half the lawns may not receive any fertilizer. Those lawns are presumably satisfactory to their owners. Further N reductions can be made if white clover is incorporated into turf and grasses such as fescues are selected for amenable parts of the country, which require little or no N supplements once mature. These practices can potentially reduce N fertilization (and subsequent leaching risk) on turf by one third or more, saving 0.4 or more Tg N/year. When properly managed, turf grass provides a variety of services that include decreasing runoff, sequestering carbon dioxide and providing a comfortable environment in which to live (Beard and Green, 1994).

Table 9: Estimate of Fertilizer N used on turf grass in the US in the year 2000, based on a total area of 12.6 million ha.

Type of Turf Fertilized	Area (Million ha)	N rate (kg/ha/yr)*	Total N Used (Tg/yr)
Nominal Fertilization	4.7	73	0.35
Professional Lawn Care	0.93	296 (195-488)	0.27
High Maintenance Areas (golf/sports)	1.26	390	0.49
Total	**6.89**	**--**	**1.11**

*1000 m^2/ha, used values of 0.73, 2.92 and 3.89 kg N/100 m^2

In recent years, about 11Tg of fertilizer N /year was used in the US. The above numbers convert to 1.1 Tg/year of N being used on turf (roughly 10% of US Total)

<u>Finding 7</u>

Synthetic N fertilizer application to urban gardens and lawns amounts to approximately 10% of the total annual synthetic N fertilizer used in the U.S. Even though this N is a large part of N fertilizer used little attention is paid to how efficiently it is used.

Recommendation 7a: *To ensure that urban fertilizer is used as efficiently as possible, the committee recommends that EPA work with other agencies such as USDA as well as state and local extension organizations to coordinate research and promote awareness of the issue.*

Recommendation 7b: *Through outreach and education, supported by research, improved turf management practices should be promoted, including improved fertilizer application and formulation technologies and maintenance techniques that minimize supplemental Nr needs and losses, use of alternative turf varieties that require less fertilization, alternative ground covers in place of turf, and use of naturalistic landscaping that focuses on native species.*

2.3. Nr transfer and transformations in and between environmental systems

This chapter discusses the transfers and flows of Nr within and between environmental systems (ES) which include atmosphere, terrestrial, and aquatic environments. The first section (2.3.1) contains information on Nr deposition from the atmosphere to terrestrial and aquatic systems, presents estimates of input and recycling of Nr within terrestrial systems, and discusses movement of Nr from the terrestrial to the aquatic system. The second section (2.3.2) presents an estimate of storage of Nr within the terrestrial system. The input and transfers of Nr within 16 northeast US watersheds is discussed in Section 2.3.3. Within the nitrogen cascade there are a number of places where the flow of Nr is constrained or regulated. In the final section (2.3.4) a list of critical information needs is presented.

2.3.1 Input and transfers of Nr in the US.

This section contains discussions on inputs and transfers between and within environmental systems. First Nr deposition from the atmosphere to earth's surface is considered. Second is input and transfer of Nr within terrestrial systems, and finally the transfer of Nr into aquatic systems is discussed.

2.3.1.1 Nitrogen deposition from the atmosphere to the earth's surface

Introduction. The magnitude and mechanisms of Nr deposition to the Earth's surface remain major unanswered environmental questions for the US, but atmospheric input contributes substantially to the Nr content of terrestrial and aquatic ecosystems. "Along the eastern US coast and eastern Gulf of Mexico, atmospheric deposition of N currently accounts for 10% to over 40% of new N loading to estuaries" (*Paerl et al.*, 2002). Other watershed contribution estimates range widely throughout the US, depending on size of the watershed related to the size of the estuary, and the magnitude of contributing sources of atmospheric N enrichment. Valigura et al. (2001) identified a median atmospheric nitrogen contribution of about 15% for 42 watershed located throughout the US, although the maximum estimate was 60%.

NO_x, NH_3 and their reaction products not deposited onto the continent are generally lofted into the free troposphere where they can have a wide range of influence and, in the case of NO_x, because of nonlinearities in the photochemistry, generate substantial amounts of tropospheric ozone (EPA, 2006). Total N deposition involves both gases and particles, and both dry and wet (in precipitation) processes. Rates of deposition for a given species (in units of mass of N per unit area per unit time) can be measured directly, inferred from mass balance of the atmospheric budget, or modeled numerically, but substantial uncertainties remain with each of these techniques when applied to deposition of any Nr species. A portion of the Nr deposited to the earth's surface is re-emitted as NH_3, NO, or N_2O (*Civerolo and Dickerson*, 1998; *Crutzen et al.*, 2008; *Galbally and Roy*, 1978; *IPCC*, 2007; *Kim et al.*, 1994). Although naturally-produced Nr is involved, anthropogenic Nr dominates over most of the US. In this section we review the state of the science concerning the total annual Nr deposition and trends in that deposition to the contiguous 48 states.

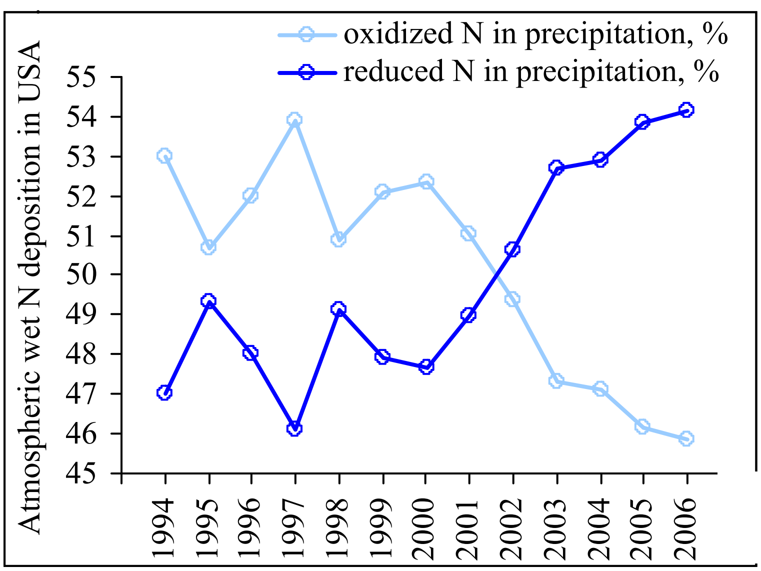
Deposition involves both oxidized and reduced N species. Of the oxidized forms of atmospheric N, all the members of the NO_y family (NO, NO_2, NO_3, N_2O_5, HONO, HNO_3, NO_3^-, PAN and other organo-nitrates, $RONO_2$) can be transferred from the troposphere to the surface, and some undergo bidirectional flux, e.g., NO. Note that volatile amines are also detected as NO_y compounds (*Kashihira et al.*, 1982; *Wyers et al.*, 1993). Although a potent greenhouse gas, N_2O is only emitted, not deposited and therefore will not be considered here. Of the reduced forms of atmospheric nitrogen, NH_3 and NH_4^+ play a major role. There is also evidence of deposition of organic N such as amino acids and isoprene nitrates, and recent observations suggest that these can account for as much as 10% (possibly 30%) of the US NO_x budget, especially in summer (*Duce et al.*, 2008; *Horowitz et al.*, 2007; *Keene et al.*, 2002; *Sommariva*, 2008). While this is a worthy research topic, measurements are still limited and deposition of organic N compounds will not be reviewed here. The wide array of relevant atmospheric compounds makes direct measurement, and accurate load quantification challenging.

Review of Nr wet deposition. Substantial progress has been made in monitoring wet deposition, as is summarized by the National Atmospheric Deposition Program/National Trends Network (NADP), established in 1979, which monitors precipitation composition at over 250 sites in the US and its territories (http://nadp.sws.uiuc.edu). Precipitation at each station is collected weekly according to well established and uniform procedures from which it is sent to the Central Analytical Laboratory for analysis of acidity, NO_3^-, NH_4^+, chloride, as well as the base cations calcium, magnesium, potassium and sodium. For greater temporal resolution, the Atmospheric Integrated Research Monitoring Network AIRMON, comprised of seven sites, was formed in 1992 as part of the NADP program to study wet deposition composition and trends using samples collected daily. The same species are measured as in NADP. By interpolating among sites, NADP is able to estimate the wet deposition of NH_4^+ (reduced N), and NO_3^- (oxidized N) for the 48 contiguous states (Table 10 and Figure 12).

Table 10: Annual wet deposition of reduced (NH_4^+), oxidized (NO_3^-), and total N to the 48 contiguous states, from the NADP/National Trends Network (NTN) http://nadp.sws.uiuc.edu

NADP/NTN deposition estimates	reduced N in precipitation, kg/ha/yr	oxidized N in precipitation, kg/ha/yr	total wet N deposition, kg/ha/yr
1994	1.49	1.68	3.17
1995	1.63	1.67	3.30
1996	1.66	1.80	3.45
1997	1.49	1.74	3.24
1998	1.72	1.78	3.49
1999	1.46	1.58	3.04
2000	1.48	1.62	3.10
2001	1.50	1.57	3.07
2002	1.59	1.55	3.14
2003	1.72	1.55	3.27
2004	1.70	1.52	3.22
2005	1.65	1.41	3.06
2006	1.65	1.40	3.05

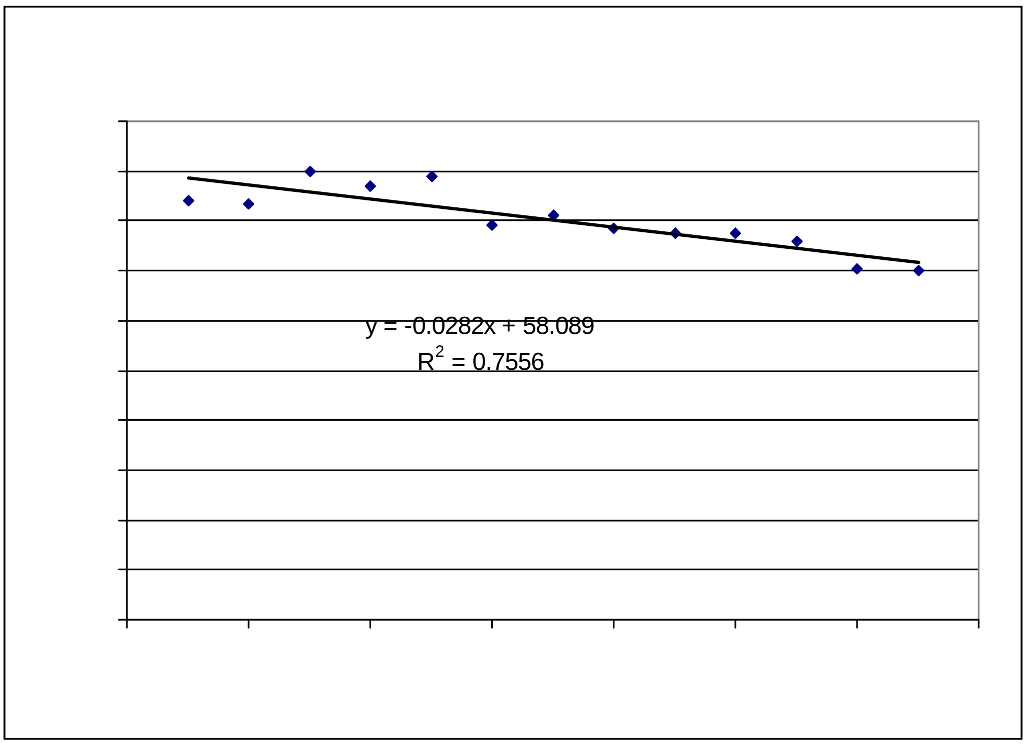
Figure 12: Percent change in relative contribution of oxidized (NO_3^-) and reduced (NH_4^+) nitrogen wet deposition from 1994 to 2006. As emissions of NO_x have decreased, the relative importance of NH_x has increased.



Although individual regions vary, the NADP data for the entire 48 states indicate an apparent decrease in NO_3^- wet deposition, but not in NH_4^+ deposition (Table 10 and Figure 13). This suggests that as NO_x controls have become more effective, the role of reduced N has grown in relative importance. The nitrate data appear to show a strong trend (data from Table 10 plotted in Figure 13) and quantifying the response of deposition to a change in emissions would be useful to both the scientific and policy communities. A notable reduction in power plant NO_x emissions occurred as the result of the NO_x State Implementation Plan (SIP) call (*Gilliland et al.*, 2008; *McClenny et al.*, 2002). EPA should pursue a rigorous analysis of the emissions and deposition data, including identifying monitors and methods that are consistent from the beginning to the end of the record, as indicated in Recommendation D.

Figure 13: Trend in reported wet deposition of NO_3^- for the 48 contiguous states; data were taken from Table 10.

Note the sampling methods and locations have not been tested for temporal or spatial bias.

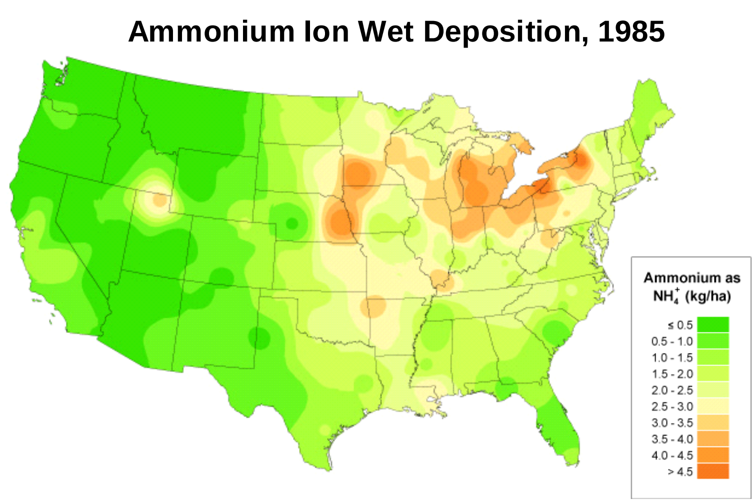


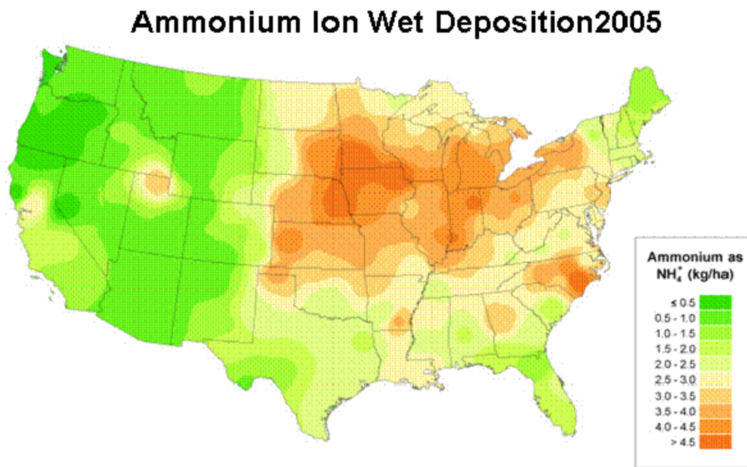
How is Nr deposition related to emissions? The relationship between emissions of Nr and observed deposition is critical for understanding the efficacy of abatement strategies as well as for partitioning local and large-scale effects of emissions. Only a few studies covering several individual sites have sufficient monitoring consistency and duration to determine rigorously long-term trends in NO_3^- and NH_4^+ and their relationship to emissions, and here we consider several examples (*Butler et al.*, 2005; *Kelly et al.*, 2002; *Likens et al.*, 2005). These sites tend to be in the eastern US where monitoring is more concentrated and has a longer history and where upwind sources and downwind receptors are relatively well known. Examination of these studies reveals that concentrations of gaseous and particulate N species in the atmosphere, as well as the Nr content of precipitation over the eastern US shows significant decreases.

Correlation with regional emissions is stronger than with local emissions, in keeping with the secondary nature of the major compounds – NO_3^- and NH_4^+. Decreases in NH_4^+ concentration and wet deposition are attributed to decreases in SO_4^{2-} concentrations meaning that more of the reduced Nr remains in the gas phase. For the period 1965 to 2000, NO_3^- levels in bulk deposition correlate well with reported NO_x emissions. For shorter and earlier time periods the correlation is weaker, and the authors attribute this to changes in the EPA's methods of measuring and reporting emissions; they find evidence of continued errors in emissions from vehicles. Decreases in deposition will probably not be linearly proportional to decreases in emissions; for example a 50% reduction in NO_x emissions is likely to produce a reduction of about 35% in concentration and deposition of nitrate.

The relationship between reduced N emissions and deposition is more complex. When looked at over the full extent of the record, from 1985 to 2005, the increase in NH_4^+ wet deposition becomes apparent (Figure 14), especially in selected areas of the country http://nadp.sws.uiuc.edu/amaps2/. The southeastern US, particularly North Carolina, has seen a long-term increase [*Aneja et al.*, 2000; *Aneja et al.*, 2003; *Stephen and Aneja*, 2008]. The increase in deposition coincides with the increase in livestock production, but a swine population moratorium appears to have helped abate emissions (*Stephen and Aneja*, 2008). Concentrations of aerosol NH_4^+ have decreased in many parts of the country, and this may appear to contradict the trend in wet deposition, but a decrease in condensed phase NH_4^+ will be accompanied by an increase in vapor phase NH_3 if SO_4^{2-} and NO_3^- concentrations decrease; see http://vista.cira.colostate.edu/improve/. This potentially misleading information highlights the need for measurements of speciated NH_x (*Sutton et al.*, 2003).

Figure 14: Total annual NH_4^+ deposition for the years 1985 and 2005 showing increases in the Midwest and Southeast, especially North Carolina commensurate with increases in livestock production. http://nadp.sws.uiuc.edu/amaps2/





Review of dry deposition observations for the eastern US. Monitoring dry deposition presents a greater challenge. The Clean Air Standards and Trends Network (CASTNET) and Atmospheric and Integrated Research Monitoring Network (AIRMON) were established to monitor chemical and meteorological variables to infer dry deposition in order to study the processes leading from emissions to atmospheric concentrations and through deposition to ecosystem effects. AIRMON dry deposition monitoring was discontinued in 2003. See http://www.epa.gov/castnet/, http://www.arl.noaa.gov/reserach/programs/airmon.html, and http://nadp.sws.uiuc.edu.

Recent reviews (*Sickles and Shadwick*, 2007a; *Sickles and Shadwick*, 2007b) analyzes the seasonal and regional behavior of concentration and deposition of a variety of primary and secondary pollutants including reactive N and investigated trends from 1990 to 2004 for the US east of the Mississippi River. The investigators evaluated observations from more than 50 sites in the eastern States and concluded that for 2000-2004, the mean annual total measured N deposition for this area was 7.75 kg N per hectare per year (expressed as kg N/ha/yr); see Table 11. This value includes vapor phase HNO_3, particulate NO_3^-, and NH_4^+; it does not include deposition of other oxidized species such as NO_x and PAN, nor gas-phase reduced N species most notably NH_3. The measured deposition rates peak in spring and summer, but unaccounted for ammonia deposition is probably a substantial fraction of the total, and the true annual cycle remains uncertain.

Table 11: Deposition of N to the eastern US in units of kg N/ha/yr

	Annual deposition **kg N/ha/yr**
Dry NH_4^+	0.41
Wet NH_4^+	2.54
Dry $HNO_3 + NO_3^-$	1.88

Wet NO_3^-	2.92
Total measured N Dep.	7.75
Est. dry other NO_y	0.94
Est. dry NH_3	1.90
Est. total NO_y	5.74
Est. total NH_3 + NH_4^+	4.85
Est. Grand Total	**10.59**

Data are from the US CASTNET program for the period of 2000-2004. Monitored species for 34 sites east of the Mississippi include vapor-phase HNO_3, particulate NO_3^-, and NH_4^+; unmonitored are other oxidized species such as NO_x and PAN and gas-phase reduced N species most notably NH_3 (Sickles and Shadwick, 2007a). For an explanation of how deposition of unmeasured species was estimated see text.

Estimated total N deposition to the eastern US. CASTNET monitors HNO_3 and NO_3^-, but not other members of the NO_y family – notably NO_x. Dennis (EPA, 2007) estimated that the unmeasured NO_y species account for about 50% of the dry deposition of nitrates. Half of 1.88 (see Table 11) is 0.94 kg N /ha/yr. Ammonia is also unmeasured by CASTNET, and model estimates [*Mathur and Dennis*, 2003] of NH_3 indicate that dry deposition should account for 75% of wet NH_4^+ deposition; 75% of 2.54 is 1.9 kg N /ha/yr. Adding these two values to the total from Table 11 yields a reasonable estimate, within about ±50% absolute accuracy, of total deposition of about 10.6 kg N /ha/yr for the eastern US.

Characteristics of N deposition to the eastern US. Highest N deposition occurs in the spring and summer, when chemical thermodynamics and photochemistry are conducive to removal from the atmosphere. As temperatures warm, HNO_3 formation accelerates and fertilizer application to agricultural fields steps up. Dry deposition for gases is faster than for particles; for example the mean CASTNET reported HNO_3 deposition velocity is 1.24 centimeters per second (cm/s) while that for particulate NO_3^- is 0.10 cm/s. Conversion of condensed ammonium nitrate to gaseous NH_4^+ and HNO_3 is favored at high temperatures. Oxidation of NO_x to HNO_3 is faster in the spring and summer due to greater ozone and hydroxyl radical (OH) concentrations than in the winter. Warm temperatures favor release of NH_3 from soils, and summer months are the season of fastest conversion of SO_2 into H_2SO_4; NH_3 combines rapidly with SO_4^{2-} to form ammonium sulfate or bisulfate that are then washed out of the atmosphere.

Wet deposition of NH_4^+ and NO_3^- dominates deposition, averaging for the sum of NH_4^+ and NO_3^- 5.46 kg N /ha/yr, or 70 % of the total, but dry deposition cannot be neglected; it averaged 2.29 kg N /ha/yr or 30 % of the measured total. Because foliar resistance to HNO_3 is weak, dry deposition of nitrate accounts for a large fraction of the total Nr deposition. When we add estimated NO_x and NH_3 dry deposition (Table 11), the sum of 0.41, 1.88, 0.94, and 1.90 is 5.13 kg N /ha/yr and rivals that delivered in precipitation.

The regional gradient is relatively modest, with the least annual average N deposition occurring in the Southeast (6.77 kg N /ha/yr) and the greatest in the Midwest (8.74 kg N /ha/yr). These gradients are driven primarily by differences in abundance – the annual mean concentration of total measured atmospheric N was 1.68 μg/m^3 in the Southeast and 2.40 μg/m^3 in the Midwest. Because only the secondary products of primary pollutants were measured, such relatively uniform concentrations are typical. Also contributing to the relative spatial uniformity of deposition is the greater rate of precipitation in the southeast. The policy-relevant implication of the large spatial scale nature of N deposition is that large-scale control measures are called for.

Trends in measured and inferred deposition over the 15-year monitoring period (Figure 13) reflect trends in emissions. In 2003 and 2004 substantial reductions in emissions from electric generating units (power plants) were implemented under the NO_x State Implementation Plan (SIP) call. Many of these power plants are located along the Ohio River generally upwind of the measurement area. The observed trend between 1990 and 1999 was weak, but significant reductions ($p = 0.05$) were found between the 1990-1994 and 2000-2004 periods (*Sickles and Shadwick*, 2007a). The concentration of nitric acid fell from 1.99 to1.74 μg N/m^3 or by 13%, and total nitrate deposition fell by 0.56 kg N/ha/yr or 11%. NO_x emissions controls are implemented primarily in the ozone season (May to September) and greatest reductions in N deposition were observed in the summer. For NH_4^+, the average concentration fell from 1.83 to 1.61 μg N/m^3 probably as a result of lower sulfur emissions. No change was observed in wet NH_4^+ deposition.

Sickles and Shadwick (2007b) attributed the reduction in NO_3^- deposition to reductions in NO_x emissions. They also reported that the relationship between emissions and deposition was less than 1:1. In other words, emissions were reduced by about 22%, but deposition fell by only about 11%. This nonlinearity may be a function of the time intervals chosen. The second five-year period averages from 2000-2004, but reductions went into effect over the 24-month period 2003-2004. Deposition depends on both chemistry and climate, and weather shows substantial interannual variability.

Uncertainty in measured deposition. Analysis of uncertainties in the deposition of Nr is challenging. The coefficient of variation for total, regional N deposition for 2000-2004 is 23%, representing a minimal value of uncertainty. Concentrations of some of the NO_y species are monitored, as is the wet deposition of major oxidized and reduced N species, but concentrations of ammonia and other Nr species are not monitored. The network for monitoring dry deposition is sparse and has not been evaluated for spatial bias. The monitors are located in flat areas with uniform surfaces – advective deposition into for example the edges of forests are estimated to contribute substantially to the uncertainty (*Hicks*, 2006). Other sources of error include the model used to convert weekly average concentrations and micrometeorological measurements into depositions. Precision can be determined from collocated sites and is estimated at 5% for nitrate and 15% for ammonium in precipitation (*Nilles et al.*, 1994). The uncertainty in estimated dry deposition arises primarily from uncertainty in deposition velocities (*Brook et al.*, 1997; *Hicks et al.*, 1991) and can be as high as 40% for HNO_3. Total uncertainty for deposition of Nr based on measurements is at least 25% and may be as high as 50%.

Deposition estimates from numerical models. The EPA Community Multiscale Air Quality model (CMAQ) was run for North America at 36 km resolution (R. Dennis et al., personal communication January 2008). Calculated nitrogen deposition for the 48 contiguous states (Table 12) was broadly consistent with direct measurements (Table 11). This run of CMAQ did not account for NO_x emissions from marine vessels, and these accounted for about 4% of the total NO_x emissions in 2000. CMAQ NO_x emissions were 5.84 Tg N for the year 2002; of that 2.74 Tg N were deposited. This suggests that ~50% was exported – a number somewhat higher than has been reported in the literature; this discrepancy is discussed below.

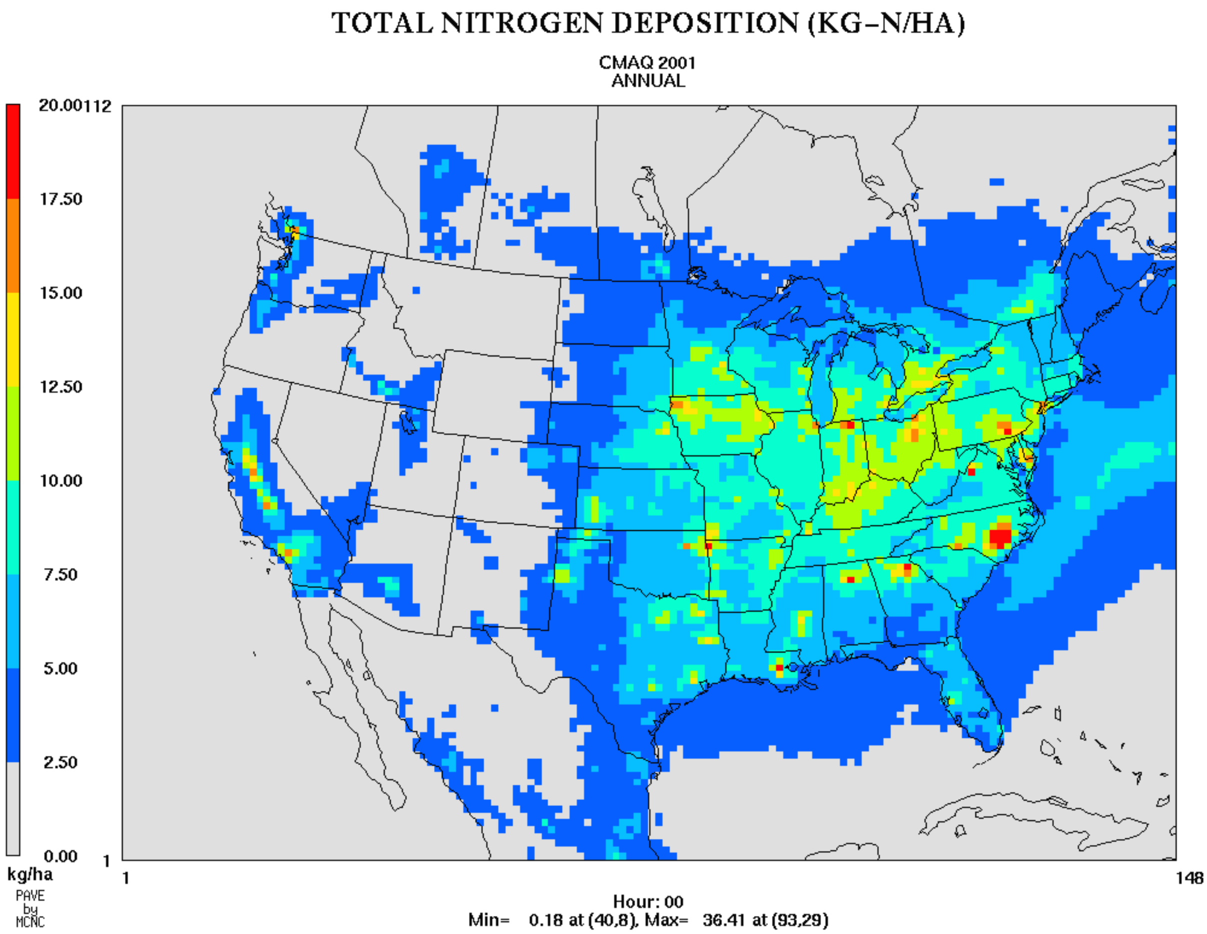
Table 12: Results from CMAQ for total deposition in 2002 to the 48 contiguous states of oxidized and reduced N.

	kg N/ha/yr	Tg N/yr
Oxidized N	3.51	2.74
Reduced N	2.66	2.07
Total N Depos.	6.17	4.81

Ammonia emissions and ambient concentrations can be measured, but are not routinely monitored. For Nr, the CMAQ numerical simulation employed inverse modeling techniques – that is NH_3 emissions were derived from observed NH_4^+ wet deposition (*Gilliland et al.*, 2006; *Gilliland et al.*, 2003; *Mathur and Dennis*, 2003). Model determinations therefore do not provide an independent source of information on NH_4^+ deposition.

The three-year CMAQ run gives an indication of the spatial pattern of deposition (Figures 15). For NH_x, wet and dry are equally important, but for NO_y, dry deposition accounts for about 2/3 of the total deposition while wet deposition accounts for about 1/3. For NH_x, wet and dry are equally important, but for NO_y, dry deposition is greater than wet. While this is not true for the eastern US it is true for the US as a whole; in arid southern California, for example, dry deposition of Nr dominates. Based on CMAQ, total NO_y deposition is 2.79 times the wet deposition and total NH_x deposition is 1.98 times the wet deposition. Using the data from Table **12** for the average wet deposition for the period 2000- 2004, total deposition of oxidized N is 4.36 kg N /ha/yr (2.79 * 1.56 = 4.36). The total deposition for reduced N is 3.17 kg N /ha /yr (1.98 * 1.60). The grand total (wet and dry oxidized and reduced) is then about 7.5 kg N /ha /yr[1].

<u>Figure 15: CMAQ annual average (wet plus dry and oxidized plus reduced) nitrogen deposition (in kg-N /ha/yr) across the US based on 3 years of differing meteorology - one dry, one wet, and one average precipitation year - across the Eastern US (Source: US EPA, 2007).</u>



The model has highly simplified organic N deposition. Note these values reflect emissions before the NO_x SIP-call which resulted in substantial reductions in NOx emissions from point sources over the eastern US.

For comparison purposes, a collection of Chemical Transport Models (CTM's) (Dentener e*t al.*, 2006) yielded total (wet plus dry) deposition to the whole US of about 3.9 Tg N /yr oxidized Nr and 3.0 Tg N /yr ammoniacal N for current emissions. The fate of NO_x is assumed to be primarily HNO_3 or aerosol NO_3^- ; organic N species are generally not modeled in detail. Because this analysis includes Alaska, a better estimate for NO_x for the 48 contiguous states is 4.6 Tg N /yr. The variance among models was about 30% (one б) for deposition fluxes in regions dominated by anthropogenic emissions. Globally, the calculations from the ensemble of 23 CTM's estimated 36-51% of all NO_y and NH_x emissions are deposited over the ocean. This load could be important to estuarine N loading estimates as offshore N is carried inshore by currents or through advective processes.

Deposition estimates from mass balance. If the total emissions of Nr compounds are known, and if the deposition is rapid, then a reasonable estimate of rate of deposition can be obtained by mass balance – deposition equals emissions minus export. Although substantial uncertainty (about a factor of two) exists for the emissions of NH_3, NO_x release is reasonably well known. In general, advection in the boundary layer and lofting through convection followed by export at higher altitudes are the two main mechanisms that prevent removal of NO_y and NH_x by deposition to the surface of North America (Li et al., 2004; Luke et al., 1992).

As early as 1985, experiments were devised to measure the transport of N pollutants offshore of North America (Galloway et al., 1988; Galloway and Whelpdale, 1987; Galloway et al., 1984; Luke and Dickerson, 1987). Galloway et al. (1984) estimated, based on the limited data available at the time, an annual average eastward NO_y flux of 3.2 Tg N /yr between the surface and 5000 m altitude. For the early 1980's this represents about 40% of the NO_x emitted, but more recent estimates have yielded a lower value. Dickerson et al. (1995) estimated that about 0.4 Tg N was advected at altitudes below 3000 m off the North American East Coast in winter; this represents about 6.5% of the total N emissions at the time. More recent estimates, again using data from lower to mid tropospheric altitudes over the eastern US (Li et al., 2004; Parrish et al., 2004b), estimated that 10 - 15% of the emitted NO_x was exported in the spring and fall. A summer season determination (Hudman et al., 2007) indicated about 15% NO_x export in the 2.5–6.5 km altitude range.

None of these studies, based on observations or combinations of observations and models, evaluated N flux resulting from deep convection, which can account for substantial transport of boundary layer (BL) air in the summer (Chatfield and Crutzen, 1984; Luke et al., 1992; Ryan et al., 1992). Uncertainty in the convective mass flux and in NO produced by lightning make direct determination of NO_y vented from the BL difficult. The convective mass flux is at present a poorly constrained quantity, uncertain to about a factor of two (Doherty et al., 2005; Lawrence et al., 2003).

In an early, model-based mass balance study (Kasibhatla et al., 1993), wet and dry deposition in source regions were estimated to account for 30% and 40-45% of the emissions, respectively. The authors reported that the remainder (25-30%) was exported off the continent, and more recent modeling studies tend to agree with a determination of 65-75% deposition (Doney et al., 2007; Galloway et al., 2004; Holland et al., 1997; Holland et al., 2005; Horowitz et al., 1998; Liang et al., 1998). In general, these CTM's derived small export values – on the order of 30% of the total NO_x emitted into the lower atmosphere. For example, Park et al. (2004) used a stretched-grid global model with highest resolution over the US to estimate NO_x and NO_y export for June 1985. They reported boundary layer NO_y advection of 0.56 Tg N /yr and total exports of 1.94 Tg N /y; deposition accounted for approximately 76% of the emitted NO_x. There is substantial model-to model variability, with one model (Penner et al.,

1991) putting more nitrate deposition into the Gulf of Mexico. The models appear to match well the measured boundary layer export and the ratio of NO_x/NO_y, e.g., (Luke et al., 1992; Parrish et al., 2004a) and generally agree with direct measurements. In summary, reviewed publications using the mass balance approach have substantial uncertainty but indicate with some consistency that 25-35% of the NO_y emitted over the US is exported.

Comparison of models and measurements of oxidized N deposition. Both ambient measurements and numerical models of NO_y have reached a level development to allow reasonable estimates of deposition. For reduced nitrogen, neither ambient concentrations nor emissions are known well enough to constrain models. Here we will review published research on NO_y export and deposition. Recent model estimates of the US N budget are reasonably uniform in finding that about 25-35% of total NO_x emissions are exported. From those studies we can estimate the vertical flux into the surface of the 48 contiguous states. For the 2000-2002 period, total NO_x emissions were about 4.5 Tg N /yr. The upper limit to deposition, if all of this is deposited onto the continent, would have been 5.7 kg N /ha /yr for the 7.8 x 10^6 km^2 (7.8 x 10^8 ha) surface area of the 48 contiguous States. The studies reviewed above suggest that 70% of the N released is deposited, and this works out to ~4.0 kg N /ha /yr. This is comparable to the oxidized N deposition of 5.7 kg N /ha /yr estimated from CASTNET observations for the eastern States (Table 10).

Results from CMAQ runs, described above, indicate that of the NO_x emitted over the continental US, 50% is deposited and 50% is exported. This is within the combined error bars of other studies, but well under the best estimate of 70% deposition. One possible source of this discrepancy is organo-nitrogen compounds. The mechanism for formation and deposition of organic nitrates is uncertain, and the chemical mechanism used in CMAQ was highly simplified – only about 2-3% of the total Nr deposition can be attributed to organo-nitrogen compounds (R. Dennis personal communication, 2008). Several recent studies (Duce et al., 2008; Neff et al., 2002; Perring et al., 2009) suggest that organic Nr constitutes a substantial, but still highly uncertain fraction of the Nr load. Many of these compounds (such as peroxy-methacrylic nitric anhydride, $CH_2C(CH_3)C(O)OONO_2$) are formed by reactions between VOC's and NO_x. Such compounds are detected as NO_y and are thus included in measurements of Nr export. Arbitrarily up-scaling of CMAQ deposition would then violate mass balance. EPA should investigate the source of this discrepancy and support research to reduce the uncertainty in Nr deposition and export.

The total wet deposition of nitrate to the 48 contiguous states averaged 1.6 kg N/ha/yr for the period 2000-2002 (Table 11). If we assume an equal amount is lost from the atmosphere through dry deposition, then the total deposition of oxidized N to the surface is 3.2 kg N/ha/yr, close to the implied model results of ~4.0 kg N/ha/yr. The estimate of equal fractions wet and dry deposition carries substantial uncertainty – NADP maps show, for example, little wet deposition of nitrate in southern California, but this region is known to experience high concentrations of NO_y. Neither approach to determining deposition is certain to be better than about ± 50%, so additional work is called for.

Emissions from Canada and Mexico can have a substantive impact on atmospheric Nr over the US near major sources such as downwind of industrial Ontario and major cities of Mexico, such as Tijuana and San Diego, CA (Wang et al., 2009). While Nr is imported into the US from these boarder countries, there is also export. The emissions from Canada and Mexico are each 10-15% of those of the US and the bulk of the Mexican population is distant from the US. We expect the overall impact of neighboring countries to add about 10% uncertainty to the estimated Nr budget for the 48 contiguous states.

Although fossil fuel combustion represents only a small fraction of the total NHx emissions in heavily trafficked corridors vehicles may represent a major local source of atmospheric NH_3 and Nr deposition. On an urban or roadside scale, NH_3 emissions from vehicles can be important (Cape et al., 2004; Kirchner et al., 2005; Li et al., 2006). These effects are limited to these small scales, however, because the total amount of ammonia emitted is small relative to agricultural emissions. At an emission rate of ~27 mg NH_3 (N) km^{-1}) the entire fleet of American vehicles emits less than 0.1 Tg NH_3 (N) yr^{-1}, a small fraction of that emitted by agricultural practices. Cape et al (2004) saw a rapid decrease in vehicular ammonia deposition in the first 10 m downwind of the roadway.

Major sources of uncertainty in modeled and observed values include missing deposition terms and poorly constrained convective mass flux. As indicated above, convective mass flux (rapid vertical transport) is uncertain because most convective clouds are smaller than a grid box in a global model. There is evidence for nonlinearities in NO_2 deposition velocities with greater transfer from the atmosphere to the surface at higher concentrations (Horii et al., 2004; 2006).

Conclusions on atmospheric deposition of Nr. Downward transport from the atmosphere is a major source of Nr to the Earth's surface, but there are uncertainties in the characteristics and absolute magnitude of the flux. Pollutants not deposited are exported from the continent and alter the composition and radiative balance of the atmosphere on a large scale. A review of the literature revealed the following major points concerning the present state of the science:

1. Measurements from the National Atmospheric Deposition Program (NADP) indicate that wet deposition of ammonium plus nitrate for the period 2000 – 2006 averaged 3.1 kg N/ha/yr over the 48 contiguous States.
2. The reduced (NH_4^+) and oxidized (NO_3^-) forms of reactive N contributed about equally to the flux, but input to the eastern US was greater (and less uncertain) than to the western US.
3. For the US east of the Mississippi River, dry deposition data have also been analyzed – the Clean Air Standards and Trends Network (CASTNET) monitors vapor phase HNO_3, as well as particulate NO_3^- and NH_4^+. These measurements indicate 7.75 kg N/ha/yr total deposition (5.46 wet 2.29 dry) over the East. Conspicuous by its absence from this number is dry deposition of ammonia.
4. Decreases in NOx emissions appear to lead to decreases in deposition. NADP data show a national decreasing trend in the wet nitrate deposition and some individual sites show statistically significant decreases in deposition and correlations with emissions.
5. A thorough review of all published studies of the US NO_y budget indicates that about 70 % of the NO_x emitted by the US is deposited onto the continent with the remainder exported, although substantial uncertainty remains. Major sources of error include dry deposition of unmonitored members of the NO_y family, uncertainties in the chemistry of organic N, and poorly constrained estimates of convective venting of the planetary boundary layer.
6. Based on observations and model estimates of the relative deposition of unmeasured quantities, total estimated deposition of all forms of Nr for the period 2000-2004 is ~11 kg N /ha /yr for the eastern US, and for the 48 States ~7.5 kg N /ha /yr with a range of 5.5 to 9.5 kg N /ha /yr.

Finding 8

Scientific uncertainty about the origins, transport, chemistry, sinks, and export of Nr remains high, but evidence is strong that atmospheric deposition of Nr to the Earth's surface as well as emissions from the surface to the atmosphere contribute substantially to environmental and health problems. Nitrogen dioxide, NO_2, is often a small component of NOy, the total of oxidized nitrogen in the atmosphere. The current NAAQS for NO_2, as an indicator of the criteria pollutant "oxides of nitrogen," is inadequate to protect health and welfare. Serious consideration should be given to replacing or supplementing the NO_2 measurements and standard with NOy. Atmospheric emissions and concentrations of Nr from agricultural practices (primarily in the form of NH_3) have not been well monitored, but NH_4^+ ion concentration and wet deposition (as determined by NADP and NTN) appear to be increasing. This suggests that NH_4^+ emissions are increasing. Both wet and dry deposition contribute substantially to NH_x removal, but only wet deposition is known with much scientific certainty. Thus consideration should be given to adding these chemically reduced and organic forms of Nr to the list of Criteria Pollutants.

Recommendation 8a. *Increase the scope and spatial coverage of the Nr concentration and flux monitoring networks (such as the National Atmospheric Deposition Program and the Clean Air Status and Trends Network) and appoint an oversight review panel for these two networks.*

Recommendation 8b. *Monitor NH_3, NH_x, NO_y, NO_2, NO, and PAN concentrations, measure or infer deposition, and support the development of new measurement and monitoring methods.*

Recommendation 8c. *Measure deposition directly both at the CASTNET sites and in nearby locations with non-uniform surfaces such as forest edges.*

Recommendation 8d. *EPA should continue and support research into convective venting of the Planetary Boundary Layer and long range transport.*

Recommendation 8e. *Develop and support analytical techniques and observations of atmospheric organic N compounds in vapor, particulate, and aqueous phases.*

Recommendation 8f. *Increase the quality and spatial coverage of measurements of the NH_3 flux to the atmosphere from major sources especially agricultural practices.*

Recommendation 8g. *Improve numerical models of NOy and NHx especially with regard to chemical transformations, surface deposition, and off shore export; develop linked ocean-land-atmosphere models of Nr.*

2.3.1.2 Input and recycling of Nr within terrestrial systems in the US

This section builds upon Section 2.2 by integrating the information in that section on Nr introduction into the US and its transfer to environmental systems by energy and food production into the overall picture of Nr cycling within terrestrial systems.

Annual input of newly created Nr onto terrestrial ecosystems comes from atmospheric deposition, synthetic fertilizer and BNF in managed and unmanaged ecosystems (Table 1.). Although Nr from atmospheric deposition is formed inadvertently during fossil fuel combustion and from volatilization of NH_3 from agricultural activities it serves to provide nutrients, along with biological N fixation and synthetic fertilizer, for food, feed and fiber production in the agricultural sector. Forests and grasslands use Nr for growth and home gardens, parks and recreational areas utilize Nr within the urban landscape. Approximately 32 Tg of new Nr reached the land of the 48 contiguous states in 2002 (Table 1). An additional ~0.2 Tg of N was imported mainly as food and drink products (FAO, 2008). An additional ~12 Tg of Nr was recycled back to terrestrial and aquatic systems in livestock (~6 Tg N) excreta, human (~2 Tg N) excreta, and crop residue from the previous year's production (~4 Tg N; USEPA, 2007). Of this N ~ 1.3 Tg (~1.2 from livestock manure and <0.1 from sewage sludge) was used as fertilizer for crop production (USEPA, 2007).

Most of the new Nr (~17 Tg total with 9 Tg from synthetic fertilizer and ~8 Tg from biological N fixation; Table 13) was used to produce food for human consumption and forage and feed for livestock and poultry. In addition to new Nr and Nr that was recycled from livestock and human excreta, crop production releases Nr that is stored in soil organic matter (see section 3.3.2). The N in cereal crops is typically derived from added fertilizer (synthetic or manures) and from mineralization of soil organic matter (conversion of complex organic molecules to ammonium) in about equal amounts. As discussed in Sections 2.2 and 2.3.5.1, crop production is not efficient in using Nr so only 30-70% (a global average of 40%) of all of the N mobilized for crop production is harvested in the crop. The remainder is stored in the soil, leached to aquatic systems as NO_3^-, volatilized to the atmosphere as NH_3 or NO_x or denitrified (see Section. 2.4) to produce NO_x, N_2O and N_2. An additional ~1.1 Tg of synthetic fertilizer N is used to maintain turfgrass in the urban environment (see Section 2.2.4) and another 0.1-0.2 Tg N is used to enhance forest production.

Table 13: Sources of reactive N input into terrestrial systems in the US in 2002 (from Table 1; in Tg N).

Source	Environmental System				
	Agricultural	Vegetated		Populated	Total
		Forest	Grassland		
Atmospheric	1.3	1.4	1.9	0.4	6.9*
N fixation	7.7	--	6.4	--	14.1
Synthetic N	9.7	0.1	**	1.1	10.9
Animal manure	1.2	--	3.8#	--	6.0#
Human sewage	0.1	--	--	1.2	1.3
Total Nr input	**20.0**	**1.5**	**12.1**	**2.7**	**39.2**

*The amount of atmospheric Nr deposition is based on area of each environmental system within the continental US. The total area does not sum to 100% because non arable lands are not included in this table.

**Synthetic fertilizer N used for managed pasture fertilization is included in the agricultural land classification.

#Unrecoverable livestock manure deposited on grasslands, the remaining N is assumed to be lost through ammonia volatilization, leaching or denitrification (EPA, 2007).

Within the nitrogen cascade (Figure 3), the interactions between the agricultural and populated portions of the terrestrial system dictate the production and flow of Nr. Although occupying the largest area, forest and grassland portions of terrestrial ecosystems serve mainly to absorb atmospheric deposition and provide a source of forest products and forage for livestock production. Reactive nitrogen input into these systems is from biological N fixation in unmanaged lands, atmospheric deposition and Nr from livestock manure that is deposited, while the livestock is grazing within grasslands (Table 13) may lead to the N saturation of unmanaged forest and grassland ecosystems (Galloway et al. 2004; Bobbink et al., 2009).

This report uses the Nr input numbers from Table 13 and food production numbers to estimate the flow of Nr through agricultural and populated parts of the terrestrial system (Table 14). The FAO (2008; www.fao.org/statistics/toptrade/trade.asp) lists the 20 largest agricultural commodities produced, imported and exported in the US in 2002. Of these commodities, corn (229 Tg), soybeans (75 Tg), wheat (44 Tg) and cow's milk (77 Tg) were produced in the greatest amount. Using commodity N content data (derived from data used to calculate crop residue N in the EPA (2007g) inventory of US greenhouse gas emissions and sinks, an estimated 9 Tg of N was marketed in three crops, soybeans (4.4 Tg N; from EPA, 2007g), corn (3.2 Tg N), and wheat (0.9 Tg N). Whole milk contained ~ 0.5 Tg of N while other meat and egg produce contained ~1.4 Tg of N, totaling ~ 1.9 Tg N. Grain, fruits, nuts and vegetables contained ~9.3 Tg of N. If the total N input use efficiency is 40% then ~23 Tg of N from all sources is required to produce 9.3 Tg of vegetative commodities. Table 14 lists the estimated Nr input into agricultural systems (~ 20 Tg) and additional N input from crop residue that was returned to the field the previous year (4.4 Tg) and from mineralization of soil organic matter (4.7 Tg). All of this N input totals ~29 Tg of N that is actually involved in the production of the 9.3 Tg of crop commodity N. If one assumes that return of crop residue to the field is directly proportional to crop production, then 24.3 Tg of N was required to produce the 9.3 Tg of crop commodity N. These estimates indicate that ~38% of the total annual input of N that went into the agricultural crop production system was contained in the main crop commodities produced in the US in 2002.

Of this 24.3 Tg N approximately 2.5 Tg was used to grow feed used for milk, egg and meat production. This estimate is made assuming that 4 units of N are required to produce a unit of milk, eggs or meat (see section 3.2.5.1.). This estimate also assumes that 1/3 of N required for livestock production comes from commodities in the FAO top 20 list and the remaining 2/3 comes from alfalfa, silage and grass over the course of a year (Oitjen and Beckett, 1996) Approximately 4.3 Tg of N in agricultural commodities (2.8 Tg in soybeans, corn and wheat) were exported, while ~0.15 Tg N was imported in various food and drink commodities. The US human populace consumed ~1.96 Tg of N in 2002 (292 million people, consume 114.7 g protein/person^{-}/day, 0.16 g N/g protein, 365 days) (approximately 1.2 Tg from animal protein-N and 0.7 from vegetative protein).

These three consumption areas, internal consumption of vegetable N for livestock production, human consumption, and export account for 77% of the commodities produced. The unaccounted for commodity N is likely partly in annual storage. Some smaller fraction of annual production is used for pet food and a small fraction is returned to the terrestrial environment because of spoilage and handling losses.

In forests and grasslands (vegetated system) N input in 2002 was ~3.5 Tg of anthropogenically introduced N, with the remaining ~10.1 Tg derived from BNF and livestock manure deposition. Of this anthropogenic N, ~21% was retained in soil and tree biomass while the remainder was removed in tree harvest (~0.2 Tg, see section 2.3.2.3) or lost to other parts of the environment through NH_3 volatilization and NO_3^- leaching and runoff (Table 14). Total N input into agricultural systems was ~20 Tg with ~ 11 Tg being removed as products which includes the transfer of ~2 Tg N as food to the human population. Almost 40% of the N input into agricultural systems is lost through NH_3 volatilization, nitrification/denitrification and NO_3^- runoff. The 4.2 Tg of Nr of Haber-Bosch N that is used for industrial feedstock is not included in this assessment. Of the input of ~3.3 Tg of N into the populated system ~80% is lost through human excreta processed in sewage treatment plants, denitrification in soils and leaching and runoff of NO_3^- (Table 14).

Table 14 summarizes the input and flow of Nr in the main terrestrial systems within the continental US. Anthropogenic input of Nr into forests and grasslands totaled ~3.5 Tg in 2002 with an estimated 6.4 Tg of Nr being introduced through natural biological N fixation. Of this Nr ~ 0.7 Tg was stored in vegetation and soils (see section 3.3.2) and ~2 Tg removed as livestock forage, while the remainder was lost to the atmosphere and aquatic systems, or removed as forest products and livestock forage. The largest anthropogenic Nr input (~20 Tg) was into agricultural production where ~11.2 Tg was removed as agricultural product, ~ 2 Tg transferred as edible product to the "populated" portion of the terrestrial system, ~0.8 Tg was stored in agricultural lands, and ~7.6 Tg N was lost to the atmosphere and aquatic systems. New N input into the "Populated" portion totaled ~3.3 Tg, which came from N transfer in food and use of fertilizer N in lawns, gardens and recreational areas. Within these areas an estimated 0.12 Tg was stored in urban forests.

Table 14: Nr input and flows (Tg N/yr) in the terrestrial portion of the Nitrogen Cascade (Figure 3) within the continental US in 2002

Environmental System*	N Input	N Storage**	Products	Transfers to Other Systems
Vegetated	13.6	0.7	2.2	10.7
Agricultural	19.6	0.8	11.2	7.6
Populated	3.3	0.1	0	3.2

*The Environmental Systems are those noted in the Terrestrial portion of the N Cascade shown in Figure 3. **Estimates are from section 2.3.2.

Finding 9

Although total N budgets within all terrestrial systems are highly uncertain, Nr transfers from grasslands and forests (vegetated) and urban (populated) portions of the N Cascade appear to be higher, on a per cent of input basis, than from agricultural lands. The relative amount of these transfers ascribed to leaching, runoff and denitrification, are as uncertain as the N budgets themselves.

Recommendation 9: *EPA should join with USDA, DOE, and universities should work together in efforts to ensure that the N budgets of terrestrial systems are properly quantified and that the magnitudes of at least the major transfer vectors are known.*

2.3.1.3 Transfer of Nr to aquatic systems

Within the nitrogen cascade, Nr flows from the atmosphere and terrestrial systems into aquatic systems. Aquatic systems include groundwater, wetlands, streams and rivers, lakes and the coastal marine environment. Nr is deposited directly into surface aquatic systems from the atmosphere (direct deposition) and Nr that is not either stored or removed as products on terrestrial systems eventually moves into aquatic systems (indirect deposition). What is the concern about too much Nr in aquatic systems? EPA's Office of Water (EPA, 2007d) notes the following reasons for implementing numerical water quality standards:

- *Excessive nutrients (nitrogen and phosphorus) can cause negative ecological impacts to water bodies on a national scale by stimulating harmful algal blooms.*
    - *Algal blooms block sunlight and result in the destruction of submerged aquatic vegetation which serves as critically important habitat and food for many organisms.*
    - *Algal blooms eventually die off and consume dissolved oxygen from the water column which can lead to die off of aquatic organisms.*
    - *One result of algal blooms is decreased biological diversity and populations, including smaller populations of game and commercial fish.*
    - *Some blooms, considered "harmful algal blooms" or "HABs", have a toxic effect on living organisms and are disruptive of ecosystem structure and transfer of energy to higher trophic levels.*

- *Excessive nutrients also pose public health risks.*
    - *Algal blooms can cause taste and odor problems in drinking water.*
    - *Hazardous algal blooms can cause respiratory distress and neurological problems in swimmers.*
    - *Excessive nitrates can cause blue baby syndrome.*

- *Nutrient pollution is occurring at a national scale and has not been completely addressed.*
    - *49 states and 4 territories have 303(d) listings due to nutrients, and about 50% of the states have greater than 100 water quality impairments due to nutrients.*
    - *Over 10,000 impairments are a result of nutrient pollution.*

Mitsch et al. (2001) suggest that streams and rivers themselves are not always as much affected by nutrient loading as are lakes, wetlands, coastal areas and other lentic bodies of water. However, in most cases, these nutrient-enriched waterways flow to the sea, with eutrophication of coastal waters the unfortunate result. This problem now occurs regularly throughout the world (WRI, 2007), in locations such as the Gulf of Mexico (Rabalais et al. 1996), the Baltic Sea (Larson et al. 1985), and the Black Sea (Tolmazin 1985).

During the past century, following large-scale use of synthetic N fertilizers in agriculture, rapid expansion of industrial and transportation-related fossil fuel combustion and coastal urbanization, humans have significantly altered the balance between "new" N inputs and N losses in the marine environment (Codispoti et al. 2001, Galloway and Cowling 2002). During this time frame, terrestrial discharge and atmospheric N emissions have increased by 10 fold (Howarth et al. 1996, Holland et al., 1999). This number keeps growing as human development continues to expand in coastal watersheds (Vitousek et al. 1997).

For at least 50 years, researchers have recognized this growing imbalance, especially in estuarine and coastal waters where anthropogenically-derived N over-enrichment has fueled accelerated primary production, or "cultural" eutrophication (Vollenweider et al. 1992, Nixon 1995). Eutrophication is a condition where nutrient-enhanced primary production exceeds the ability of higher ranked consumers and organic matter-degrading microbes to consume and process it. D'Elia (1987) characterized this condition as "too much of good thing" or over-fertilization of N-limited marine ecosystems with "new" N, a bulk of it being anthropogenic (Howarth et al. 1986, Vitousek et al. 1997, Galloway and Cowling 2002). Symptoms of N-driven eutrophication vary from subtle increases in plant production to changes in primary producer community composition, to rapidly accelerating algal growth, visible discoloration or blooms, losses in water clarity, increased consumption of oxygen, dissolved oxygen depletion (hypoxia), which is stressful to resident fauna and flora, or in the case of total dissolved oxygen depletion (anoxia), elimination of habitats (Paerl 1988, 1997, Diaz and Rosenberg 1995, Rabalais and Turner 2001). Other effects include submerged aquatic vegetation (SAV) losses, possible impacts on tidal wetland health, and disruption of estuarine food chain dynamics that may favor an imbalance towards lower trophic levels (e.g., jellyfish).

Anthropogenic or cultural eutrophication has been closely linked to population densities in coastal watersheds (Peierls et al., 1991, Nixon 1995, Vitousek et al. 1997). Primary sources of N enrichment include urban and agricultural land uses as well as wastewater treatment plants, most of which have not been designed to remove nutrients. A significant, and in many instances increasing, proportion of "new" N input can also be attributed to remote sources residing in airsheds. Delivery routes can also be complex, especially when via subsurface aquifers outside the immediate watershed, which can confound source definition and create long delays in delivery and management response (Paerl 1997, Jaworski et al., 1997, Galloway and Cowling 2002, Paerl et al. 2002).

The area of an airshed generally greatly exceeds that of a watershed for a specific estuary or coastal regions. For example, the airshed of the Baltic Sea includes much of western and central Europe (Asman 1994, Hov et al., 1994), while the airsheds of the US's two largest estuarine ecosystems, the Chesapeake Bay and Albemarle-Pamlico Sound, are 15 to over 30 times the size of their watersheds (Dennis 1997). Thus, the airshed of one region may impact the watershed and receiving waters of another, making eutrophication a regional-scale management issue (Paerl et al. 2002, Galloway and Cowling 2002). Furthermore, atmospheric N inputs do not stop at coastal margins. Along the North American Atlantic continental shelf, atmospheric N inputs more than match riverine inputs (Jaworski et al., 1997, Paerl et al., 2002), underscoring the fact that N-driven marine eutrophication may require regional or even global solutions. Even in truly oceanic locations (e.g. Bermuda), North American continental atmospheric N emissions (reduced and oxidized N) are commonly detected and significant (Luke and Dickerson 1987, Prospero et al. 1996). Likewise, islands in the North Pacific receive N deposition originating in Asia (Prospero et al., 1989).

Riverine and atmospheric "new" Nr inputs in the North Atlantic Ocean basin are at least equal and may exceed "new" Nr inputs by biological N_2 fixation (Howarth et al. 1996, Paerl and Whitall 1999, Paerl et al. 2002). Duce et al. (2008) estimate that up to a third of ocean's external Nr supply enters through atmospheric deposition. Schlesinger (2009) estimated that global atmospheric transport of Nr from land to sea accounts for the movement almost one third of the annual terrestrial Nr formation. This deposition leads to an estimated ~ 3% of new marine biological production and increased oceanic N_2O production. Therefore, our understanding of marine eutrophication dynamics, and their management, needs to consider a range of scales reflecting these inputs, including ecosystem, watershed, regional and global levels.

Scope of the Problem in the US. Over the past 25 years, there has been a growing recognition of cultural eutrophication as a serious problem in coastal estuaries (NRC, 2000). Globally, Selman et al. (2008) have reported "Of the 415 areas around the world identified as experiencing some form of eutrophication, 169 are hypoxic and only 13 systems are classified as 'systems in recovery'". Comprehensive surveys of US estuaries have been conducted by NOAA as part of the National Estuarine Eutrophication Assessments (NEEA) in 1999 and 2004 (Bricker et al. 1999; 2007). The most recent report, released in 2007 (Bricker et al., 2007) focused 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 for nearly 100 assessed US estuaries were that 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 a high chlorophyll *a* rating.

Further field evaluations by EPA and state and university collaborators under the National Coastal Assessment (NCA) using probabilistic monitoring techniques The NCA National Coastal Condition Reports (EPA, 2001, 2004 and 2006) are more closely related to nutrient enrichment assessments, especially for manifestations of nutrient enrichment such as hypoxia, nuisance algal blooms, and general habitat degradation. The last comprehensive national NCCR was published in 2004 (EPA, 2004) with a more recent assessment focused on 28 National Estuary Program estuaries published in 2007 (EPA, 2006). The 2004 NCCR included an overall rating of "fair" for estuaries, including the Great Lakes, based on evaluation of over 2000 sites. The water quality index, which incorporates nutrient effects primarily as chlorophyll-a and dissolved oxygen impacts, was also rated "fair" nationally. Forty percent of the sites were rated "good" for overall water quality, while 11% were "poor" and 49% "fair".

A recent evaluation of decadal-scale changes of NO_3^- concentrations in ground water supplies indicates that there is a significant increase in nitrate concentrations in well water across the US (Rupert, 2008). This study compared the nitrate content of 495 wells during 1988-1995 with nitrate content found during 2000-2004 as a part of the United States Geological Survey, National Water-Quality Assessment Program. From a subset of wells that had data on ground water recharge so that correlations with historic fertilizer use could be made, the study concluded that nitrate concentrations in ground water increased in response to the increase of N fertilizer use.

<u>Text Box 1: Hypoxia in the Gulf of Mexico</u>

An example of a problem of excess Nr that moves from one part of the US to another is the movement of Nr from the states that make up the Mississippi River drainage to the Gulf of Mexico. A hypoxic zone covers a significant area of the receiving bottom waters of the continental shelf of the northern Gulf of Mexico (details may be gleaned from SAB, 2007). This is a seasonally severe problem that has persisted there for at least the past 20 years. Between 1993 and 1999 the hypoxia zone ranged in extent from 13,000 to 20,000 km^2 (Rabalais et al. 1996, 1999, Rabalais and Turner 2001). The hypoxia is most widespread, persistent, and severe in June, July, and August, although its extent and timing can vary, in part because of the amplitude and timing of flow and subsequent nutrient loading from the Mississippi River Basin. The waters that discharge to the Gulf of Mexico originate in the watersheds of the Mississippi, Ohio, and Missouri Rivers (collectively described here as the Mississippi River Basin). With a total watershed of 3 million km2, this basin encompasses about 40% of the territory of the lower

48 states and accounts for 90% of the freshwater inflow to the Gulf of Mexico (Rabalais et al. 1996; Mitsch et al. 2001; EPA, 2007b).

The report, *Hypoxia in the Northern Gulf of Mexico. An update by the EPA Science Advisory Board.* December, 2007 (SAB, 2007) determined that "To reduce the size of the hypoxic zone and improve water quality in the Basin, the SAB Panel recommends a dual nutrient strategy targeting at least a 45% reduction in riverine total nitrogen flux (to approximately 870,000 metric tons/yr) and at least a 45% reduction in riverine total phosphorus flux (to approximately 75,000 metric tons/yr). Both of these reductions refer to changes measured against average flux over the 1980 - 1996 time period. For both nutrients, incremental annual reductions will be needed to achieve the 45% reduction goals over the long run. For nitrogen, the greatest emphasis should be placed on reducing spring flux, the time period most correlated with the size of the hypoxic zone."

Finding 10

Over the past 25 years, there has been a growing recognition of eutrophication as a serious problem in coastal estuaries (NRC, 2000). The last comprehensive national NCCR was published in 2004 (EPA, 2004) included an overall rating of "fair" for estuaries, including the Great Lakes, based on evaluation of over 2000 sites. The water quality index, which incorporates nutrient effects primarily as chlorophyll-a and dissolved oxygen impacts, was also rated "fair" nationally. Forty percent of the sites were rated "good" for overall water quality, while 11% were "poor" and 49% "fair".

Recommendation 10: *The committee recommends that EPA consider a range of scales reflecting ecosystem, watershed, and regional levels that include all inputs, e.g. atmospheric and riverine, of marine eutrophication dynamics and management.*

2.3.2. Storage of Nr within terrestrial ES

According to the nitrogen cascade conceptualization, terrestrial environmental systems are compartmentalized into agriculture, populated and vegetated systems. Annual input of Nr is greatest in agricultural ecosystems (farmland, cropland, and grazed pastureland) including Nr inputs, using 2002 as the base year, of 9.8 Tg from synthetic fertilizer, 7.7 Tg from biological N fixation in crops (mainly soybeans), and 1.3 Tg from atmospheric deposition. Nr input into vegetated systems (mostly forested, but including non-cropland grasslands and other natural vegetation types as well) comes mostly from atmospheric deposition (3.2 Tg). Annual input of Nr into populated systems includes synthetic fertilizer application to urban turfgrass and recreational areas (~1.1 Tg), and atmospheric deposition 0.2 Tg (Table 13.).

Much of the annual Nr input into these terrestrial systems passes through, and is transferred within, terrestrial systems or atmosphere via NH_3, NO_x or N_2O, or aquatic environmental systems via NO_3^- and organic N leaching and runoff or NH_x and NO_y deposition.

The largest single reservoir of total N in the terrestrial environmental system is soil organic matter (SOM). Approximately 52,000 Tg C and 4,300 Tg N are contained in the upper 100 cm of soil in the 48 contiguous states (N is estimated from assumed C/N ratio of 12) (Lal et al. 1998). For comparison, the total above ground biomass of US forests of these states contains ~ 15,300 Tg of C and ~ 59 Tg N (estimated using a C/N ratio of 261, and 15,500 Tg of SOM-C, 1290Tg total N (estimated using a C/N ratio of 12) (EPA, 2007g). Most of this SOM-N is bound within complex organic molecules that remain in the soil for tens to thousands of years. A small fraction of this SOM is mineralized, converted to carbon dioxide and Nr, annually. The total N contained within above and below ground compartments isn't really of concern. What is of interest in addressing issues of Nr, is the change in N stored within the compartments of terrestrial systems. The pertinent question is whether N is being retained or released from long-term storage. The committee evaluated estimates of annual change of N storage within important components of terrestrial systems. The EPA Inventory of US Greenhouse Gas Emissions and Sinks 1990-2005 (USEPA 2007g) carbon stock information obtained from chapter 7 of the report provided information used by the committee to estimate N storage in US terrestrial systems. Nitrogen stock change was determined by simply assigning a C/N ratio of 12 for soils and 261 for trees and making the appropriate conversions from C to N.

2.3.2.1. Agricultural

Croplands within the contiguous 48 states occupy ~149 million ha (19%) of the 785 million ha of land area, of which 126 million ha were cultivated in 2002 (NRCS, 2007; www.nrcs.usda.gov/technical/land/nrio3/national_landuse.html). Croplands are generally found on well drained mineral soils (organic C content 1-6% in the top 30 cm). Small areas of drained organic soils are cultivated (organic C content of 10-20%) in mainly Florida, Michigan and Minnesota (EPA, 2007g). Organic soils lost ~0.69 Tg of Nr in 2002 while mineral soils accumulated ~1.5 Tg of Nr (Table 15). Much of the accumulation of SOC was due to the use of conservation tillage and high yielding crop varieties (EPA, 2007g). Losses of Nr from organic soils are due to mineralization of SOM and release of Nr input. In cultivated soils annual input of new Nr is approximately 9.7 Tg from fertilizer N, 1.1 Tg from livestock manure (recycled N), ~7.7 Tg from biological N fixation and 1.2 Tg from atmospheric deposition. Assuming that loss of fertilizer N from the small area of organic soils is a minor fraction of the total, then ~17% of N input from synthetic fertilizer, ~12% of total N input, is stored in cropland mineral soils annually.

2.3.2.2. Populated systems—urban lands

Populated or "developed land" (developed land is the terminology used by NRCS) occupied ~42.9 million ha of the US land area in 2002. This equates to approximately 5.5% of the US land area (NRCS, 2007). The EPA Inventory of U.S. Greenhouse Gas Emissions and Sinks (EPA, 2008) indicates that urban areas cover over 4.4% of the land area with tree canopy covering 27.1% of the urban area. The tree-covered area constitutes approximately 3% of total tree cover in the continental U.S. If the NRCS value of 42.9 million ha is used, then trees cover ~11.3 million ha of urban land in the contiguous 48 states. Another ~ 14.2 million ha of land is covered by turf grass in parks, golf courses, and lawns. In both urban forests and turf grass, Nr storage is dependent upon the age of the trees or turf. In young, pre-steady state, systems N is being accumulated while at steady state no net change occurs. Some areas may be degrading and actually loosing biomass and returning N to the environment. EPA (2007g) does not estimate carbon changes in turf grass, but does estimate changes in carbon storage in urban forests. Urban trees sequestered an estimated net 22 Tg of carbon and 0.12 Tg of N in 2002 (using the hardwood C/N ratio of 186) (EPA, 2007g). Annual fertilizer N input into the urban landscape is approximately 10% of total fertilizer N consumption in the U.S. (EPA, 2007g), or ~1 Tg of N in 2002. Another 0.2-1.0 Tg N is deposited from atmospheric deposition, which can be disproportionately high due to locally high NO_y concentrations. Storage of ~0.12 Tg N in urban forests constituted approximately 3% of Nr input annually.

2.3.2.3. Vegetated systems—forests and grasslands

Forests. Forests cover approximately 164 million ha, ~21% of the land area of the contiguous 48 states (NRCS, 2007). The forest carbon stocks analysis by EPA (2007g) is based on state surveys that are conducted every 1 to 10 years. Annual averages are applied to years between surveys. Changes in C related to the rate of tree growth, the highest rate is in the Pacific northwest. Birdsey (1992) estimated that 52,500 Tg of C in above and below ground in U.S. forests; soil contains 59% of total C, 9% in litter and 5% in tree roots**.** The EPA estimate for 2002 is 43,600 Tg of C. For the following N storage estimate, based on EPA (2007g) data, we have assumed that forests are 85% softwood and 15% hardwood with an average C/N ratio = 261. These estimates indicate that forests and forest products stored ~0.43 Tg of N in 2002 (Table 15)

Grasslands. Grasslands, including rangelands and pasturelands, occupy approximately 213 million ha (27.1%) of the contiguous 48 state land area. The NRCS divides these grasslands into pastureland (48.2 million ha) and rangeland (164 million ha). Pastureland is managed, may be fertilized and mown, and rangeland is managed only to the extent that livestock grazing intensity on the land used for livestock grazing is regulated. Changes in the N status of grasslands are dependent upon changes in soil organic matter as the above ground biomass produced annually is either consumed by livestock or decomposed in the field. Soil organic C stocks were estimated using the Century biogeochemical model and data used were based upon the NRCS/National Resources Inventory (NRI) survey (EPA, 2007g). Changes in soil N content were estimated using a C/N ratio = 12. Nitrogen input into rangelands is generally only from atmospheric deposition, which contributes 1.9 Tg N each year to range production (Table 15). Rangeland tends to be in relatively remote areas where atmospheric Nr deposition is low

Collectively, forests and grasslands stored ~0.74 Tg of N in 2002. Much of the soil N storage in grasslands is a result of conversion of croplands to grasslands, mainly due to the conservation reserve program. Forest soils appear to be loosing N while overall N storage is from accumulation in above ground biomass and that that remains in forest products that are stored for long periods.

2.3.2.4. Summary of estimates of Nr stored in terrestrial systems in 2002

An estimated 1.7 Tg of N was stored in the terrestrial systems of the contiguous 48 states in 2002 (Table 15). Soils were the largest reservoir with croplands (0.82) and grasslands (0.31) sequestering most of the N. Estimated total Nr input from synthetic fertilizer, biological N fixation and atmospheric deposition into terrestrial systems within the contiguous 48 states in 2002 was ~32 Tg . Although uncertainty of the storage estimate needs to be assessed, it is probably at least +/-50%. Annual storage in agricultural, grassland and forest soil and in forest biomass is approximately 6 to 10 % of annual Nr input. All of the input and outflow numbers are highly uncertain, but N loss through denitrification appears to be the major loss mechanism. As with the 16 northeastern US watershed example, discussed in section 2.3.3, and, as concluded in a recent global Nr review by Schlesinger (2009), storage in soils and trees accounts for only a small portion of the annual N input while apparent loss through denitrification dominates the budget. Some small fraction is re-volatilized and exported from the continent.

Table 15: Net Annual Change in Continental US Croplands soil C and N, Forest C and N, and Grassland Soil C and N in 2002

Measurements in Tg. Negative sign indicates a decrease in storage: positive number indicates increase in storage, soil C/N ratio = 12; wood C/N = 261 (C storage numbers were obtained from EPA, 2007g).

	C	N
Cropland		
Cropland remaining cropland		
Mineral soil	17	1.4*
Organic soil	-8.3	-0.69
Land converted to cropland	0.8	0.067
Total	9.6	0.80
Forests		
Forests and harvested wood products		
Above ground biomass	85	0.32
Belowground biomass	16	0.063
Dead wood	9.1	0.035
Litter	7.2	0.028
Soil organic matter	-2.8	-0.23
Harvested Wood	59	0.22
Total	173	0.43
Grasslands		
Grasslands remaining grasslands		
Mineral soil	-0.8	-0.067
Organic soil	-1.3	-0.11
Lands Converted to Grasslands	5.8	0.48
Total	3.7	0.31
US Total C & N Storage in 2002	186	1.7[7]

[7] According to the USEPA National Greenhouse Inventory (EPA 2007b) the net increase in soil C stocks over the period from 1990 through 2005 was largely due to an increase in annual cropland enrolled in the Conservation Reserve Program, intensification of crop production by limiting the use of bare-summer fallow in semi-arid regions, increased hay production, and adoption of conservation tillage (i.e., reduced- and no-till practices). The above EPA estimates assume that no-till crop production results in net carbon sequestration. Recent publications indicate, however, that no-till cropping practices do not result in net carbon sequestration (Baker et al. 2007; Blanco-Canqui, H. and R. Lal. 2008; Verma et al., 2005), which means the above estimates of soil C and N storage in mineral soils may need to be reconsidered. These new studies suggest that organic C conservation by reduced tillage practices has been overestimated because soil sampling and analysis has been confined to the top 30 cm of soil when the top meter of soil needs to be considered. Baker et al. and Verma et al. also show that long-term, continuous gas exchange measurements have not detected C gain due to no-till. They concluded that although there are other good reasons to use no-till, evidence that it promotes C sequestration is not compelling. These findings highlight the need for appropriate assessment of ecosystem N storage so that this committee's conclusion that only a small part of annual Nr input is stored in agricultural lands, forests, and grasslands can be confirmed or disproven.

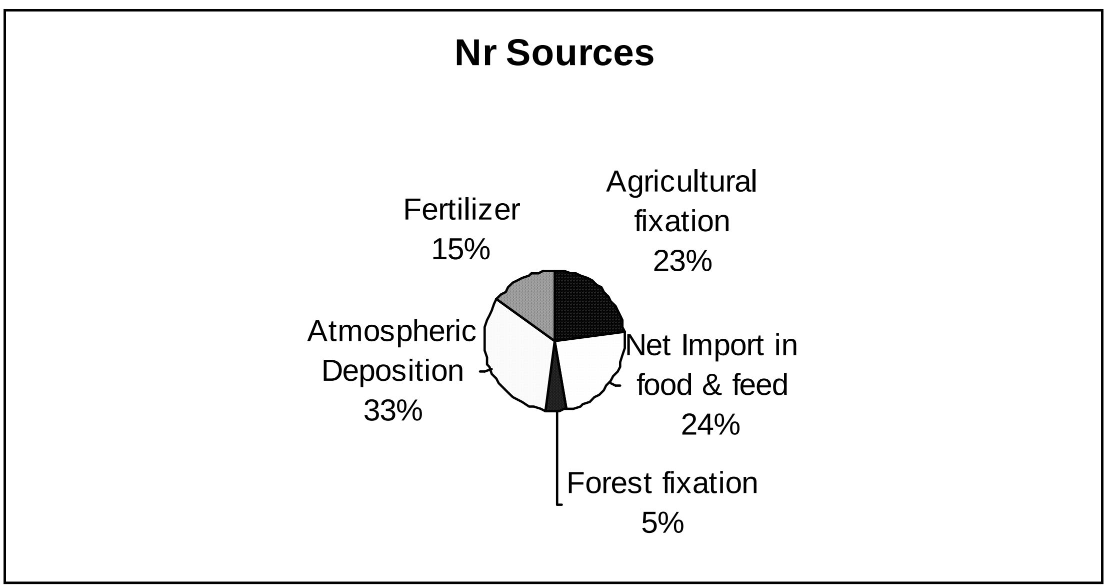
2.3.3 Input and fate of Nr in 16 watersheds in the northeast US.

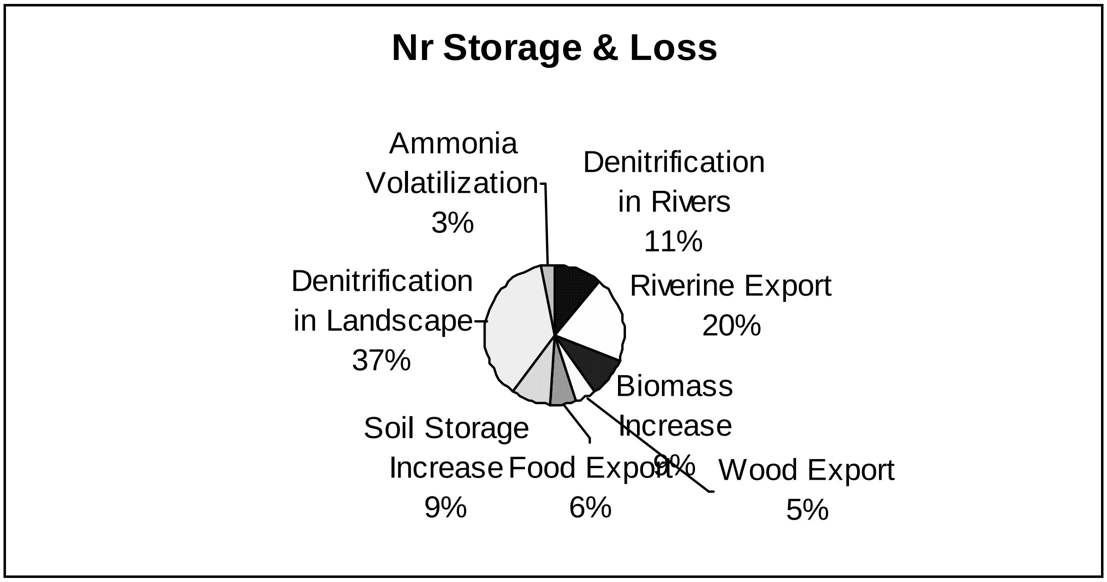
There are no comprehensive data available to assess the transfer and transformations in and between the atmosphere, terrestrial systems (agriculture, populated and vegetated systems) and aquatic systems nationally. Determining a national N budget is a priority research area. As there are no national data available, an example analysis of Nr input and fate in 16 watersheds in the northeast US, for which data are available, is used to show an evaluation of the inputs and fate of Nr for a large watershed (Fig. 3-14) (Van Breemen et al. 2002).

The watersheds in this study encompass a range of climatic variability, Maine to Virginia. The watersheds are a major drainage to the coast of the North Atlantic Ocean. Using data from the early 1990s, Boyer et al. (2002) reported the quantification of N inputs to each watershed from atmospheric deposition, N fertilizers, biological N fixation, and import of N in agricultural products (food and feed). They compared inputs with N losses from the system in riverine export. As a part of the same study, Van Breemen et al. (2002) analyzed the fate of N inputs to these watersheds and developed budgets for each watershed. The total area of the watersheds was 32,666 km^2 with land use categories of forest (72%), agricultural (19%), urban (3%), wetlands (5%), and 1% other uses. The Nr input into the watersheds (using weighted averages for all 16 watersheds) was 3,420 kg per square km per year (hereafter expressed as kg /km^2 /yr). Figure 16 shows the Nr sources and the estimated fate of this Nr as a per cent of the weighted average Nr input.

<u>Figure 16: Nr input and loss from 16 watersheds in the northeast US.</u>

The Nr input into the watersheds (using weighted averages for all 16 watersheds) was 3,420 kg /km^2/yr (Van Breemen et al. 2002).





Van Breemen et al. (2002) indicate that Nr inputs and storages and losses were well correlated (R^2 = 0.98). Denitrification in landscape soils is the most uncertain estimate, because rates are calculated by difference between total inputs and outputs, so they accumulate errors from all estimates. They suggest that the denitrification loss term may also reflect the change in N storage in groundwater. The net storage of N in the soil (18% of total storage and losses) indicates that there is a non-steady state condition in the soil. Increasing storage of Nr on land implies that drainage and denitrification exports of Nr are likely to increase when a new steady state condition is reached.

These data suggest that Nr research need to focus on understanding the "denitrification" loss term in this analysis. The losses occur in the terrestrial landscape, before Nr enters the river. Where do these losses occur, within the agricultural field, in drains and ditches near the agricultural field, in riparian areas, or wetlands? Understanding this term may help in the management of Nr in watersheds to decrease nitrate movement into aquatic systems as well as to limit N_2O emissions to the atmosphere.

The Van Breemen et al. (2002) study also estimated that approximately 30% of N input was exported to the rivers and about two thirds (20% of total N input) of this N was exported to coastal waters by rivers. The remaining one third (11% of total N input) was considered to have been denitrified in the rivers. These examples also demonstrate that Nr in the atmosphere, terrestrial systems and aquatic systems are not separate and must be considered collectively. Atmospheric deposition is a variable, but important input into aquatic systems that contributes to Nr enrichment problems. Aquatic and terrestrial systems process this Nr and return other Nr gases (NH_3, NO_x and N_2O to the atmosphere). Nr from terrestrial systems impacts both the atmosphere and aquatic systems through emission of NH_3, NO_x, N_2O and leaching and runoff of NO_3^-.

<u>Finding 11</u>

Denitrification of Nr in terrestrial and aquatic systems is one of the most uncertain parts of the nitrogen cycle. Denitrification is generally considered to be a dominant N loss pathway in both terrestrial and aquatic systems, but it is poorly quantified

Recommendation 11: *EPA, USDA, DOE, and universities should work together to ensure that denitrification in soils and aquatic systems is properly quantified, by funding appropriate research.*

2.3.4. Areas of great uncertainty in Nr transfer and transformations.

In developing the discussion of Nr transfers and transformations in and between the environmental systems of the nitrogen cascade, the committee has encountered a number of areas where quantities or flows of Nr are highly uncertain. All of these areas need attention from EPA in conjunction with other federal and state agencies and universities. Although most of the following points have been highlighted in various "Findings" and "Recommendations," within chapter three of this report we feel the need to highlight the following areas:

- Total denitrification in animal feeding operations, in soils, and in aquatic systems needs to be quantified along with all gaseous products produced and released to the atmosphere during nitrification/denitrification. These gases include NO_x, N_2O and N_2.
- The amount of Nr transferred to each environmental system as dry deposition needs to be quantified and monitored.
- The fraction of NOy in the form of organo-nitrates and other organic nitrogen species is poorly quantified, but may play a major role in air quality and Nr cycling.
- Rates and amounts of ammonia emissions from fertilized soils and animal feeding operations need to be quantified and the fate of this ammonia determined.
- The annual change in N storage in soils (agricultural, forest , grassland and urban areas) needs to be quantified in conjunction with the change in carbon

These areas of high uncertainty are highlighted because very little information exists in some of the areas while in other areas, such as denitrification and the relative release of N_2O from soils and aquatic systems, the sparse data are highly variable which makes developing meaningful guidelines for control difficult.

2.4 Impacts, metrics, and current risk reduction strategies for Nr

2.4.1 Measurement of Nr in the environment

Although N 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^-.

<u>Finding 12</u>

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. Since 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 12. *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 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, with parallel interests in advancing the necessary underlying science of Nr creation, transport and transformation, impacts, and management.*

2.4.2 General considerations for Nr impacts

2.4.2.1 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 3, 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 risk 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. Risk "end points" are typically established through reference to supporting scientific studies, location-specific conditions, and economic, safety, and social factors.

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.

2.4.2.2 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 16 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 this context INC 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 plans (USEPA, 2009a). Recently, the Science Advisory Board completed a self-initiated study on "Valuing the Protection of Ecological Systems and Services" (EPA 2009b). This report explores the concept of ecosystem services as a basis for regulatory action and presents a roadmap for implementing this approach.

Table 16: Ecosystem service and corresponding function categories (Costanza et al. 1997)

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

2.4.2.3 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 (see Chesapeake Bay box), 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.

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 ecosystem services 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 13

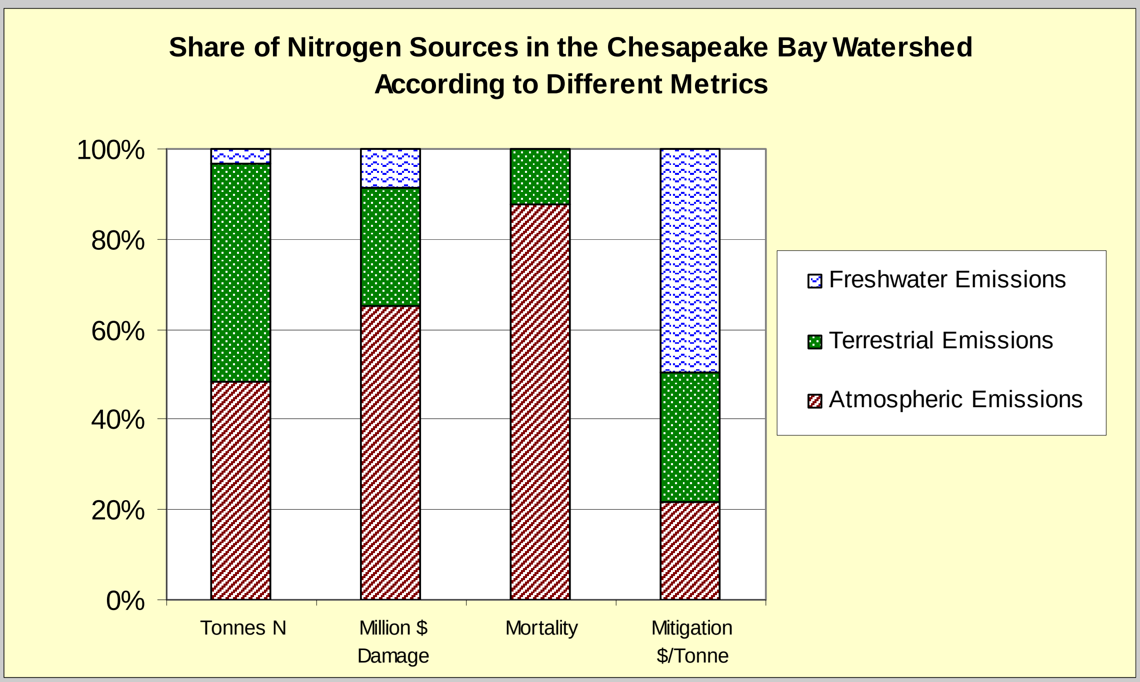
The committee finds that reliance on only one approach for categorizing the measurement of Nr is unlikely to result in the desired outcome of translating N-induced degradation into the level of understanding needed to develop support for implementing effective Nr management strategies.

Recommendation 13. *It is, therefore, recommended that the EPA 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.*

Text Box 2: Economic Impacts and Metrics for Chesapeake Bay

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). 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 17 below. Note that approximately 48% of N entering the watershed is coming through emissions to the atmosphere, but they are causing 65% of the dollar damages and 88% of the human mortality. A nearly equal percentage, 49%, of the Nr involves runoff from the land, but it accounts for only 26% of the damage costs and 12% of the mortality. Fresh water releases of Nr account for only 3% of the Nr and 9% of the cost damages and contribute nothing to mortality losses. Hence freshwater releases in the Chesapeake Bay ecosystem cause the smallest damage but account for the largest cost per MT to mitigate. Costs of reactive N mitigation provide an additional economic measure of the cost effectiveness of actions to reduce a ton of N.

Figure 17: Relative importance of all reactive nitrogen sources in the Chesapeake Bay Watershed according to four different metrics.



The metrics are broken down further by the specific source of NO_x and NO_y emissions into each of the three media in Table 17

Table 17: Alternative metrics for different atmospheric emissions and for terrestrial and freshwater releases of reactive NO_x and NO by source

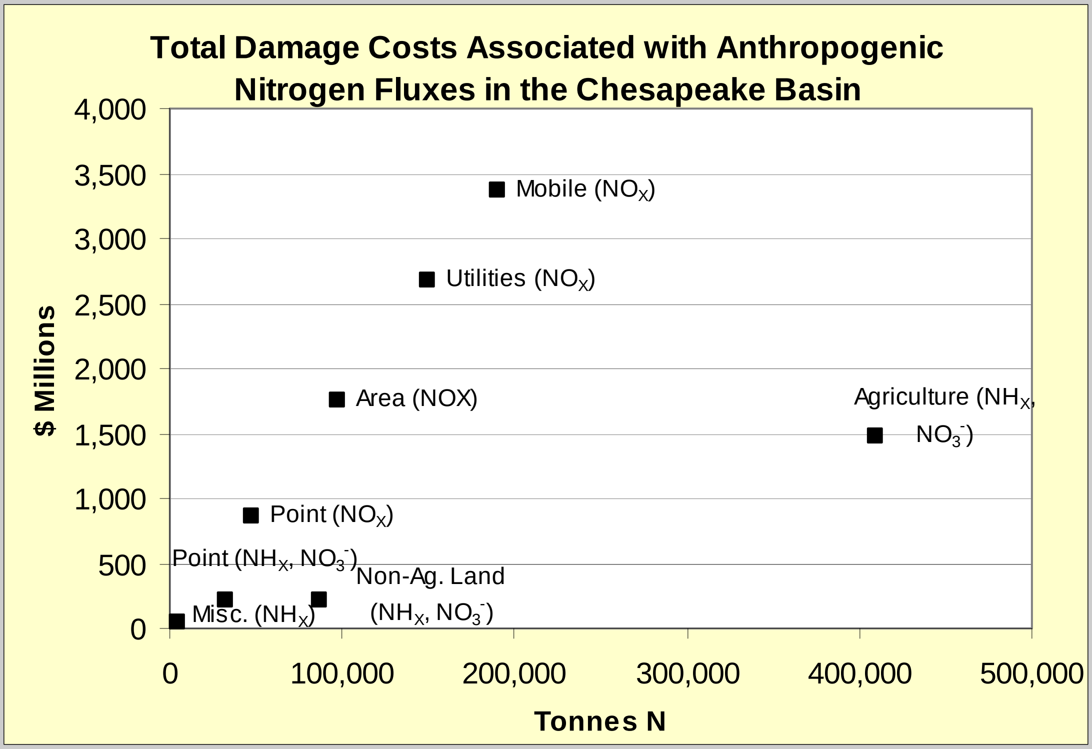
	Nr (mt)	$ Damage (millions)	Mortality	Mitigation ($/mt)
Atmospheric - Utility Emissions	150,000	510	**309**	$6,500
Atmospheric - Mobile Source Emissions	190,000	642	**389**	$15,000
Atmospheric - Point Source Emissions	48,000	162	**98**	$23,000
Atmospheric - Area Source Emissions	98,000	334	**203**	$5,100
Terrestrial Emissions	490,000	668	**141**	$11,000
Freshwater Emissions	32,000	223	**0**	$19,000

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 Nr releases from the land even though the two sources are comparable in scale in terms of reactive forms of N released to the watershed. This difference occurs because emissions to the air cascade through more parts of the watershed ecosystem than do releases directly to the Bay. If human health effects are monetized, then the economic gains are even greater from reducing atmospheric emissions. See Figure 18.

Looking at the remediation cost of controlling releases to the environment, the least costly per ton of Nr or per dollar saved 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. These 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 could be quantified, but the necessary data have yet to be collected. Economic losses due to damage to recreational and commercial fisheries in the Bay and in freshwater are examples that are likely to be significant but have not yet been quantified.

And, finally, 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 management strategies are effectively minimizing risks and implemented.

Figure 18: Total damage costs associated with anthropogenic nitrogen fluxes in the Chesapeake Basin



Scatter plot of all quantifiable damage costs (including health impacts) relative to tons of Nr showing the significant difference in emphasis of the two different metrics

Similarly, economic losses due to climate change and ozone depletion from N_2O emissions have not been evaluated, as have a variety of other environmental and health effects. Other parts of the country such as the Mississippi Valley would show a very different pattern 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.

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 CAA as the CWA. This challenges our traditional approach to regulation, but that is a consequence of comprehensively examining Nr guided by the nitrogen cascade.

2.4.3 Reactive nitrogen and aquatic ecosystems

2.4.3.1 Impacts of Nr on aquatic systems

The availability of N controls primary production in much of the world's estuarine, near-shore coastal and open ocean waters (Dugdale 1967, Ryther and Dunstan 1971, Nixon 1995, Paerl 1997; Boesch et al. 2001). Nitrogen can also play a role as either a primary or secondary limiting nutrient in freshwater environments, especially large lakes (e.g. L. Tahoe, L. Superior). As such, the fertility of these waters is often closely controlled by N inputs, which are provided either internally by regeneration of pre-existing N and biologically-fixed atmospheric N_2, or supplied externally (i.e. "new" N) as combined N sources delivered via surface runoff, sub-surface groundwater or atmospheric deposition.

The extent to which accelerated N loading promotes eutrophication and its symptoms varies greatly among marine ecosystems. Receiving waters exhibit variable sensitivities to N and other nutrient [phosphorus (P), iron (Fe), and silica (Si)] loads that are controlled by their size, hydrologic properties (e.g. flushing rates and residence times), morphologies (depth, volume), vertical mixing characteristics, geographic and climatic regimes and conditions. In addition, the magnitude and distribution of N in relation to other nutrient loads can vary substantially. In waters receiving very high N loads relative to requirements for sustaining primary and secondary production, other nutrient limitations may develop. This appears to be the case in coastal waters downstream of rivers draining agricultural regions that are enriched in N.

On the ecosystem level, estuarine and coastal waters exhibit individualistic responses to N loads over seasonal and longer (multi-annual, decadal) time scales. The degree to which these systems are exposed to freshwater discharge, tidal exchange and vertical mixing is critical for determining how they respond to specific N loads (Vollenweider et al., 1992, Nixon 1995, Cloern 1999, 2001; Valdes-Weaver et al. 2006; Paerl et al. 2007). Another variable is the manner in which N loading takes place, which may range from acute pulsed events such as storms and associated flooding, to longer-term gradual (chronic) increases in N loading associated with more predictive seasonal, annual and inter-annual hydrologic cycles. There are striking contrasts in ecosystem response to N inputs that reflect a range in physical (hydrodynamic, optical) and climatic conditions (Cloern 1999, 2001). Examples include contrasts between strong tidally-driven estuarine systems, such as Delaware Bay and San Francisco Bay, and non-tidal, lagoonal systems, such as North Carolina's Pamlico Sound and Texas's Luguna Madre, or semi enclosed coastal systems, such as Florida Bay and the Long Island Sound (Bricker et al., 1999; Valdes-Weaver et al. 2006; Paerl et al. 2007).

Externally-supplied N comes in various forms, including organic N and inorganic reduced (NH_3 and NH_4^+ ion) and oxidized (NO_3^-) N, all of which are potentially available to support new production and eutrophication. Laboratory experiments on phytoplankton isolates and bioassays with natural phytoplankton communities have indicated that these contrasting forms may be differentially and preferentially utilized, indicating that, depending on composition of the affected phytoplankton community, some forms are more reactive than others (Collos, 1989; Stolte et al., 1994: Riegman, 1998). Phytoplankton community composition can also be altered by varying proportions and supply rates of different forms of N (Dortch, 1990; Stolte at al., 1994; Harrington, 1999; Pinckney et al., 1999; Piehler et al., 2002). Monitoring and research on dissolved organic N inputs and their effects should be conducted in receiving streams, rivers, lakes, estuarine and coastal waters, since there is evidence that these compounds can be utilized by phytoplankton, including harrmful bloom species (Paerl 1988, Antia et al. 1991, Carlsson and Granéli 1998, Gilbert et al. 2006). In addition, specific N compounds may interact with light availability, hydrodynamics and other nutrients, most notably P, Si, Fe, and trace metals, to influence phytoplankton community growth rates and composition (Harrison & Turpin, 1982; Smith, 1990, Dortch & Whitledge, 1992).

One example of shifting N inputs is the proliferation of intensive livestock operations in coastal watersheds, which has led to large increases and changes in chemical composition of nitrogenous compounds discharged to estuarine and coastal waters via runoff, groundwater and atmospheric deposition (Paerl, 1997; Howarth, 1998; Galloway & Cowling, 2002). In general, coastal waters under the influence of these operations are experiencing increases in total N loading as well as a shift toward more reduced N (NH_4^+, organic N) relative to oxidized N (NO_3^-) (Howarth et al., 2002; Galloway & Cowling, 2002). These increases, combined with increases in hypoxia and anoxia in receiving waters, are leading to more NH_4^+-rich conditions, which will favor algal groups able to best exploit this N form, including some harmful algal bloom (HAB) taxa (Paerl and Whitall 1999; Paerl et al. 2007). Similarly, conversion of forest and agricultural lands to urban lands can alter landscapes and promote N loading to estuaries by increasing impervious pathways and removing natural landscape filters for Nr. Development also destroys and eliminates wetlands, leading to more NO_3^--enriched conditions, potentially favoring plant taxa best able to exploit this N form.

2.4.3.2 Water quality regulation and management

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”, “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 Sec. 131.2). Further, “Such standards serve the dual purposes of establishing the water quality goals for a specific water body and serve 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 Sec. 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 antidegradation policy (40 CFR Sec. 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, hereafter 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 (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 (EPA, 1998) mandated numeric nutrient criteria to begin to address the problem (EPA 1999). To move the objectives of the Clean Water Action Plan forward, EPA published national nutrient criteria guidance for lakes and reservoirs (EPA 2000b), rivers and streams (EPA 2000c), estuaries and coastal waters (EPA 2001c), and wetlands (EPA 2007e), 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.

Multimedia and multijurisdictional N management can be complicated because the CWA has little authority over atmospheric sources, and individual states explicitly lack authority to control upstream sources. Quite often in estuaries such as the Gulf of Mexico or Chesapeake Bay, management goals that meet water quality standards cannot be attained without interstate compacts or a strong federal role that 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.

2.4.3.3 Aquatic thresholds for Nr

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 (Table 19), 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 (EPA 2000b; 2000c; 2001c; and 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 US) has no regulatory framework but rather sets the threshold of Nr loading at which negative 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., 2009). There are numerous locations within the U.S. where deposition to surface waters falls within this range.

2.4.3.4 Water management in urbanized areas

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 imported food). 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 14

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.

Recommendation 14. *To better address Nr runoff and discharges from the peopled landscape the committee recommends that EPA*:

14a. *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 with special attention to the need to regulate nonpoint source and related land use practices.*

14b. *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.*

14c. *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.*

14d. *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.*

14e. *Use ecosystem-based management approaches that balance natural and anthropogenic needs and presence in the landscape.*

2.4.3.5 Attainment of water quality management goals and standards

Estuarine systems, where bio-available Nr is more likely to be the limiting nutrient, are most often susceptible to Nr enrichment (Paerl 1997; Boesch et al. 2001). Defining single number criteria for nutrients or related indicators representative of undesirable levels of productivity (e.g., chlorophyll *a*) is difficult, even using the ecoregional approach recommended by EPA. State managers more often use the formal TMDL process or collaborative estuarine management plans to set site- or estuary- specific N management targets to meet existing, related water quality criteria (e.g., dissolved O_2 or chlorophyll *a*). Some of the more prominent efforts and targets for nitrogen control are summarized in Table 18.

Table 18: Estuaries with nitrogen management plans or TMDLs and percent nitrogen load reduction targets

Estuary	Nitrogen Load Reduction Target	TMDL or Plan
Casco Bay, Maine	45%	Plan
Chesapeake Bay	>40%	Plan
Northern Gulf of Mexico Mississippi Plume Region	45%	Plan
Long Island Sound	60% for CT & NY sources	TMDL
Neuse River Estuary, NC	30%	Plan
Tampa Bay, FL	Maintain TN load at 1992-1994 levels	TMDL & Plan

These targets all exceed the national estimates for nitrogen load reductions the INC has identified in this report, which are generally less than 25% from specific source categories. Since not all sources offer management opportunities, the expectation is that reductions in Nr loadings to estuaries would cumulatively be less than 25%, which is below the targets identified in Table 18. Many of the management actions the committee has proposed would also require substantive changes in national programs, regulatory authority, management technologies and societal demands to be accomplished. This is a nutrient management concern state managers are well aware of as they develop TMDLs and management plans that range above attainment potential, not only for Nr but more frequently for other pollutants that are predominately nonpoint source and stormwater loaded (including atmospheric source contributions).

The Chesapeake Bay Program, for example, is a model for Nr and P management in many ways. Considerable resources were committed, and many BMPs implemented, with disappointing results. Despite regional efforts and commitments from all watershed states, and more funding than any other estuary program is likely to see, management targets have not been met, and recent data (2007) reveal the occurrence of a severe hypoxic episode. Concerns over the slow progress in restoring the Chesapeake Bay led to the issuance of an Executive Order on May 15, 2009, establishing a Federal Leadership Committee led by the EPA to develop and implement a plan to restore the Bay in collaboration with state agencies (Federal Register: 74(93): 23097-23104). Similarly, the adoption of the Long Island Sound TMDL (See text box 3) sets an implementation plan that could attain Connecticut and New York dissolved oxygen criteria, but only if “alternative technologies” such as mechanical aeration of the Sound or biological harvesting of nutrients, are used.

Finding 15

Meeting Nr management goals for estuaries, when a balance should be struck between economic, societal and environmental needs, under current federal law seems unlikely. Enforceable authorities over nonpoint source, stormwater, air (in terms of critical loads), and land use are not adequate to support necessary Nr controls. Funding programs are presently inadequate to meet existing pollution control needs. Furthermore, new technologies and management approaches are required to meet ambitious Nr control needs aimed at restoring national water quality.

Recommendation 15. *INC recommends that EPA reevaluate water quality management approaches to ensure Nr management goals are attainable, enforceable, and affordable and that monitoring and research are adequate to problem definition and resolution, particularly in the development of nitrogen removal technologies. This may require changes in the way EPA sets water quality criteria and some compromises in ecosystem goals to accommodate human uses of the air, land and water.*

Text Box 3: Long Island Sound Total Maximum Daily Load (TMDL)

A TMDL sets a goal for reducing the load of a specific pollutant that is causing impairment to a waterbody. In the case of Long Island Sound, the impairment constitutes low concentrations of dissolved O_2 that violate both Connecticut's and New York's water quality standards. Nitrogen has been identified as the pollutant that causes substandard levels of dissolved oxygen in Long Island Sound and, accordingly, Connecticut's and New York's environmental agencies have developed a TMDL that assigns nitrogen reductions from both point sources (the wasteload allocation or WLA) and nonpoint sources (the load allocation or LA) in their respective states to meet the established 58.5% reduction of anthropogenic sources.

The Long Island Sound TMDL is set at 23,966 tons of N/year, which represents a 23,834 ton/year reduction from the total baseline (anthropogenic + natural sources considered) of 47,788 tons/year from Connecticut and New York only. Most of that N load comes from point sources – POTWs (publicly owned treatment works) and CSOs (combined sewer overflows) – accounting for 38,899 tons/yr of the total N load from the two states, or 81% of the load. For that reason, the focus has been on managing point sources, although attainment of water quality standards will require more widespread reductions from atmospheric deposition, stormwater and nonpoint sources, and from other watershed states north of Connecticut.

Connecticut and New York have some flexibility in the apportionment of those reductions between the WLA and the LA, but must have completed 40% of the required reductions by 2004, 75% by 2009 and 100% by 2014 when the final TMDL will be met. However, the TMDL is presently undergoing revision to incorporate findings from a new model of Long Island Sound, and to reflect changes in dissolved O_2 criteria in both states. The revised TMDL will likely require more aggressive reductions of nitrogen to meet dissolved O_2 criteria and may formalize targets for upstream state contributions and atmospheric deposition.

2.4.3.6 Water quality monitoring and assessment

Under Sec. 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 Sec. 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 has published these reports up until 1998 (EPA 2000a), after which it transitioned into a Water Quality Report in 2000 (EPA 2002) and a National Assessment Database in 2002 (http://www.epa.gov/waters/305b/index.html. States also prepare a list of "impaired" waters under Sec. 303(d) of the CWA (EPA, 1999). Subsequent reports will provide a synthesis of CWA Sec. 305(b) and 303(d) reporting under a Consolidated Assessment and Listing Methodology or "CALM" approach.

The EPA compiles the approved state 303(d) lists into a national listing (http://iaspub.epa.gov/waters/national_rept.control). 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 2004. The report identifies 5,617 impairments related to "nutrients" (almost 9% of all identified impairments), although other impairments may ultimately have a nutrient enrichment cause. For example, oxygen depletion (4,540), turbidity (2,050), algal growth (510), ammonia (generally toxicity – 416), and HABS (4) 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 (WSA) (EPA 2006a), the National Coastal Assessment (NCA) and its National Coastal Condition Reports (EPA 2001a, 2004, and 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 (http://www.epa.gov/owow/monitoring/reporting.html). Many of these are aimed at including a biological assessment component that is often lacking in water pollutant and chemistry efforts described above.

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.

Finding 16

The committee has determined that an integrated approach to monitoring that includes multimedia (air, land and water) components and considers a suite of environmental and human concerns (e.g., Nr effects, climate change, human health) would be most useful and efficient. Some of the phenomena that we present in this report simply need more definition and verification but, more importantly, as control is brought to bear on Nr, improvements need to be measured (i.e. monitored) to validate the success of one control or another. If the desired improvements are not realized as shown by the collected data, corrective measures will be required. The pool of data would be used to formulate new management procedures. The process of monitoring and control revisions is termed adaptive management—a process that INC supports as it does not delay actions that can be taken immediately, but acknowledges the likelihood that management programs will be altered (adapted) as scientific and management understanding improve**.**

Recommendation 16. *The committee recommends that EPA initiate discussions and take action to develop a national, multimedia monitoring program that monitors sources, transport and transition, effects using indicators where possible, and sinks of Nr in keeping with the nitrogen cascade concept. This comprehensive program should build upon existing EPA and state initiatives as well as monitoring networks already underway in other federal agencies such as the U.S. Geological Survey programs and the NADP effort.*

2.4.4 Reactive nitrogen and air quality

2.4.4.1 Impacts of Nr on atmospheric systems

The atmosphere is 78% N, mostly N_2, with just trace amount of Nr. The former has a long residence time (millions of years) and has no negative impact on ecosystems or humans. With one exception, the N species that constitute Nr (see footnote in the Executive Summary and Chapter 2) have residence times of less than a year (many on the order of days), and contribute to all the negative impacts associated with excess Nr in the atmosphere. In addition, the exception (N_2O), with a residence time of over a century, also contributes to negative impacts on ecosystems and humans.

The atmosphere receives Nr mainly as air emissions of NO_x, NH_3, and N_2O from aquatic and terrestrial ecosystems and of NO_x from combustion of biomass or fossil fuels. Once emitted NO_x can be transformed into a variety of oxidized N species. Ultimately much of the NO_x is converted to HNO_3, which is either converted to an aerosol (e.g., ammonium nitrate) or deposited on land, surface waters, or other surfaces. NH_3 emitted to the atmosphere is either deposited or transformed into an ammonium aerosol (e.g., ammonium bisulfate or ammonium sulfate). Before deposition, NH_4^+ aerosols contribute to fine particulate matter and regional haze concentrations in the atmosphere. Due to the short residence time of NO_x, NH_3 and their reaction products, can only accumulate in the troposphere on a regional scale. Almost all Nr emitted as NO_x and NH_3 is transferred back to Earth's surface within hours to days.

Six major atmospheric effects are associated with increased NO_x and NH_3 emissions, and two with N_2O emissions (Galloway et al., 2003). For NO_x and NH_3 emissions they are: (1) fine PM decreases atmospheric visibility; (2) elevated ozone concentrations enhance the greenhouse potential of the atmosphere; (3) ozone and fine particulate matter have serious impacts on human health (Brunekreef et al. 2005, Brook et al. 2003,Pope 2000a, 2000b Pope et al. 1995, Pope 2009); (4) NH_3 plays an important role in the direct and indirect effects of aerosols on radiative forcing and thus on global climate change (Seinfeld and Pandis 1998, Penner et al. 2001; Lelieveld et al. 2001); (5) ozone deposition can decrease productivity of crops, forests, and natural ecosystems; and (6) atmospheric deposition of NH NH_3, NO_y, and organic forms of Nr can contribute to ecosystem acidification, fertilization, and eutrophication. For N_2O they are: (1) the greenhouse effect in the troposphere and, (2) O_3 depletion in the stratosphere.

2.4.4.2 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's) and to prepare state of the science reviews such as the Criteria

Documents and more recently the Integrated Science Assessments (ISA) (EPA 2004, 2006, 2007). 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 again 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, but the secondary effects of several these gases are also pose health and welfare concerns. If a city had an annual average NO_2 level 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.

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 between current levels of N and S deposition and numerous biologically adverse effects on ecosystems across the U.S. exists (EPA 2008)."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 US, sulfate and nitrate are responsible for about 2/3 and 1/3 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 changed the standards for ozone and PM (see Table 3-23). 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. By the year 2064, states must restore Class I areas to their natural levels of atmospheric clarity (EPA 2004).

Ozone and PM, the two most recalcitrant of the criteria pollutants, cover large spatial scales. These secondary pollutants are not released at the tailpipe; rather they form in the atmosphere. 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 US, ozone episodes often cover several states and involve pollutants emitted in upwind states that do not themselves experience violations (Chen, et al., 2003; Husar, et al., 1977; Logan, 1989; Moy, et al., 1994; Ryan, et al., 1998). The 1990 amendments to the Clean Air Act established, 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, DC to Maine. Progress has been made on regional control of emissions; the NO_x 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, 2007). Experiences with ozone and PM provide a useful demonstration of why it is necessary to develop an integrated approach to management of Nr.

Table 19: Federal primary ambient air quality standards that involve Nr, effective January 2008. Secondary standards are currently identical to the primary standards

Pollutant	Federal Primary Standard (NAAQS)
Ozone (O_3)	
1-hr average	0.12 ppmv
8-hr average	0.08 ppmv
Nitrogen Dioxide (NO_2)	
Annual average	0.053 ppmv (100 µg/m^3)
Particulate Matter, coarse (PM_{10})	
Diameter ≤ 10 µm, 24-hr average	150 µg/m^3
Annual average	50 µg/m$^-$
Particulate Matter, fine ($PM_{2.5}$)	
Diameter ≤ 2.5 µm, 24-hr average	35 µg/m^3
Annual average	15 µg/m^3

2.4.4.3. Atmospheric thresholds for Nr

As shown in Table 19 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 hr, 8 hr, 24 hr, 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 ug/m^3) for the annual arithmetic mean. 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 areas currently in violation of the ozone standard typically have NO_2 concentrations below 0.020 ppm (http://www.epa.gov/air/airtrends/nitrogen.html). The NO_2 concentration required to achieve the current 15 ug/m^3 $PM_{2.5}$ standard is probably also below the 100 ug/m^3 standard for NO_2 because of the role of NO_2 in secondary particulate formation.

The INC is recommending that NO_x emissions be decreased by 2 Tg N/yr, relative to the baseline level in 2002. Emissions decreases implemented since 2002 have already substantially improved (Gégo, et al., 2007) ozone concentrations. 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 are 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.

2.4.5 Reactive Nitrogen and terrestrial ecosystems

2.4.5.1 Impacts of Nr on terrestrial ecosystems

In many terrestrial ecosystems the supply of biologically available Nr is a key factor controlling the nature and diversity of plant life, and vital ecological processes such as plant productivity and the cycling of carbon and soil minerals. Human activities have not only increased the supply but enhanced the global movement of various forms of nitrogen through air and water.

The primary source of excess Nr for most unmanaged terrestrial ecosystems is atmospheric deposition. This additional Nr causes a wide variety of sometimes beneficial effects (increased growth and productivity of forests, natural grasslands, and crops planted in nutrient deficient soils) and also sometimes adverse effects on terrestrial and aquatic ecosystems in many parts of our country. Forests and grasslands exposed to excess Nr can respond in numerous ways. General effects include the following (Cowling, 1989, Cowling et al. 1990, Cowling et al. 2002, Garner et al.1989, Woodman and Cowling1987; Vitousek et al., 1997):

1. Increased productivity of forests soils most of which are Nr-limited throughout the US, Nr deficiency of forest soils has been most fully quantified for pine forests in 14 southeastern states
2. Acidification of forest soils leading to decreased availability of nutrient cations including calcium, magnesium, and potassium and aluminum toxicity, established most clearly in the eastern U.S. and both central and northern Europe
3. Nr saturation of forest soils, presently occurring mainly in high-elevation forests of the eastern U.S. and southeastern Canada
4. Ozone-induced predisposition of forest trees to damage by fungal diseases and insect pests, most clearly established in the case of root disease and bark beetles in the pine forests of southern California
5. Ozone-induced inhibition of photosynthesis in both softwood and hardwood tree species most clearly established in controlled exposure studies in both the U.S. and Europe at ambient concentrations of ozone above 60 ppb. Such concentrations occur frequently throughout the eastern U.S. and southeastern Canada
6. Ozone induced direct injury to foliage, most clearly established in the case of “emergence tip burn” in eastern white pine
7. Acidification induced decrease in frost hardiness of high-elevation conifer forests, most clearly established in the case of red spruce in the northeastern US
8. Acidification induced alteration of beneficial symbiotic relationships in forest soils, especially mycorrhizae, most clearly established in both northern and central Europe

9. Biodiversity losses in natural grasslands and forest areas caused by Nr induced decreases in abundance of Nr-limited tree and grass species and replacement by Nr-loving weed species, most clearly established in both Minnesota and California, and even more vividly in The Netherlands
10. Decreases in visibility and increased haziness of the atmosphere at scenic vistas in national and state parks and wilderness areas
11. More leaching of Nr to aquatic systems via both groundwater and surface runoff – a cascade effect

2.4.5.2 Nr saturation and ecosystem function

There are limits to how much plant growth can be increased by N fertilization. At some point, when the natural N deficiencies in an ecosystem are fully relieved, plant growth becomes limited by availability of other resources such as phosphorus, calcium, or water and the vegetation can no longer respond to further additions of Nr. In theory, when an ecosystem is fully Nr-saturated and its soils, plants, and microbes cannot use or retain any more, all new Nr deposits will be dispersed to streams, groundwater, and the atmosphere. Nr saturation has a number of damaging consequences for the health and functioning of ecosystems. These impacts first became apparent in Europe almost three decades ago when scientists observed significant increases in nitrate concentrations in some lakes and streams and also extensive yellowing and loss of needles in spruce and other conifer forests subjected to heavy Nr deposition. In soils, most notably forest soils because of their natural low pH, as NH_4^+ builds up it is converted to nitrate by bacterial action, a process that releases hydrogen ions and contributes to soil acidification. The buildup of NO_3^- enhances emissions of nitrous oxides from the soil and also encourages leaching of highly water-soluble NO_3^- into streams or groundwater. As negatively charged NO_3^- seeps away, positively charged alkaline minerals such as calcium, magnesium, and potassium are carried along. Thus, soil fertility is decreased by greatly accelerating the loss of calcium and other nutrients that are vital for plant growth. As calcium is depleted and the soil acidified, aluminum ions are mobilized, eventually reaching toxic concentrations that can damage tree roots or kill fish if the aluminum washes into streams (Vitousek et al. 1997).

Forests, grasslands, and wetlands vary substantially in their capacity to retain added nitrogen. Interacting factors that are known to affect this capacity include soil texture, degree of chemical weathering of soil, fire history, rate at which plant material accumulates, and past human land use. However, we still lack a fundamental understanding of how and why N-retention processes vary among ecosystems much less how they have changed and will change with time (Vitousek et al. 1997).

An over-arching impact of excess Nr on unmanaged terrestrial ecosystems is biodiversity loss. In North America, dramatic reductions in biodiversity have been created by fertilization of grasslands in Minnesota and California. In England, N fertilizers applied to experimental grasslands have led to similarly increased dominance by a few N-responsive grasses and loss of many other plant species. In formerly species-rich heathlands across Western Europe, Nr deposition has been blamed for great losses of biodiversity in recent decades, with shallow soils containing few alkaline minerals to buffer acidification (Vitousek et al. 1997; Bobbink et al., 2009).

Losses of biodiversity driven by Nr deposition can in turn affect other ecological processes. Experiments in Minnesota grasslands showed that in ecosystems made species-poor by fertilization, plant productivity was much less stable in the face of a major drought. Even in non-drought years, the normal vagaries of climate produced much more year-to-year variation in the productivity of species-poor grassland plots than in more diverse plots (Vitousek et al. 1997).

2.4.5.3 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, thresholds in general and critical loads specifically for Nr effects on terrestrial ecosystems in the United States 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 United States (e.g., Fenn et al., 2002), it is imperative to understand the extraordinarily wide diversity of types and Nr-sensitivity of various components of terrestrial ecosystems in various parts of the U.S. and the huge differences in purposes and intensity of management and public perceptions of the value of these components to various sectors of American society. Thus, the critical loads appropriate for maintaining species diversity in a natural grasslands in northern Minnesota or a wilderness area in the Mediterranean climate of southern California are likely to be very different from those for 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 states of United States are of the order of 3 to 15 kg N/ha/year [National Atmospheric Deposition Program (NADP, 2008), CASTNET, 2008]. As shown in Figure 15, a three-year run of 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 states of the US. 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 US, 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 (2009) in their excellent review of scientific evidence regarding the impacts of atmospheric nitrogen deposition on plant diversity in terrestrial ecosystems.

2.4.6 Additional 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, as well as 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) to, among other goals, "Provide consistency in development and use of critical loads in the US." 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 (NOx) and Sulfur Oxides (SOx)" expansively reviewed both ecosystem as well as human health effects toward revision of the NAAQS. In policy discussions at this workshop it was asked if critical loads assessments were an effective means of improving ecosystem management and if 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 United States. The Multi-Agency Workshop on Critical Loads mentioned above was cited at this workshop as an agenda-setting effort to resolve some of the science and policy issues that could help advance critical loads approaches in the US.

Finding 17

In this connection, the INC strongly commends EPA for its recently increased willingness to think more holistically – and in more fully integrated ways – about both the policy-relevant science and the practical arts of air quality management aimed at protection of both aquatic and terrestrial ecosystems from adverse effects of Nr. These shifts in both emphasis and approach have included:

1) Increased emphasis in the NAAQS review processes on scientific questions that are as directly relevant as possible to well-defined policy questions of concern to EPA;
2) More frequent discussion about both public-welfare and public-health impacts of mixtures of air pollutants;
3) More frequent discussion about the critical loads concept as an alternative or complement to the more familiar NAAQS Standards;
4) Separation of the preparation and review of documentation for a Secondary (public-welfare-based) NAAQS from the (previously always dominating) Primary (public-health-based) NAAQS review processes;
5) The decision by the Science Advisory Board of EPA to establish this special Integrated Nitrogen Committee (INC); and
6) The unprecedented decision to undertake an integrated (simultaneous) review/ /of the Secondary NAAQS for two Criteria Pollutants at the same time [Oxides of Nitrogen (NO_x) and Oxides of Sulfur (SO_x)].

Especially notable evidence for EPA's "increased willingness to think more holistically – and in more fully integrated ways" is the following statement of Conclusion in the Executive Summary of the December 2008 Integrated Science Assessment for Oxides of Nitrogen and Sulfur (EPA, 2008):

> The main effects of N and S pollution assessed in the ISA are acidification, N enrichment, and Hg methylation. Acidification of ecosystems is driven primarily by deposition resulting from SO_x, NO_x, and NH_x pollution. Acidification from the deposition resulting from current emission levels causes a cascade of effects that harm susceptible aquatic and terrestrial ecosystems, including slower growth and injury to forests and localized extinction of fishes and other aquatic species. In addition to acidification, atmospheric deposition of reactive N resulting from current NO_x and NH_x emissions along with other non-atmospheric sources (e.g., fertilizers and wastewater), causes a suite of ecological changes within sensitive ecosystems. These include increased primary productivity in most N-limited ecosystems, biodiversity losses, changes in C cycling, and eutrophication and harmful algal blooms in freshwater, estuarine, and ocean ecosystems.

In addition, 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 17. *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*.

The European Union has undertaken broad measures, based on the critical loads concept, to manage Nr. Tables 20, 21, and 22 summarize several different environmental impacts, currently used indicators, and whether there are current limit values set by the United Nations Economic Commission for Europe (UNECE) or European Union (EU). These tables identify the main links to the cascade of reactive nitrogen in the environment, the relevance and link to Nr of the effect/pollutant, and existing agreements in which the effect is currently addressed. In addition, some impacts are more relevant than others in relation to societal importance and the connection to the nitrogen cascade. The categorization on a scale of 1 (highest relevance) to 5 (unimportant) provides a first level prioritization for future mitigation activity. The last column summarizes existing links to international regulations and conventions.

Where there is a limit and the relevance for the nitrogen cascade is high, then this might be the limiting factor for Nr production and its associated transfers to the environment. Some limits might be more relevant in specific areas and less relevant in others. For example NO_2 concentrations relevant for human health are limited to 40 ppb in urban areas, limiting industry and traffic, but are probably not an issue in remote areas with low population densities. In these areas, however, loss of biodiversity might limit nitrogen deposition and therewith the sources in the region. The only way to determine the extent that critical thresholds are limiting is by overlaying them for different regions and determining by monitoring data or by model exercises where and which sources contribute to exceeding the critical threshold, and then identifying the best methods for putting caps on relevant sources. A pre-classification of regions might be useful, e.g. urban regions, remote regions, marine areas, etc.

One aspect of this global view of nitrogen impacts and metrics that is evident is the mix of "classical"- and "service"-based categories, consistent with the need for an integrated approach to the management of nitrogen.

Table 20: Summary of the effects of excess Nr on human health in relation to metrics, current international regulations and conventions, and the link to the nitrogen cascade

	Metrics	Regulated?	Link to Nr cascade	Relevance*	Regulatory or political convention
Respiratory disease in people caused by exposure to high concentrations of:					
Ozone	Sum of ozone over 35 ppb	Y	NO_x emissions	3	Convention on Long-range Transboundary Air Pollution Clean Air for Europe
other photochemical oxidants	Org. NO_3, PAN conc (atm)	N	NO_x emissions	5	indirectly Convention on Long-range Transboundary Air Pollution et al.
fine particulate aerosol	PM_{10}, $PM_{2.5}$ conc (atm)	Y	NO_x, NH_3 em	1	Convention on Long-range Transboundary Air Pollution Clean Air for Europe
direct toxicity of nitrite NO_2^-	NO_2^- conc	Y	NO_x	2	World Health Organization Convention on Long-range Transboundary Air Pollution Clean Air for Europe
Nitrate contamination of drinking water	NO_3^- conc (aq.)	Y	NO_3^- leaching	2	EU Essential Facilities Doctrine
Depletion of stratospheric ozone	NO_x, N_2O conc/flux (atm)	N	NO_x, N_2O	3	Montreal Protocol
Increase allergenic pollen production, and several parasitic and infectious human		N		5	None
Blooms of toxic algae and decreased swimability of in-shore water bodies	Chlorophyll a NO_3^- (&P) conc (aq)	N	Runoff, Nr deposition	1	Convention for the Protection of the Marine Environment of the North-East Atlantic Helsinki Commission Barcelona Convention

*Relevance and link to nitrogen incorporates societal priority and N contribution: 1) highest relevance, 2) high relevance, 3) significant relevance, 4) some relevance, 5) unimportant.

Table 21: Summary of the effects of excess nitrogen on ecosystems related to to currently used metrics, the existence of European regulatory values, and the link to the nitrogen cascade

	Metrics	Regulated?	Link to Nr cascade	Relevance*	Regulatory or political convention
Ozone damage to crops, forests, and natural ecosystems	AFstY (O_3 flux), AOT40	Y	NO_x	2	Convention on Long-range Transboundary Air Pollution Clean Air for Europe
Acidification effects on terrestrial ecosystems, ground waters, and aquatic ecosystems	Critical loads	Y	Nr deposition	2	Convention on Long-range Transboundary Air Pollution Clean Air for Europe WFD
Eutrophication of freshwaters, lakes (incl. biodiversity)	Biological Oxygen Demand, NO_3^- conc (aq) Critical loads	Y N	Runoff, Nr deposition	3	Water Framework Directive
Eutrophication of coastal ecosystems inducing hypoxia (incl. biodiversity)	BOD, NO_3^- conc (aq) Critical loads	BOD, NO_3^- conc (aq) Critical load	Runoff, Nr deposition	1	Convention for the Protection of the Marine Environment of the North-East Atlantic Helsinki Commission Barcelona Convention
Nitrogen saturation of soils (incl. effects on GHG balance)	Critical loads	Y	Nr deposition	1	Convention on Long-range Transboundary Air Pollution Clean Air for Europe
Biodiversity impacts on terrestrial ecosystems (incl. pests and diseases)	Critical loads, critical levels (NH_3, NOx)	Y	Nr deposition	1	Convention on Long-range Transboundary Air Pollution Clean Air for Europe, Convention on Biological Diversity

*Relevance and link to nitrogen incorporates societal priority and N contribution: 1) highest relevance, 2) high relevance, 3) significant relevance, 4) some relevance, 5) unimportant.

Table 22: Summary of the effects of excess N on other societal values in relation to metrics and regulatory values in current international regulations and conventions and the link to the nitrogen cascade.

	Metrics	Regulated?	Link to Nr cascade	Relevance*	Regulatory or political convention
Odor problems associated with animal agriculture	Acidity in prec., prec./T O_3, PM	Y	NO_x, NH_3	3	Convention on Long-range Transboundary Air Pollution
Effects on monuments and engineering materials	$PM_{2.5}$ conc (atm)	N	NO_x, NH_3	4	
Global climate warming induced by excess nitrogen	N_2O, conc/flux (atm)	N	NO_x, NH_3	1	United Nations Framework Convention on Climate Change
Regional climate cooling induced by aerosol)	$PM_{2.5}$ conc (atm)	N	NO_x, NH_3	1	United Nations Framework Convention on Climate Change

*Relevance and link to nitrogen incorporates societal priority and N contribution: 1) highest relevance, 2) high relevance, 3) significant relevance, 4) some relevance, 5) unimportant.

2.4.7 Tradeoffs of Nr impacts

Because N is such an abundant and widespread element, and Nr such a critical component of the Earth's biosphere, associated impacts are many and pervasive. In many cases 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 aimed at limiting one form of Nr in air or water that can 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, which may, in turn lead to the production of more reactive nitrogen. 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 why a life cycle approach is necessary in evaluating any remediation or treatment scheme.

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.

2.4.7.1 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 18

Current EPA policy (EPA 2007e) discourages states from controlling ammonia emissions as part of their plan for reducing $PM_{2.5}$ concentrations. Ammonia is a substantial component of $PM_{2.5}$ in most polluted areas of the U.S. 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 US. Lower NH_3 emissions will lower $PM_{2.5}$ concentrations. Such reductions in $PM_{2.5}$ concentrations have been linked to reductions in morbidity and mortality.

Recommendation 18. *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.*

2.4.7.2 Unintended impacts of lower application rates of nitrogen for crop production

Crop production and environmental quality are potentially lost or gained at the expense of each other. Although leakage of N from crop production systems cannot be eliminated, N losses can be minimized substantially. One mechanism of decreasing leakage is to apply less N fertilizer to croplands. For example Hu et al. (2007), using the SWAT model, predict that decreasing N fertilizer application rates used in the late 1990s by 10 to 50% in the upper Embrarras River watershed in east central Illinois, would decrease NO_3^- output to the river by 10 to 43%. This simple “solution” can cause problems for crop production as yields and crop quality (protein content) may decrease, causing economic loss to the farmer, decreased food quality for the consumer, and, at a global scale, a reduction in food security.

Cropping systems managed in a consistent manner over time reach a functional equilibrium between nitrogen inputs and outputs. Because crop yields are closely linked to the quantity of N accumulation in above ground biomass at maturity (Cassman et al., 2002), there would be a proportional decrease in crop yields in response to a decrease in the amount of N fertilizer application. The magnitude of this yield reduction would depend on the magnitude of decrease in the rate of applied N and the efficiency of N uptake from the applied N, as well as interrelationships with other nutrients’ availability. Hence, yield reductions can be mitigated, or even eliminated, if methods and fertilizer formulations used in fertilizer-N application increased the efficiency of nitrogen uptake to offset the reduction in the amount of applied nitrogen. It is also important to note that reduced or insufficient N rates for crop production risk impairment of long-term soil productivity. Jaynes and Karlen (2005) reported that N rates below the agronomic and economic optimum could degrade the soil resource and decrease soil organic matter over time. Thus care must be exercised in any N rate adjustments to protect soil productivity and to support soil resource sustainability.

An example of the effect of decreasing N fertilizer input to cereal crop production on crop production and crop quality as a result of national efforts to decrease Nr losses to the environment from crop production is the situation in Denmark. In response to the European Union Nitrate Directive synthetic fertilizer nitrogen use in Denmark decreased (Figure 19) from approximately 0.4 Tg N in 1991 to 0.2 Tg in 2002. Animal manure N application decreased from 0.25 Tg to approximately 0.24 Tg N during this time period. Nevertheless, although N input into Danish cereal crop production decreased, cereal crop yield remained relatively constant, as shown in Figure 20.

Figure 19: Synthetic fertilizer and livestock manure N used as fertilizer in Denmark 1985-2003 (IFA 2004).

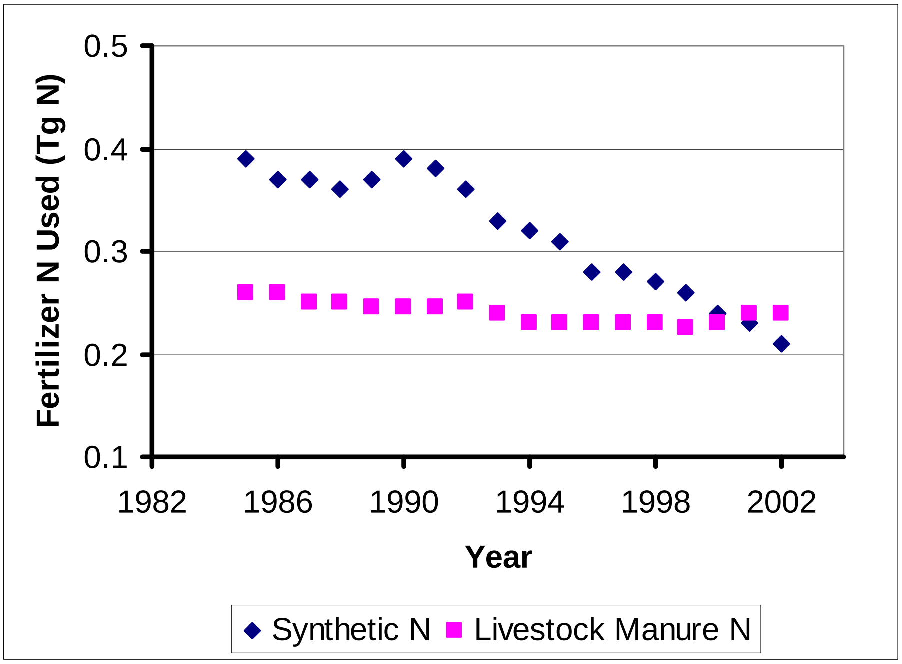
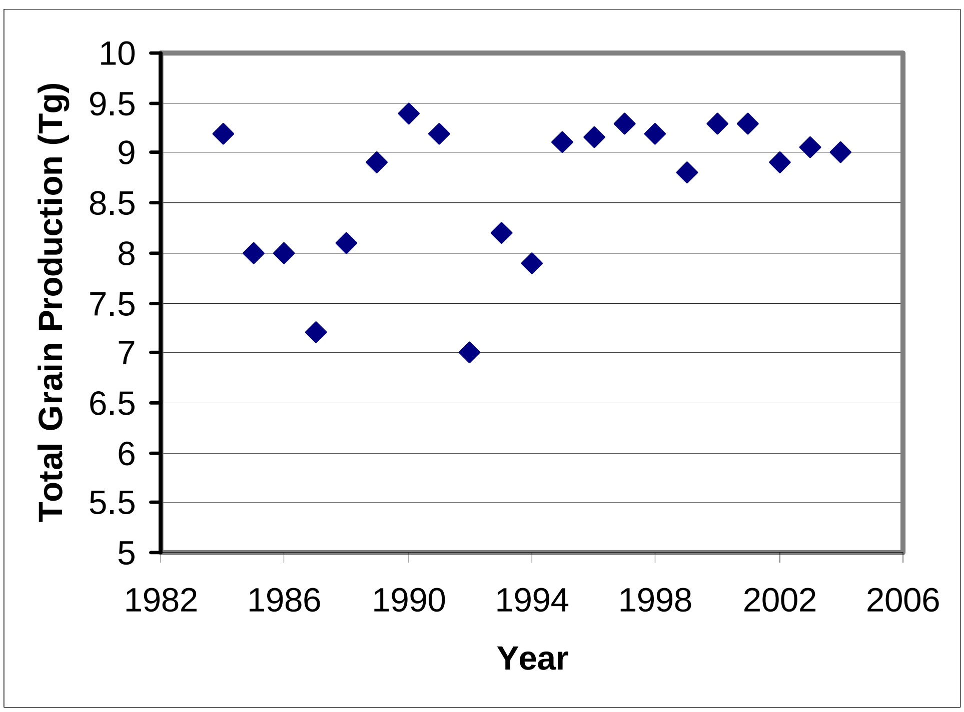
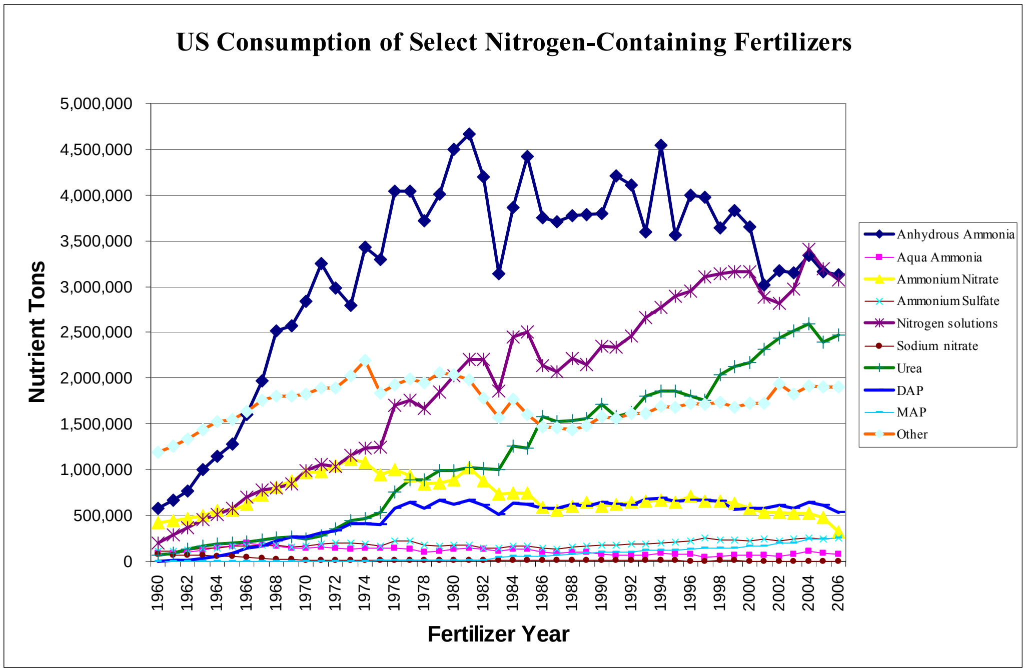


Figure 20: Total cereal grain production in Denmark 1985-2004 (FAOSTAT, 2007)



If the methods used to apply N were to be modified to improve its overall efficiency, then it is possible to reduce N fertilizer inputs and maintain, or even increase crop yields depending on the magnitude of the improvement in NUE (see section 2.2). Although U.S. fertilizer application has not declined over time, it has leveled off in recent years, as shown in Figure 22. Even so, yields, at least for corn grain, have continued to increase, a trend that has been in evidence since the mid 1970s, as shown in Figure 7, section 2.2.3.1.

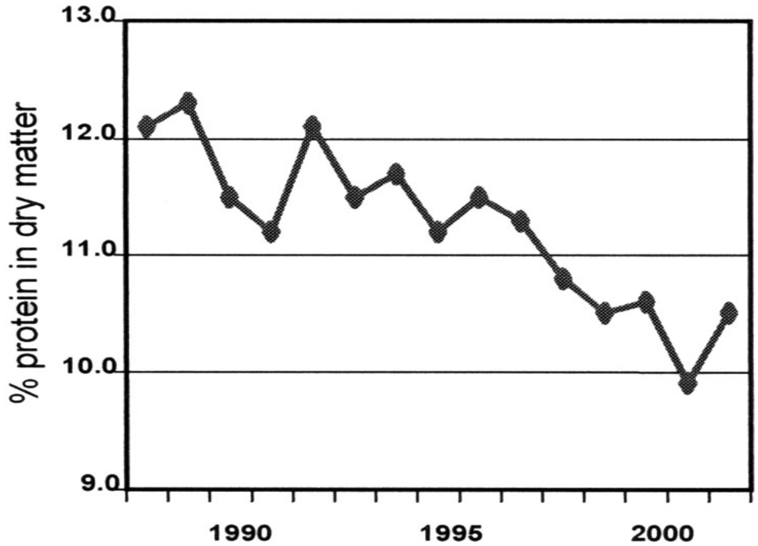
Figure 21: Consumption of N-containing Fertilizers in the U.S. (USDA)



The effect of decreasing N fertilizer input can be estimated based on the current level of PFP for applied N (average U.S. grain yield divided by the average N fertilizer application rate). Making the assumption that without a concerted effort to improve N fertilizer application methods, yields will decrease at 90% of the current PFP for N fertilizer (Cassman et al., 2003; Dobermann and Cassman, 2004). With a 10 to 50% decrease in N fertilizer application the calibrated SWAT model predicted a 6 to 38% reduction in maize yield in the upper Embrarras River watershed (Hu et al. 2007).

A negative impact of decreased Nr input into cereal crop production is the potential for a decline in grain quality as shown in Figure 22 as a decrease in grain protein content in Denmark. Grain protein content in wheat is critical for determining its quality for bread, for example in the U.S. a grain protein content of 12% is considered the threshold for good quality bread wheat, and N fertilizer application rate has a large influence on determining this trait (Cassman et al., 1992). As can be seen, grain protein content has declined from 12 to 10% in Denmark over the same period of lower fertilizer application rates.

Figure 22: Protein content of cereal grain in Denmark (IFA, 2004).



Such trends raise several questions if declines continue or are found to be widespread. What is the cost to the farmer and in terms of human nutrition and end-use value? Do these costs offset the environmental benefits created by decreasing N flows from crop production areas? And, what would be the regional and global impact if similar reductions in nitrogen fertilizer inputs to agriculture were put in place in developed countries that represent the largest source of grain exports to international markets?

One caveat to this decline in cereal protein content is the fact that grain protein content in maize hybrids used in the USA declined at a linear rate between 1934 and 1991 (Duvick et al. 2005). Over the time period 1970-1992 hybrid maize grain protein concentration in central Iowa declined from ~9.7 to 9% (Duvic and Cassman, 1999). Although we do not know the genetic makeup of the cereal grain shown in Figure 22, it does not seem likely that genetics alone could account for the large difference in grain protein content between 1990 and 2001.

Finally, if protein yields are significantly reduced as a result of lower N fertilization rates, more land may need to be brought into production. Because nearly all prime agricultural land is already used for crop production, expansion of crop area will most likely occur on more marginal land, such as the land currently in the CRP. Such conversion would result in additional N losses from these acres due to relatively low N fertilizer efficiency that typically occurs on marginal land that has multiple soil constraints to crop growth and yield.

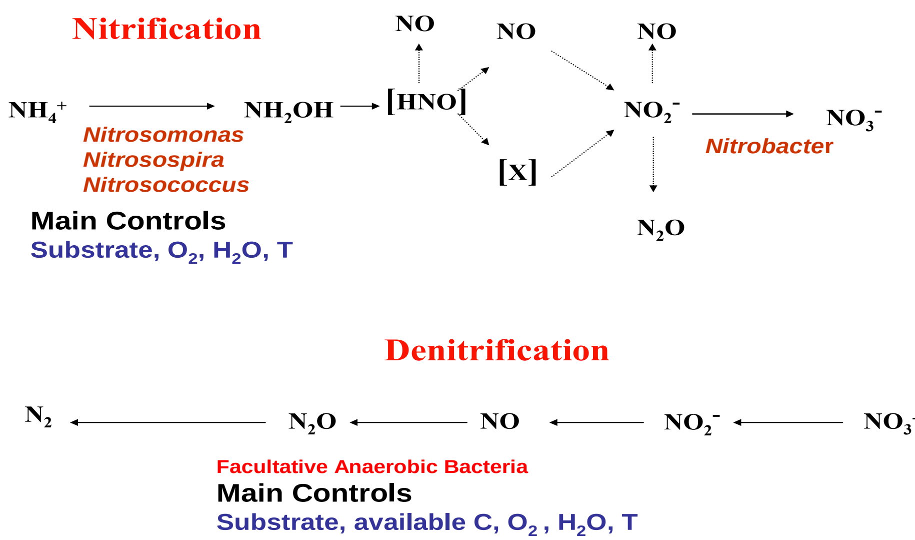
2.4.7.3 Unintended impacts: swapping N between environmental systems

Nitrous oxide is produced in “natural” and agricultural soils, and all aquatic systems almost exclusively as a result of microbial processes, nitrification and denitrification. As NH_4^+ ion is the initial mineral N product formed during organic matter mineralization and most 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 23). Most commonly nitrification is a chemolithotropic process which consists of the conversion of NH_3 to nitrite, which is then converted to NO_3^- by a second group of bacteria. The ammonia 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,* which result in the formation of nitrite. AOB are found in most aerobic environments where NH_3 is available through the mineralization of organic matter or 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 23. 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).

Figure 23: Diagram of the nitrification and denitrification processes (from 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, it is important to recognize the potential for unintended consequences to occur as a result of strategies aimed at limiting one form of Nr in air or water that can 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 because NO continues to be reactive in the atmosphere and is eventually redeposited in aquatic or terrestrial systems and N_2O is a greenhouse gas that has an atmospheric life time of approximately 120 years and a radiative forcing of approximately 300 times that of CO_2 on a hundred year time frame (IPCC 2001), and is a major source of NO in the stratosphere which 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, for which 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 the ~301 million humans in the U.S. results in the processing of ~ 2 Tg of N annually (~18.4 g N/ person/d), much of which flows through sewage treatment facilities and ultimately leads to the production of 0.06 – 0.1 Tg of N_2O-N /yr in aquatic systems or soils to which sewage sludge is applied.

2.4.7.4 Tradeoffs among C and N-driven impacts

Many of the impacts on the environment to which reactive N contributes are also impacted by other chemical species, notably carbon; there are several points of tangency between the global C and N cycles, as depicted in Figure 24: 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 impact-free. Such "trade-offs" may involve a single impact, e.g. global climate change for which both carbonaceous gases and N_2O contribute, but may also involve trade-offs between impacts that are not easily compared. Figure 25 shows the latter case in the form of climate change impacts (for which C is a principal contributor) versus eutrophication impacts (for 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. One hundred percent 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 25 represents net C sequestration. It is difficult to make direct comparisons across disparate impact categories, however Figure 25 suggests that, in choosing among alternatives, policies that aim to minimize both sets of impacts would be preferred.

Figure 24: Combined carbon and nitrogen global cycles (Miller et al. 2007)

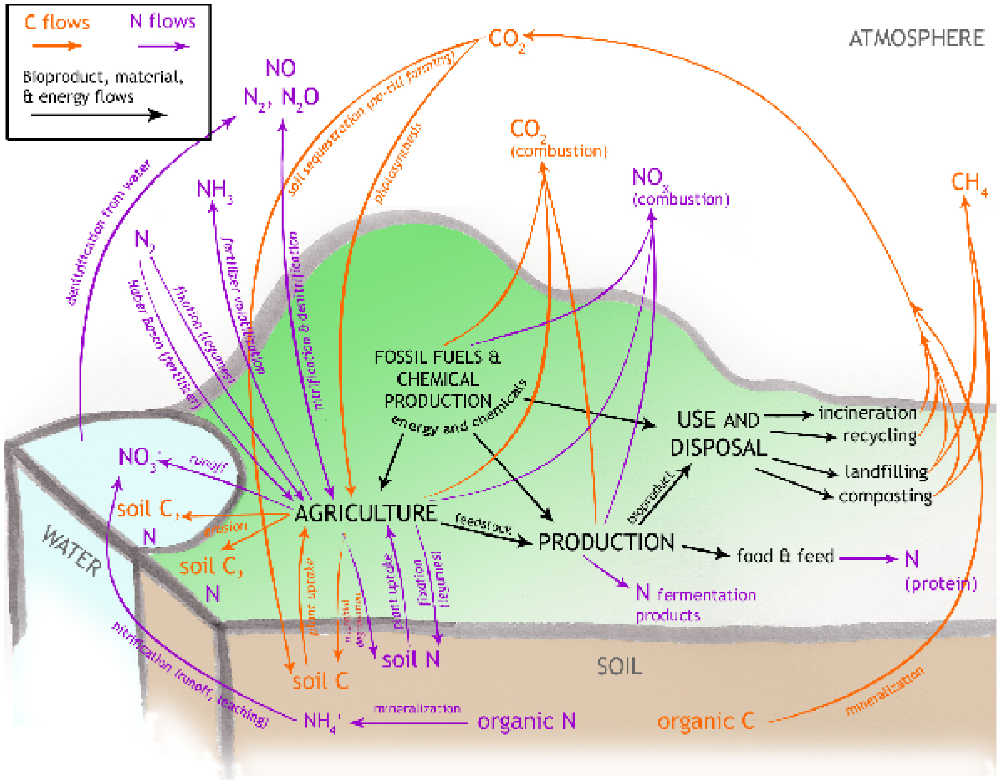
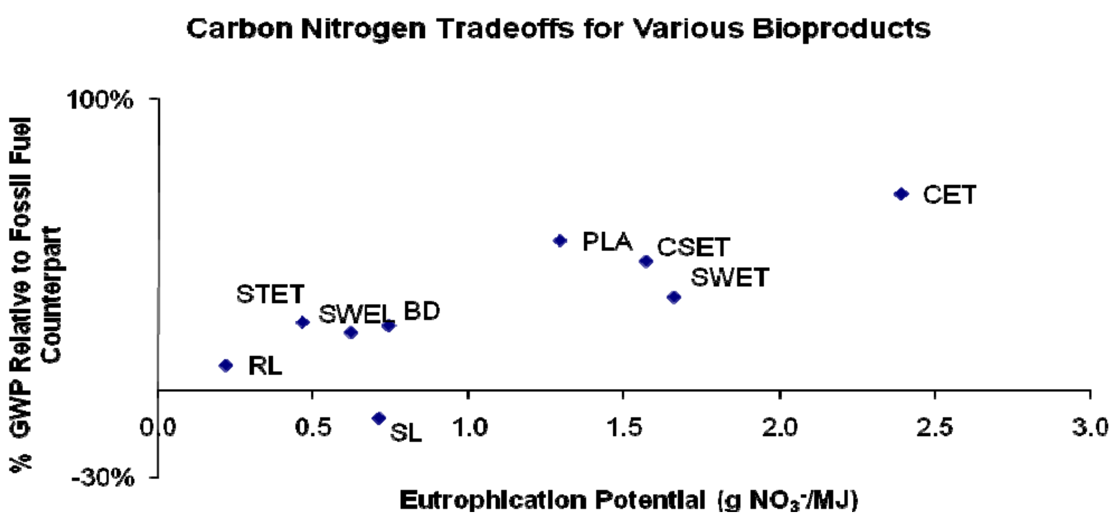


Figure 25: Comparisons Between Climate Change and Eutrophication Impact Categories for Various Bioproducts (updated from Miller et al. 2007)



(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).

Finding 19

The committee notes that the effective management of Nr in the environment must recognize the existence of tradeoffs across impact categories involving the cycling of other elements, particularly C.

Recommendation 19. *The committee recommends that the integrated strategies for Nr management outlined in this report be developed in cognizance of these interrelations and tradeoffs.*

2.4.8 Interactions of the N cascade and climate

Weather and climate vary substantially on many time scales including the interannual. Long-term (decadal or more) change in climate as have been predicted by IPCC (2007) 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 this report to fully address how cycles of C and N interact (see Figure 24 for a general treatment of the intersection points of C and N cycles), there are several ways in which climate impacts the biogeochemical cycle Nr and vice versa, e.g., Holland et al., 1997; Hungate et al., 2003; Hungate et al., 2004; Levy et al., 2008; Sokolov et al., 2008; Sutton et al., 2007; Thornton et al., 2007; Yienger and Levy, 1995 . These are highly interactive and nonlinear systems, but they include:

- 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 accompanying text box 4 The impact of climate change on agricultural discharge of Nr for Eastern Iowa).
- Increasing temperature can both increase and decrease atmospheric loading of particulate matter (PM).
- 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 US 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 NOx 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 20

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 20: *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 INC 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 or 2100.*
2. *Down-scale statistics or nest regional climate models within each of these global scenarios to generate meteorological and chemical fields (e.g., T, RH, winds, precipitation, CO_2) for a few years around 2050 and 2100.*
3. *Run several independent biogeochemical Nr models (Earth System models that include air/water/land) for N 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.*

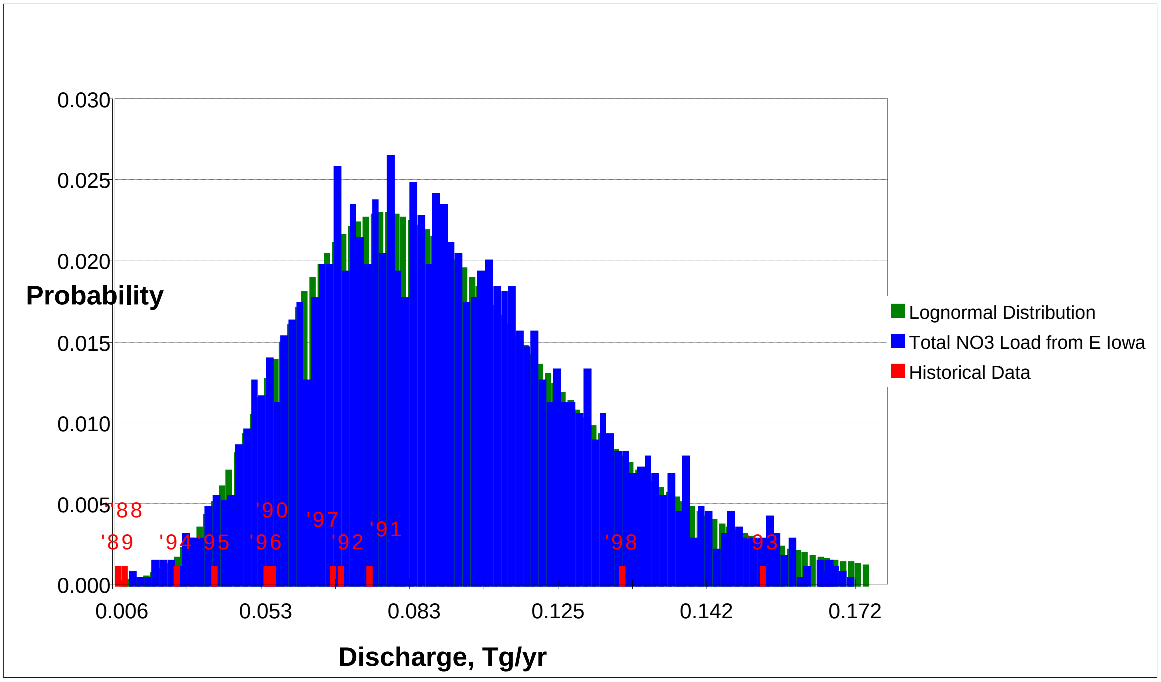
Text Box 4: The impact of climate change on agricultural discharge of reactive nitrogen

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 2007).

The general impact of climate change on the discharge of reactive nitrogen from agroecosystems can be discerned from the information in Figure 26. This figure shows a probability distribution for nitrate discharged from the watersheds of eastern Iowa (approximately 50,000 km^2), 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 26 (in green) is a standard log-normal distribution, which the simulation most closely fits, and independently measured nitrate runoff data (an output of the system) over the same time period, as reported by Powers (2007). The inclusion of these data within the simulation distribution (which is based on fertilizer inputs to the system) provides confidence in the descriptive capability of the model.

Figure 26: 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.

Figure 26 shows that the interannual variation in nitrate discharged is nearly 30-fold during the eleven year observation period. While the impact of climate change on such a system cannot be predicted for a given year, Figure 26 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 26 to the right, resulting in generally higher discharges of nitrate (see for example Vanni et al. 2001; the data point for 1993 in Figure 26 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. 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.

Chapter 3: Integrated risk reduction strategies for reactive nitrogen

3.1 Introduction

Chapter 2 presented the environmental impacts and metrics associated with the emission of the various forms of Nr and reviewed ways of organizing these into impact "categories." As noted, Nr has many impacts on the environment, impacts that are interrelated through the nitrogen cascade.

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. 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.

3.2 Control strategies for Nr

There are several ways in which the release and control of Nr in the environment can be approached. In general these can be classified as follows:

1. Improved practices and conservation—in which the flux of Nr that creates an impact is lowered through better management practices, including those that preserve or enhance Nr controlling ecosystem services (e.g. on-field agricultural practices, controlled combustion conditions, ecosystem function preservation and management)
2. Product substitution—in which a product is developed or promoted which has a lower dependency on or releases less Nr (e.g. N-bearing wastes instead of corn grain as a feedstock for biofuels, development of alternative power sources such as wind and solar)
3. Transformation—in which one form of nitrogen is converted to another form (e.g. nitrification of wastewater, denitrification in engineered or natural systems),
4. Source limitation—in which the amount of Nr introduced into the environment is lowered through preventive measures (e.g. controls on NO_x generation)
5. Removal—in which Nr is sequestered from impacting a particular resource (e.g. ion exchange)
6. Improved use or reuse efficiency—in which the efficiency of production that is dependent on Nr is improved (e.g. increased grain yields for lower Nr applied), or Nr wasted from one source is reused in another (e.g. algal farming).

Effective management of Nr requires combinations of these approaches; none is a perfect alternative for controlling Nr in the environment. Table 23 provides a summary of the pros and cons of each of these approaches.

Table 23: Advantages and limitations of various approaches to Nr control

Control strategy	Advantages	Limitations
Improved practices, conservation	Lessens one or more impacts; utilization of existing ecosystem services	Education cost; availability and cost of preserved lands
Product substitution	Lessens the need for Nr, allows for more targeted uses of Nr	Questions of acceptability, technological issues
Transformation	Reduces one or more impacts to which Nr contributes, for denitrification closes the nitrogen cycle; utilizes natural biogeochemical processes that may be available ecosystem services	May contribute to other impacts; human presence has modified and diminished ecosystem service values
Source limitation	Reduce one or more impacts to which Nr contributes	Decreased crop yields, in some cases few viable alternatives yet developed
Removal	Reduces one or more impacts to which Nr contributes; Natural land features/processes and ecosystem services may be used	Residuals containing Nr must still be managed effectively; availability, location and cost of land for natural or enhanced Nr removal
Improved efficiency	Reduces the need for Nr	Research and education costs

3.3 Management of reactive nitrogen in the environment

Four types of management strategies for the control of Nr, and other pollutants, in the environment have evolved over the past 40 years:

1. Command-and-Control—in which an entity's "right to pollute" is recognized through a series of permitted limitations on emissions, violations of which may result in penalties being assessed.
2. Government-based programs for effecting a policy, such as directed taxes, price supports for a given commodity, subsidies to bring about a particular end, and grants for capital expansion or improvement.
3. Market-based instruments for pollution control in which market trading schemes are used to bring about a desired policy end, often at reduced overall cost.
4. Voluntary programs in which desired ends are achieved using private or government-initiated agreements or through outreach and education.

3.3.1 Command-and-control[8]

Policy makers have traditionally used command-and-control strategies requiring individuals and firms to meet mandatory guidelines. Such an approach evolved as the country was gearing up to meet the requirements first established nationally through the CWA and CAA enabling legislation in the 1970's. Because U.S. capabilities to monitor contaminant concentrations and predict environmental impacts was, generally, rudimentary, early emphasis was placed on "technology-based" approaches for managing emissions, resulting in the promulgation of "best practicable technology" controls, and eventually "best available technology" controls, the idea being that mandating some level of control, even with uncertain improvements on impacts, would be better, and less arbitrary, than other approaches of the time.

Nevertheless, both the CWA and the CAA had more specific goals that were aimed at protecting human health, public welfare, and ecosystem health. For example the CAA required states to develop implementation plans (SIPs), the approval of which depended on their ability, once implemented, to meet ambient clean air standards. Likewise, the CWA required greater controls to be implemented for certain water bodies for which technology-based limits alone were insufficient to meet standards (this became the TMDL program).

Over time, and as our abilities to monitor, predict, and understand impacts improved, it became possible, or at least plausible, to tailor emission levels on a source-by-source basis, allowing the firm in question to decide its own technological approach. Thus permits, which place strict limits on the amount of pollution a firm is allowed to discharge over a specified period of time, have become the main method for managing the majority of point source contaminants, including the various forms of Nr, in the environment.

3.3.2 Market based instruments for pollution control [9]

A fundamental shift in environmental management philosophy was initiated with the 1990 Clean Air Act Amendments, which combined regulatory requirements with market flexibility allowing lower compliance costs through tradable permits. Most market-based policy instruments operate on the principle that if the regulatory framework or some other factor sufficiently alters the relative value of available decision choices for an individual or firm, subsequent decisions they make will be in alignment with the policy maker's objective.

As an example, if a government wants to limit pollution in a river where a number of polluters discharge, it need not adopt a uniform command and control limit on each firm. Instead, a regulatory cap on the total permissible pollution can be established at a lower pollution level and permits to pollute that sum to that overall cap can be issued to all firms. Those firms having low

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[8] Based on **Models in Environmental Regulatory Decision-Making**, National Research Council, 2007.

[9] Based on Canchi, D., P. Bala and O. Doering, 3/3/2006, Market Based Policy Instruments in Natural Resource Conservation, Report for the Resource Economics and Social Sciences Division, NRCS, USDA, Washington D.C., pp. 4-9.

pollution control costs will have incentive to control more pollution than their permit allowance and thus have permits they no longer need that can be sold to firms with high costs of pollution control. Because the supply of permits (and the overall cap on the pollutant) is fixed, the regulatory goal is achieved. The tradable permit thus brings about the desired reduction in pollution level at lower cost than if the firms having high costs of pollution control were required to control their full share and low cost of control firms were limited to their share of control. . Tradable permits also encourage cost effective pollution control investment by giving each firm a clear economic signal to invest in new technology to reduce pollution at a level that corresponds to the market value of the permit."

As with control strategies for Nr, there is no one universal market-based strategy that is applicable to every policy maker's objective. For example, the nature of incentives available to and effective with producers involved in over-fishing is different from landowners providing environmental amenities. In the former case, the objective is to restrict the intensity of fishing. In the latter case the objective is to encourage private landowners to provide environmental goods and services at the lowest cost possible.

Evolution of new market-based strategies is a continuous process. Most strategies have been customized over time to meet local needs. One can group such market based approaches under the following conceptual headings:

1. Water Quality Tradable Permits: Every polluting entity is allowed to discharge pollutants up to a certain pre-determined limit, defined in concordance with the terms of the CWA. The entities discharging less that their allocated limit generate credits. Under this strategy, credits can be traded with other polluting entities that have exceeded their allocated limit.

2. Auction Based Contracting: Environmental or conservation contracts are auctioned where individual landowners place their bids to provide such goods or services from their land. Two factors jointly determine the selection of the bids; the amount of the bid and the expected value of the environmental or conservation benefit resulting from accepting the bid.

3. Individual Transferable Quotas: An individual transferable quota (ITQ) is an allocation privilege to extract a specified quantity of a resource among a selected number of quota holders. The distinctive feature of the ITQ is that the privilege is transferable or leasable. An ITQ may be a right to produce under favorable circumstances, such as a tobacco quota when tobacco production would normally be limited.

4. Risk Indemnification for Specified Behavior: An example of this is crop insurance designed to protect farmers from uncertainty in the adoption of best management practices that provide a public good but are inherently riskier.

5. Easements: Conservation Easements or conservation servitudes refer to the case where a land owner enters into a legally binding agreement to surrender certain property rights for a specified period of time either voluntarily or for compensation.

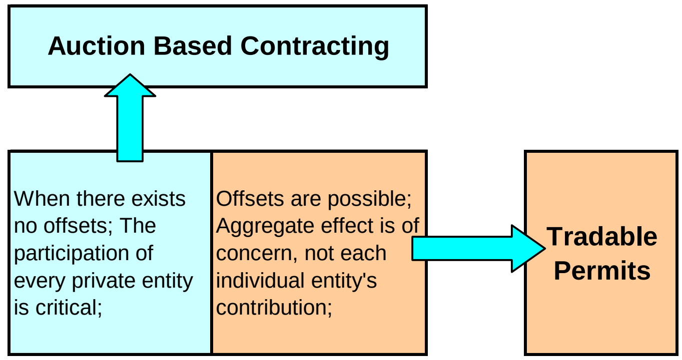
Such arrangements usually provide public goods relative to the environment or conservation.

The policy maker's objective, the local conditions, and several other factors determine the suitability of a particular market based strategy. For example, a tradable permit strategy is well suited where offsets are possible. In the case of water quality it is not uncommon to find a spectrum of polluters at different levels of contribution. A policy framework that facilitates the emergence of multiple options for polluters to buy credits from more efficient controllers of discharge or to invest in new equipment to achieve further reductions is likely to accomplish the desired level of water quality at the least possible cost to the economy. Table 24 illustrates the potential effective application of a number of market based approaches in specific situations. Accompanying this chapter are two examples of the application of market-based approaches for the design of water quality trading schemes for Nr in watersheds (text box 5: Water Quality Trading to Meet the Long Island Sound Wasteload Allocation in Connecticut and text box 6: Water Quality Trading in the Illinois River Basin).

Table 24: Summary of market-based instruments for pollution control

Auction Based Contracting		Individual Transferable Quotas		Insurance for the Adoption of BMPs		Easements		
When there exists no offsets; The participation of every private entity is critical;	Offsets are possible; Aggregate effect is of concern, not each individual entity's contribution;	When the depletion is of concern;	When the discharge is of concern;	Homogenous polluters; Offsets not feasible; Excessive pollution is primarily to mitigate uncertain profits; Modest short-term objective;	Not homogenous polluters; Offsets are possible; Pollution is an absolute consequence of the production process;	Unidirectional; When offsets are not possible; One entity retiring more property rights cannot trade with the other retiring less property rights.	Bidirectional; Offsets are possible; Requires specific action on the part of the participant to accomplish the objective;	**Tradable Permits**
		Aggregate depletion is of concern;	When there exist no offsets; The participation of every private entity is critical;	Tied to a production process; When risk averseness of the entity can be used to motivate participation;	Not tied to any production process; Suited for motivating participants to engage in secondary activities;	Auction based contracting can be seen as a refined and improved cost-efficient alternative to easements;	Designing of auction based contracting requires considerable professional expertise;	**Auction Based Contracting**
				Discharge of effluents is of concern;	Depletion of a resource is of concern;	Retirement of rights is of concern;	Acquisition of rights is of concern;	**Individual Transferable Quotas**
						No uncertainty; No action required on the part of the participant;	Tied to a production process:	**Insurance for the Adoption of BMPs**

Table 24 shows pair-wise comparison between different market-based strategies. The objective and the incentive structure of the participants determine the suitability of one market based strategy over another. Each pair of cells briefly lists the most relevant set of conditions for which the respective strategy may be optimal (left cell points to strategy at the top of the column and right cell points to the strategy at the end of the row). Consider the two strategies (illustrated on the right): Auction Based Contracting and Tradable Permits. If the participation of every private entity is essential, then Auction Based Contracting works best. For example, if the objective is to preserve a large tract of privately owned contiguous land. This requires the participation of every private land owner to set aside a portion of their land. An auction designed to reveal the individual's land owner's reserve price for participation leads to the most efficient solution. Compared to this, if the objective is an overall reduction of a pollutant regardless of the individual private entity's contribution to the abatement, Tradable Permit strategy with a cap is a more appropriate strategy.



<u>Text Box 5:Water Quality Trading to Meet the Long Island Sound and Wasteload Allocation in Connecticut</u>

Pollutant trading is increasingly being promoted as a cost-effective means for attaining water quality standards. Connecticut and New York have been working with the EPA Long Island Sound Study (LISS) for more than 20 years to address low oxygen conditions (hypoxia) in Long Island Sound that have been linked to excessive loadings of nitrogen. A Total Maximum Daily Load (TMDL) for nitrogen, drafted by the two states and approved by the EPA in 2001, set a 58.5% nitrogen reduction target in 2014 from point and nonpoint source/stormwater sources. Connecticut has initiated a point source trading program for 79 municipal sewage treatment plants (STPs) to facilitate implementation of the TMDL wasteload allocation (WLA) and is investigating the potential for incorporating nonpoint source/stormwater into the existing Nitrogen Credit Exchange (NCE).

Several prerequisite conditions essential to the success of the current point source trading program have been met. Briefly, 1) all the STPs contribute to the same water quality problem; 2) the technology to remove N and meet the targets exists; 3) there are compelling member benefits to participate, especially cost savings; 4) sources can be easily monitored and tracked by end-of-pipe monitoring; 5) credit cost calculations are based on established and agreed upon protocols founded in state legislation; 6) sources of N are diverse and create viable supply and demand conditions while reducing overall cost, with close control by a Nitrogen Credit Advisory Board; and 7) transaction costs are low relative to credit prices. In operation since 2002, the NCE has proven to be a viable and effective mechanism for meeting the nitrogen WLA.

The economic record of the NCE demonstrates the vigor of trading over the first five years of completed trades from 2002-2006 (21). In sum, more than 10 million credits have been traded on the NCE representing more than $22 million in economic activity.

The use of geographically-based trading ratios is instrumental to the relative cost of meeting N reduction limits at the 79 treatment plants, which are scattered throughout the state (Figure 27). Because N is reactive as it travels down rivers into the Sound, and the Sound's currents further affect relative impacts as they transport N and the resulting algal blooms to the hypoxic areas at varying efficiencies, location of each treatment plant makes a difference in relative impact on dissolved oxygen per pound of N discharged at end of pipe. Generally, the closer a POTW is to the edge of the Sound, and the closer to the hypoxic zone, the higher the trading ratio (Figure 28). For plants with high trading ratios, economics often favor treatment, while those with lower ratios may find the purchase of credits economically advantageous over treatment.

Table 25: Performance of the NCE, 2002-2006

Trading Year	Credit Price (Dollars)	Purchased (Dollars)	Sold (Dollars)	Purchased (1000 Credits)	Sold (1000 Credits)
2002	$1.65	$1,317,223	$2,357,323	798	1,429
2003	$2.14	$2,116,875	$2,428,636	989	1,135
2004	$1.90	$1,786,736	$2,659,804	940	1,400
2005	$2.11	$2,467,757	$1,315,392	1,170	623
2006	$3.40	$3,828,114	$2,394,956	1,126	704
Total		$11,516,705	$11,156,111	5,023	5,291

Figure 27: Relative nitrogen discharge (lbs/day) from 79 POTWs

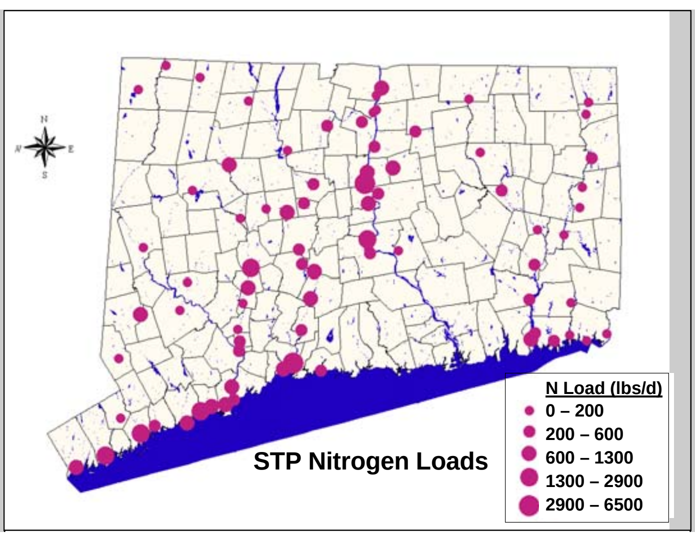
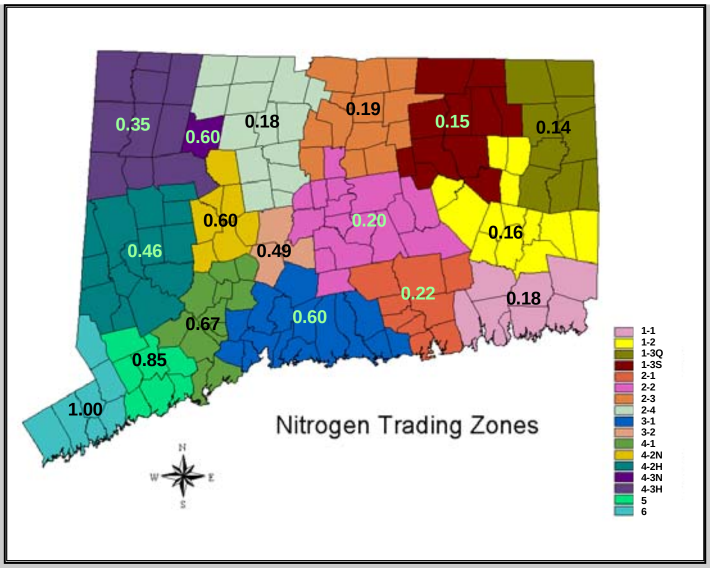


Figure 28 Trading ratios for municipalities in Connecticut



The point source NCE does not reflect a free market approach to trading. Demand is set by the annual general permit limit and supply of credits is constrained by the availability of WCF dollars and the timing and location of N removal projects. Nevertheless, there is a tendency towards implementing cost effective projects as STP authorities decide whether it is less expensive to treat or buy credits, and try to predict when that break-even point might occur that would warrant application for project funding.

Incorporating a NPS/SW component into the existing point source trading program presents some difficult challenges. Among the seven prerequisite conditions listed above that are well met by the current point source program, NPS/SW trading does not provide compelling economic benefits for members; NPS/SW N is difficult to quantify and track; credit cost estimation does not have a strong foundation in any existing programs; NPS/SW credit costs, though geographically diverse, may not result in significant implementation savings; and transaction costs (or time spent negotiating the ground rules for NPS/SW trading) may be considerably higher than for point source credits. Many of these obstacles can be overcome by deferring to models and textbook costs and efficiencies for NPS/SW BMPs. Tracking will still be a challenge because of the sheer number and distribution of BMPs that can be applied throughout the state that will have to be recorded and quantified.

Basic economic principles suggest that a free-market arrangement will not produce many NPS/SW credits for market. Costs are much higher than for point source credits and a regulatory approach must therefore be instituted to formalize the load allocation for nitrogen and to structure participation by municipalities.

If a NPS/SW trading component were to be added in the future, it would most likely also be an incentive-based program rather than a free-market approach. Nitrogen is difficult and costly to control in Connecticut's urban/suburban setting, and reductions are unlikely to be cost competitive with POTW credits in a free market system. However, because municipalities are required to implement the Phase II stormwater permit, and various federal, state and local programs that require or emphasize NPS/SW management, there may be benefits of an incentive-based approach to offset some of those costs. For example, payment for NPS/SW reductions at the same credit prices paid to POTWs under the NCE would help defray costs, and encourage additional nitrogen reductions from stormwater/NPS sources. Connecticut and the NCAB will continue to evaluate and explore the viability of these options.

Further, the cost for management will be enormous. EPA's Clean Water Needs Survey (EPA 2008) has identified more than $200 billion in wastewater management infrastructure needs that does not fully address nutrient control from both traditional point as well as nonpoint/stormwater sources or consider alternative technologies.

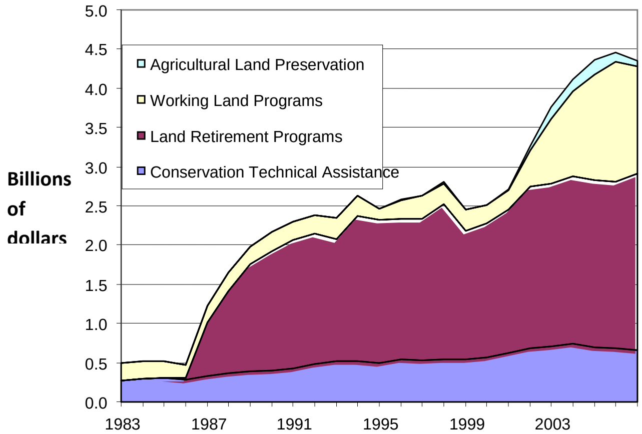
Although there are significance differences between water and air quality trading, there are also several potential barriers to effective trading systems for both media. These are related to accountability and monitoring; establishing standards and management goals; complexities of cross media and multiple source trading, including parity of sources; insurance that outcomes would reduce risk (environmental benefit); economics and marketability of traded credits; and transparency of the program including public outreach and stakeholder involvement.

3.3.3 Government programs, mandates, and policy conflicts

The direct allocation of federal funds, and government policies (for various purposes) have created a variety of results, sometimes in conflict, that directly or indirectly affect Nr generation and management. Chief among these are those associated with U.S. agricultural and land-use policies, energy and transportation policies, and both point and non-point mandated controls on N-bearing aquatic resources including domestic and industrial wastewaters and agricultural runoff.

The principal agricultural conservation programs in the U.S. are administered by the USDA, and consist of the Conservation Reserve and Wetland Reserve Programs (CRP and WRP, land retirement programs), Environmental Quality Incentives Program (EQIP, a "working lands" program), various land preservation programs, and technical assistance programs to agricultural land managers. USDA also manages price support programs and insurance and disaster programs that, collectively, have relatively little potential for impacting Nr management). Figure 29 illustrates funding trends for major initiatives showing the slowing of growth in retirement and assistance programs while preservation and EQIP have increased more recently. The committee is not able to provide guidance on the appropriate levels of funding for these programs. Slowing of the CRP may be a result of energy policy initiatives (see below). Increases in EQIP appear to be associated with greater attention to livestock production, a trend that reflects growing needs for better management practices in this area (see below and section 2.2.4). Of concern to the committee is the need for more effective approaches aimed at encouraging farmers and land managers to adopt proven conservation strategies at the field, farm, and feedlot scale (e.g. more advanced testing methods, geographic position systems-based variable rate fertilizer application, conservation practices for conserving Nr), and landscape scale (e.g. riparian buffers and filter strips, wetlands, and stream restoration). It is clear that the extent of such practices fall far below the technological frontier.

Figure 29: Trends in USDA Conservation Expenditures, 1983-2005



The construction and/or restoration of wetlands have received considerable attention in the past two decades as a conservation method. Such an approach has several positive attributes including promoting denitrification in watersheds containing or receiving Nr, flood protection, habitat preservation, and recreational potential (Hey and Philippi, 1995). In the upper Mississippi basin optimum siting of wetlands could result in as much as 0.4Tg of NO_3^- converted to N_2 (Hey, 2002; Mitsch et al., 1999). Of concern is the potential for the formation of N_2O in such systems if not operated properly. Further details of wetlands as a management tool are presented as an example in text box 6.

Text Box 6: Water Quality Trading in the Illinois River Basin

For various reasons, wetland restoration has been proposed and the magnitude of needed restoration estimated. For the Wetland Reserve Program (WRP), the Farm Bill of 1990 set a goal of restoring approximately 1 million acres. A few years later, the NRC (NRC, 1992) proposed a national goal of restoring 10 million acres of inland and coastal wetlands by 2010. The council went on to recommend that 400,000 miles of streams and rivers be restored by 2012 and that 1 million acres of lakes be restored by 2000, both of which would further the control of reactive nitrogen. While none of these goals has been or is likely to be met by the recommended date, they articulated a need for wetland restoration addressing the important relationship between wetlands and water quality.

Taking into account the economics of using wetlands to manage Nr adds yet another dimension to site selection. Based on the results of the Water Environment Research Foundation's study (Hey et al., 2005), The Kinship Foundation sponsored a study (Kostel et al. in preparation) to define the market for producing and selling Nr (as NO_3^-) credits. For this analysis, a real, potential market area was selected: the Illinois River watershed in Illinois—the tributaries draining Wisconsin, Indiana and Michigan were excluded. The producers of nitrogen credits were identified as "nutrient farmers" and they became the "sellers" of N credits. The "buyers," of nitrogen credits, were restricted to municipal and industrial wastewater treatment facilities, those facilities that hold an NPDES permit. This restriction, of course, resulted in a considerable understatement of the market size since the identified buyers emit less than 11% of the total aquatic N load (David and Gentry, 2000), which finds its way to the Mississippi River—air emission/deposition and agriculture account for the remaining 89%.

The watershed was divided into 19 sub-watersheds, spatially locating credit supply and demand. A linear programming model was developed and used to 1) examine the potential extent and distribution of nitrogen credit demand and supply; 2) compare the average seasonal demand levels to the supply capacity of nutrient farms; and, 3) evaluate the relative effects of seasonality. Market efficiency was imposed through the objective function: the least costly distribution of credit production to meet the given monthly demand. Thereby, sellers and buyers were identified and linked and the spatial characteristics of the market mapped by sub-watershed. At the same time, the equilibrium price of a credit, or the prevailing price at which buyers and sellers are willing to trade, was determined. The market, as represented by the model, determined where the most intensive wetland investment (i.e. wetland restoration) would be, the revenues returned to these investments, and the costs and savings to the buyers.

All 290 permitted dischargers (buyers) are geographically distributed as shown in Figure 30. The mass loading of the buyers (2,423 tons/month) is reflected in Figure 31. 89% of the demand comes from the northeastern corner of the basin (Upper Fox, Des Plaines, and Chicago/Calumet sub-watersheds), the Chicago metropolitan area. As illustrated by Figure 32, 41% of the wetland restoration area (using the criteria discussed above) were identified in the southwestern corner of the watershed (Lower Illinois, La Moine, Macoupin, Lower Sangamon, and Middle Illinois sub-watersheds), where the floodplain is almost entirely leveed. For the market study, the available load of Nr (NO_3^-) by season and sub-watershed was mapped as illustrated in Figure 33. The N load was computed using water quality and flow data collected by the USGS from 1987-1997. The wetland and wastewater cost functions are described in Hey et al., 2005; however, the wetland cost functions were modified for the market study to reflect the variability of land costs across the watershed (i.e., higher land values in urban Chicago vis-à-vis lower land cost in rural Illinois). This variability is reflected in the spatial distribution marginal costs shown for the spring marginal costs grafted in Figure 34. As previously noted, wetland treatment costs vary by time of year because the level of microbial activity, which drives the denitrification process, varies with water temperature. So, in the winter more wetland area is required than in the summer to treat an equivalent load of Nr.

Figure 30 Distribution of municipal (> 1 MGD discharge), and industrial dischargers in the Illinois River Watershed; symbols may represent more than one discharger at that location

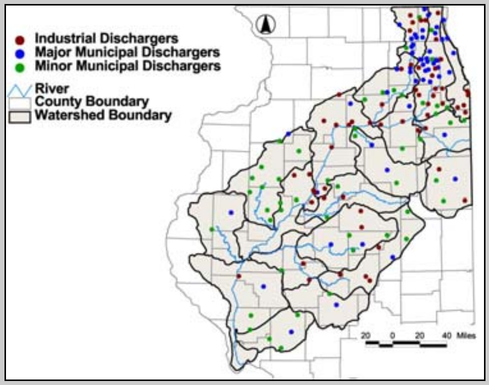


Figure 31: Distribution of total nitrogen emissions by sub-watershed

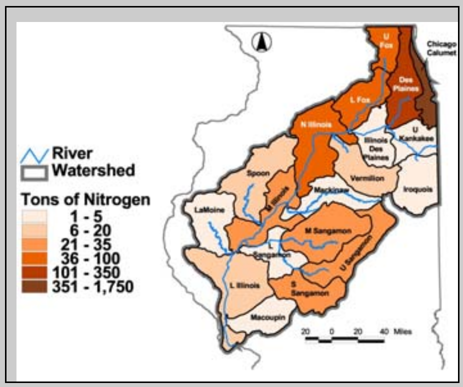


Figure 32: Potential land availability in the 100-year flood zone for nutrient farming in each sub-watershed in the Illinois River Watershed

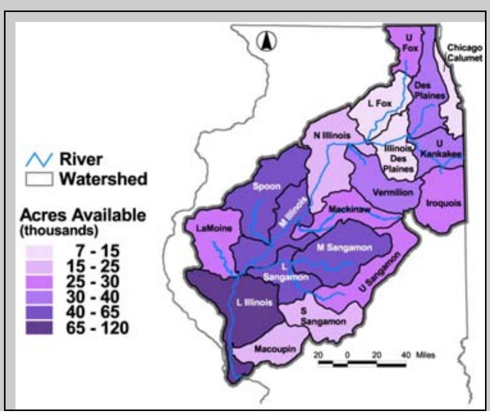


Figure 33: Spring available total nitrogen load by sub-watershed

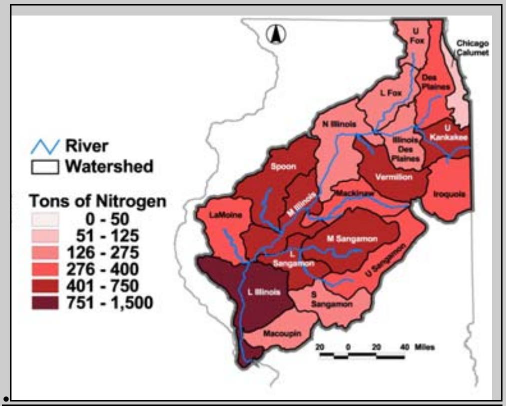


Figure 34: Spring marginal cost (price) by watershed.

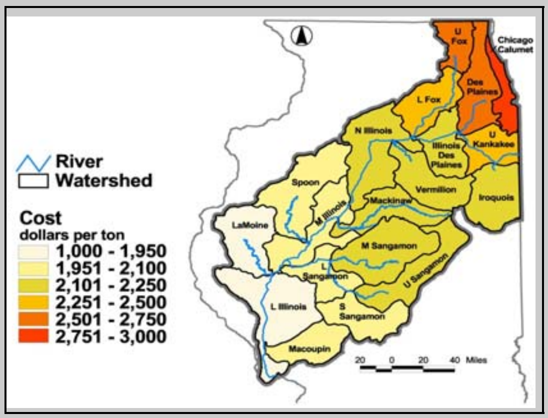
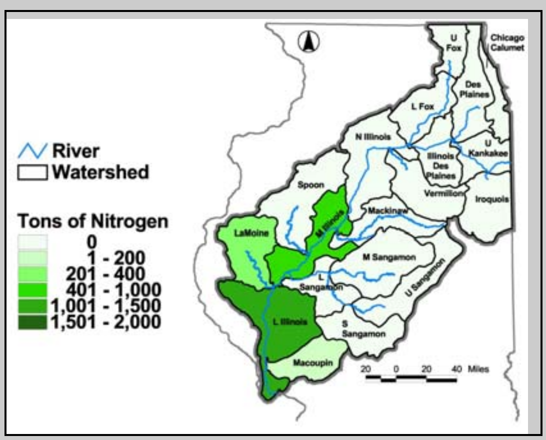


Figure 35: Unrestricted spring credit sales (tons/month) by sub-watershed



Three Regulatory Scenarios

Regulatory agencies may require that dischargers and nutrient farms be located in proximity to each other and could impose “penalties” when the two are not. Thus, for the sake of analysis, the committee created three regulatory scenarios: 1) unrestricted (buyers can purchase nitrogen credits from nutrient farmers anywhere in the watershed without regard to location (the result of this scenario is given in Figure 36); 2) restricted intra-watershed (buyers must purchase all available credits within its own sub-watershed before buying in other sub-watersheds); 3) Accrued 10% penalty (buyers pay an increasing "tax" on credits purchased in consecutive downstream watersheds). The three regulatory scenarios were analyzed for each of the four seasons. All results are can be found in Kostel et al. (2007) or Scott et al. (in preparation).

The "unrestricted" scenario is the least expensive because nutrient farms in this scenario are located downstate where land is least expensive. In the other two scenarios, credits were purchased a little more evenly throughout the watershed. Still, most of the credits in the southern corner of the watershed were purchased. The "restricted intra-watershed" and "accrued 10% penalty" scenarios resulted in more credits being purchased. This resulted in the sale of N credits exceeding the mass of Nr emitted by wastewater treatment, which would benefit the overall control of reactive nitrogen. It also would increase the value of the market and the profits of the nutrient farmer. The down side of such regulatory controls is that they would drive up the price effective price of nitrogen credits. If a buyer had to buy a 1.5 tons for every ton discharged because credits are not available in the tributary watershed, the effective price of a credit would be 1.5 times the price of the tributary sub-watershed. If prices rise too much, "concrete and steel" technologies may become competitive.

Considering all of the point source dischargers in the Illinois River watershed, between 29,000 and 36,000 tons TN/year could be removed through nutrient farming under the studied trading schemes (Table 26). The range of removal is a function of the penalties imposed on the market by the regulatory agencies. Accordingly, the market revenue would range from $70 million to $121 million/year. This is a sizeable market that could generate substantial profits, from $6 million to $38 million with the return on investment varying from 5 to 25%. If the savings are shared evenly between the seller and buyer, the nutrient farmer could earn between $200 and $300/acre/year net profit, which in many cases is greater than the profits from corn or soy bean production. Further, these profits do not include any earnings from flood control or recreation as suggested in the McKnight study report (Hey et al., 2004). With such profits, sufficient land should be available for nutrient farming.

Table 26: Nutrient Farm Market Parameters Under Three Trading Scenarios (Kostel et al. in preparation)

Parameter	Unrestricted	Restricted Intra-watershed	Accrued 10% Penalty
Total Credits Sold (tons)	29,078	29,078	35,781
Total Revenue[10]	$69,925,497	$99,571,889	$121,457,652
Total Cost to Produce Credits	$63,258,006	$66,193,924	$83,288,747
Profit	$6,667,491	$33,377,968	$38,168,905

This analysis indicates that appropriate lands are available and that wetlands can be effectively restored and efficiently used to control reactive nitrogen. The market, structured as discussed

[10] Assumes all credits were sold at the cheapest cost within the Illinois River Watershed.

above, could generate the capital to accomplish the needed large-scale wetland restoration while saving tax payers the cost of upgrading their municipal wastewater treatment plant (TWI, 2007).

3.3.4. Biophysical and technical controls (control points) on transfer and transformations of Nr in and between environmental systems.

Within the nitrogen cascade there are a number of places where the flow of Nr is constrained or regulated, either by nature or by human intervention, or a combination of the two. This report refers to these places in the cascade as "control" points. The control points may restrict the flow of Nr species within environmental systems (atmospheric, terrestrial, aquatic) or between them. The control points vary from primary controls where Nr is minimized through conservation measures or through after the fact measures that attempt to convert Nr that is emitted or not fully used to nonpolluting products, such as to N_2 by denitrification or through long-term storage. The discussion of choke points is primarily focused on biophysical controls in terrestrial and aquatic environmental systems. A concluding section on technical controls discusses possibilities for decreasing NO_x emissions from combustion.

3.3.4.1. Biophysical controls in terrestrial environmental systems

Approximately 36 Tg of new Nr is introduced into the US each year (Table 1). This new Nr is derived from consumption of ~11 Tg of synthetic N fertilizer, ~8 Tg of N is fixed by biologically by crops, and ~ 5 Tg is emitted from fossil fuel combustion annually. This N is used to produce food and fiber (~15 Tg) or is formed during electrical generation, industrial production or transportation. Efforts to decrease the creation of new Nr should first look to conservation.

Conservation of fossil fuel and resulting decrease in use of fertilizer in food and fiber production or decreased Nr emission can come through a variety of mechanisms such as more energy efficient industrial processes, energy efficient homes and vehicles. Further gains are possible through conservation practices and alternatives to wasteful approaches, such as improving public transportation to minimize use of personal automobiles, and use of local products that don't require long-distance shipping.

Conservation in food and fiber production and food consumption can also play an important role in limiting Nr. As agriculture is the largest consumer and producer of Nr, consumption of fertilizer N could be decreased by changes in diet and increasing fertilizer N use efficiency in crop and fiber production systems. The control points discussed in this section include: protein consumption in the human diet, removing croplands that are susceptible to Nr loss from crop production, decreasing fertilizer N demand by increasing fertilizer use efficiency in crop and fiber production, turf grass and nitrogen fertilization in the US, and managing Nr during recycling through livestock production.

(a) Decreasing the amount of fertilizer N needed through changes in human diet. Along with increasing fertilizer N use, continued high intake of protein in developed countries and changes in the diet of people in developing countries will likely lead to greater N losses from global food production in the future. The first aspect of changes in food production concerns increasing protein consumption as global population increases and gets wealthier, which is likely to require increased N input into food production (Galloway et al. 2007; Naylor et al, 2005).

The average protein supply per person in developed countries is presently ~100 g per day, while in the developing countries it is only ~65 g per day [Food and Agricultural Organization Statistical Database (FAOSTAT), 2003). Protein is used because there is a direct proportionality between protein and nitrogen composition of food (ca 0.16 g N per 1 g protein). On average in 1995, developed countries consumed ~55% of total protein from animal sources while developing countries derived ~25% of total protein from animals. Protein consumption was highest in the U.S. and western Europe, ~ 70 and ~60 g animal protein per person per day, respectively. In 2003, total protein consumption in the U.S. was 115 g person per person per day (74 derived from animals and 41 from vegetable (FAOSTAT, 2003). In developing countries, the greatest change in animal protein consumption has occurred in China where the consumption of meat products has increased 3.2 fold (from ~ 10 to ~32 g per person per day) since 1980. In Sub-Saharan Africa there has been no increase in either total (~ 50 g per person per day) or animal protein (~ 10 g per person per day) consumption during the past 30+ years (Mosier et al. 2002).

The reason for focusing on the consumption of animal protein is that more N is needed to produce a unit of animal protein than an equal amount of grain protein. Bleken et al. (2005) note that the N cost of animal production in Norway and the Netherlands was approximately five units of N in feeds for each unit of N produced. Approximately 2.5 units of N are required to produce a unit of wheat protein-N. Bequette et al. (2003) report that dairy cattle consume four units of N in feeds (including forage and grains) for every unit of N that appears in milk. Using a range of efficiencies for animal production practices, Kohn et al. (1997) estimated that 4 to 11 units of fertilizer N would be used in a whole farm system to produce a unit of milk protein. This ratio would be lower when using legume N to feed cattle, as is commonly done. Based upon the extra N required to produce animal protein compared to grains, continued high protein consumption in developed countries and changes to higher protein diets in developing countries will likely increase N input and losses in food production.

Moderating this increase by decreasing the average amount of total protein consumed in developed countries is one mechanism of limiting part of the expected increased N requirement in food production. One example of a country with a good food supply and moderate consumption of animal protein is Italy in 1963. At that time food supply was adequate to ensure sufficient nutrition to all groups of society (Bleken 1997). Total protein consumption was 85 g per person per day, and consumption of animal protein was 32 g, roughly half of the current US. diet, and yet much higher than the average of developing countries. Another example is Japan, where animal protein consumption has traditionally been low, although it has increased from 25 g in 1963 to 54 g animal protein per person per day in 1995. In the same period the total protein consumption has increased from 73 g to 96 per person per day.

Bleken (1997) analyzed the relation between human diet and global N need for food production. Her analysis indicates that the total N needed for diets with high animal protein intake (comparable to many industrialized countries today) are almost twice as high as the N needed for the average diet in Italy 1963, mentioned above, or for Turkey in 1993. Based on her analysis, the committee assumes that in the high-N input regions per capita N need for food production may be reduced by 45%, which would reduce present-day N inputs by 15% worldwide.

Switching to a lower protein diet may not, however, reduce N losses if the new diet includes increased quantities of fruits, vegetables, and nuts, in addition to staple grains, beans and pulses.

Vegetables, fruit and nuts are high value crops that typically require large inputs of fertilizers and pesticides when produced at a large, commercial scale, and N fertilizer losses can be considerably larger than for grain crops. Having a very diverse diet that includes a wide range of high value fruits and vegetables that are available 365 days a year whether they are in-season locally or not, also have consequences for N inputs/outputs from agriculture--both within the U.S. and globally. Additional Nr may be conserved by decreasing the amount of food that is wasted.

(b) Removing croplands that are susceptible to Nr loss from crop production. Booth and Campbell (2007) analysis of NO_3^- loading in the Mississippi River Basin provides estimates of N input from agricultural lands to be similar to those estimated by Del Grosso et al. (2006). These recommendations are essentially the same as those arrived at in the original national hypoxia assessment which suggested that the most leaky lands be taken out of production (Doering et al. 1999). Booth and Campbell state that,

> *Nitrogen derived from fertilizer runoff in the Mississippi River Basin (MRB) is acknowledged as a primary cause of hypoxia in the Gulf of Mexico. To identify the location and magnitude of nitrate runoff hotspots, and thus determine where increased conservation efforts may best improve water quality, we modeled the relationship between nitrogen inputs and spring nitrate loading in watersheds of the MRB. Fertilizer runoff was found to account for 59% of loading, atmospheric nitrate deposition for 17%, animal waste for 13%, and municipal waste for 11%. A nonlinear relationship between nitrate flux and fertilizer N inputs leads the model to identify a small but intensively cropped portion of the MRB as responsible for most agricultural nitrate runoff. Watersheds of the MRB with the highest rates of fertilizer runoff had the lowest amount of land enrolled in federal conservation programs. Our analysis suggests that scaling conservation effort in proportion to fertilizer use intensity could reduce agricultural nitrogen inputs to the Gulf of Mexico, and that the cost of doing so would be well within historic levels of federal funding for agriculture. Under this simple scenario, land enrolled in conservation programs would be increased by about 2.71 million hectares, a 29% increase over 2003 enrollments, while land taken out of traditional fertilized agriculture and enrolled in conservation programs would constitute about 3% of 2003 fertilized hectares.*

The Booth and Campbell approach places the leakiest intensively cropped lands into government programs like the Conservation Reserve Program - where they would be put into grass or cover crops. Doering et al. (1999) had a somewhat different approach. Under their analysis, nitrogen use or nitrogen loss reductions were imposed on agriculture and the USMP model adjusted crop rotations, tillage practices and fertilizer inputs within the Mississippi Basin - meeting the given Nr constraint while maximizing producer and consumer welfare. The model favored those crops and cropping systems at different points in the landscape having low nitrogen leakage. Where the model could not find a crop production system having positive returns while meeting the Nr restrictions, then that land was retired from production. This analysis suggests opportunities for maintaining land in agricultural production while still reducing Nr losses through better matching of land characteristics with crops and cropping systems.

(c) Decreasing fertilizer N demand by increasing fertilizer use efficiency in crop and fiber production. The largest input of Nr in North America is N) fertilizer used for crop production. The mean annual N fertilizer input to North America between 1999 and 2003 was 12.5 Tg. Of this fertilizer N, 66% was used to fertilize cereal crops, mainly corn and wheat (Dobermann and Cassman, 2005).

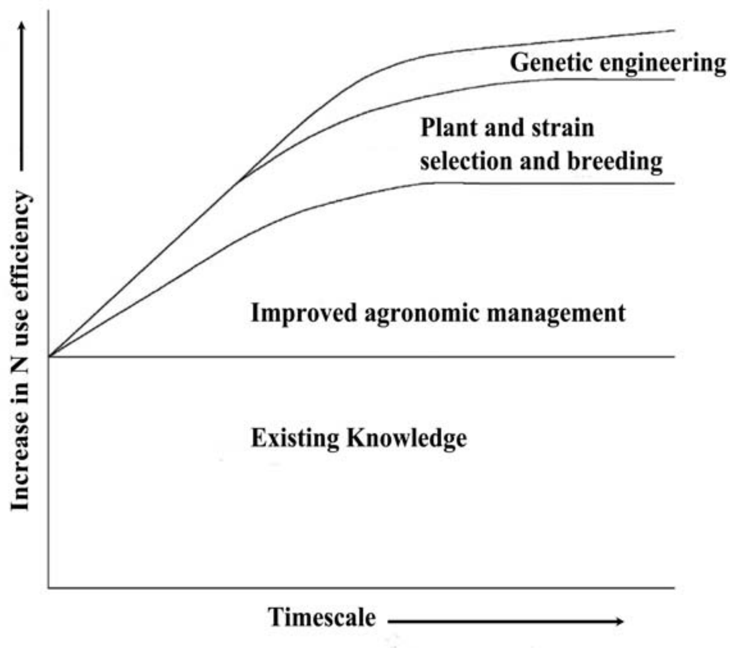
Corn yield in the U.S. has increased from an average of 100 bu/ac in 1985 to 136 bu/ac in 2005 as a result of improved nutrient and pest management, expansion of irrigated area, conservation tillage, soil testing, and improved crop genetics (yield and pest resistance) [Council for Agricultural Science and Technology (CAST), 2006]. From 1980 to 2000, N-fertilizer use efficiency (NFUE, kg grain produced per kg applied N, hereafter expressed as kg grain / kg N) increased from 42 to 57 kg grain / kg N, a 35% efficiency gain during a period when average U.S. corn yields increased by 40% (Fixen and West, 2002). Despite this steady increase in NFUE, the average N fertilizer uptake efficiency for corn in the north-central U.S. was 37% of applied N in 2000 based on direct field measurements (Cassman et al. 2002). These results indicate that a large majority of the applied N fertilizer is vulnerable to transfer pathways such as volatilization, denitrification, runoff, and leaching. The results also suggest there is substantial room for improvement in N efficiency currently achieved by farmers.

Although progress has been made to increase both cereal yield and NFUE, a concerted effort to further increase NFUE remains a logical control point to reduce production costs, because N fertilizer represents a significant input cost, and to limit Nr leakage (e.g. NH_3, NO_x, N_2O, NO_3^-) from agroecosystems.

The goal of reducing Nr while sustaining adequate rates of gain in cereal production to meet expected food demand will require increases in NFUE, which in turn will require innovative crop- and soil management practices. This need is exacerbated by the recent increase in demand for corn to produce ethanol biofuel. The concept of improved N synchrony—practices that better match the amount, timing, and geospatial location of applied N to crop-N demand and the N supply from indigenous soil resources-- is generally viewed as the most appropriate approach for improving NFUE (e.g. Appel, 1994; Cassman et al. 2002). The challenge is for greater synchrony between crop N demand and the N supply from *all sources* (e.g. soil, fertilizer, organic inputs such as manure, compost, or green manures, etc) throughout the growing season. Losses from all N-loss mechanisms increase in proportion to the amount of available N present in the soil profile at any given time.

Several promising technologies and combinations of technologies have emerged in recent years. Significant increases in NFUE are often achieved through reductions in N fertilizer use by 10 to 30 %, while increases in yield tend to be small (Giller et al. 2004). Figure 36 indicates where expected greatest gains in NFUE are to be realized in the future from different technology options. Each of these improvements in management and genetics helps to better match the amount and timing of applied N to crop-N demand and the N supply from indigenous resources. However, large investments in research, extension education, and technology transfer will be required, and significant incentives implemented, to achieve the degree of improved synchrony needed to make substantial improvements in NFUE. The need to accelerate the rate of gain in crop yields to meet increasing demand for human food, livestock feed, and biofuels represents an additional new challenge. Crop prices are expected to rise as they more closely track the price of petroleum (CAST, 2006). Higher crop prices will motivate farmers to achieve higher yields, and higher crop yields require a greater amount of N uptake to support increased biomass production (Greenwood et al., 1990). Therefore, an explicit emphasis on developing technologies that contribute to both increasing yields and NFUE will be needed to ensure that the goals of food security, biofuel production, and protection of environmental quality are met.

Figure 36: The likely impact of research investment in increasing N fertilizer use efficiency (Giller et al. 2004)



(d) Managing Nr during recycling through livestock production. Newly fixed Nr is produced biologically or added as fertilizer to meet the demand for food and fiber production. Much of the N is used in cereal crop production and cereal crops are then used to feed livestock. The new Nr is then recycled through the livestock production system and becomes again susceptible to transfers to the atmosphere as ammonia and NO_x, is available for additional N_2O production, and movement into aquatic systems as NH_4^- and NO_3.

The bulk of the N fed to livestock ends up in manure, and where this manure (~ one half in urine and one half in feces) is produced, there is often a much greater supply than can be efficient or economically used as fertilizer on crops. For large animal feeding operations (AFO's) there is considerable expense associated with disposal of the manure. Various storage systems have been developed to deal with this excess manure, the most interesting of which, from the standpoint of integrated policy on N, convert the urea to N_2. These represent a choke point where reactive N is removed, on time scales of millennia, from biogeochemical cycles. The fraction of the feed N that is converted to N_2 or even can be converted to N_2 remain major unanswered scientific or technical questions; this brief report reviews the current state of knowledge.

The NRC (2003) report noted the paucity of credible data on the effects of mitigation technology on rates and fates of air emissions from AFO's, but called for their immediate implementation.

That report also called for a mass balance approach in which the losses of N species such as NH_3, NO, N_2, and N_2O are expressed as a fraction of the total N loss. Quoting from the NRC report:

> *Storage covers for slurry storage tanks, anaerobic lagoons, and earthen slurry pits are being studied as a method to decrease emissions from those containments. Both permeable and impermeable covers are being studied. Tested covers range from inexpensive material such as chopped straw (on slurry containments only) to more expensive materials such as high density polyethylene. Covers can decrease emissions from storage but their net effect on emissions from the system is conditional on how the effluent is used on the farm.*
>
> *Anaerobic digestion in closed containment has been studied for many types of applications. Anaerobic digestion is the process that occurs in an anaerobic lagoon. When conducted in closed vessels, gaseous emissions including methane, carbon dioxide and small amounts of other gasses (possibly ammonia, hydrogen sulfide, and VOCs) are captured and can be burned for electricity generation, water heating, or simply flared. The in-ground digester being tested on a swine farm in North Carolina is an example of the ambient temperature version of this technology (there are also mesophilic and thermophilic designs). The concentration of ammonia remaining in effluent from that digester is higher than the concentration in lagoon effluent and can be volatilized once exposed to air.*

Recent research (e.g. Shores et al. 2005; Bicudo et al. 2004; Funk et al. 2004a; Funk et al. 2004b) demonstrates reduction in NH_3 emissions after a permeable cover was installed. Miner et al. (2003) reported that a polyethylene cover can reduce NH_3 emissions by ~80%, but it is not clear what fraction of that N was converted to N_2. Harper et al. (2000) reported that in a well-managed swine lagoon denitrification N_2 losses can be equivalent to N lost as NH_3, in other words about 50% efficiency. Kermarrec et al. (1998) reported that sawdust litter helps reduce NH_3 emissions from pig manure with 44-74% of manure N converted to N_2, but > 10% of the manure N was released as N_2O. Sommer (1997) found that NH_3 was emitted from cattle and pig slurry tanks at the rate of 3.3 kg N m^2/yr until covered with straw. After straw application NH_3 emissions were below detection limit. Mahimairaja et al. (1994) reported that NH_3 volatilization was reduced by 90-95% under anaerobic conditions. Section 2.2.4.5 contains a discussion of best management practices to minimize NH_3 emissions from livestock waste, and presents finding and recommendation 6 on the need for a framework for manure management.

(e) Alternatives to current urban landscaping practices. Section 2.2.5 discussed the use of turf grasses as a prominent feature in U.S. urban landscapes with over 1 TgN used to fertilize lawns each year (Table 9). New developments are most amenable to landscaping practices that may minimize the need to use supplemental fertilizer including preservation of the natural soil profile, use of turf types that require little or no fertilizer, minimizing turf areas, using organic maintenance techniques and choosing alternatives to lawns and exotic plant species such as naturalistic landscaping. Many of these practices are part of a low impact development philosophy, which can also combine other best management practices to mitigate the effects of impervious cover and landscape changes. Existing development is also amenable to many of

these practices, especially conversion of typical residential and commercial lawns to natural landscapes and retrofitting other BMPs that promote infiltration, such as rain gardens.

(f) Structural and non-structural Best Management Practices (BMP) to treat runoff: There are probably hundreds, if not thousands, of BMPs that have been designed and manufactured to treat runoff from both urban and agricultural lands. Whether applied to new development or existing agricultural or urban land use, most follow basic principles that simulate natural land features and processes that remove pollutants from runoff. They promote infiltration to take advantage of the cleansing value of passage through soils and to reduce runoff volumes, and provide for biological or chemical conditions that help remove pollutants.

The most notable of the processes for Nr is providing conditions that are adequate to denitrify Nr in the waste-stream in a process called biological nitrogen removal, or BNR. BNR simply creates conditions that convert initial forms of nitrogen to nitrate via oxidation, and convert nitrate to dinitrogen gas by providing conditions, especially high carbon and low oxygen, where the denitrification process can occur. These simulate natural conditions such as nitrification that occurs in oxic soils as water-borne nitrogen infiltrates into the soils and groundwater, and denitrification that occurs in highly-organic, saturated soils such as in wetlands, where oxygen is low.

Most BMPs are considered structural, and may be highly engineered “package” plants that can treat sewage or runoff, depending on scale and structure, or simple detention basins that allow sediments and adhered pollutants to settle out. “Artificial” wetlands are a good example of a more sophisticated BMP that takes advantage of natural processes, and may be created at the end of the stormwater pipe, or at edge of field. Non-structural BMPs are often preservation actions, as discussed earlier, or activities that prevent pollutants from entering the wastestream such as street sweeping or fertilizer limitation.

(g) Wetlands to decrease NO_3^- loading of aquatic systems. Much of the nitrate leached from agricultural fields could be removed from drainage water in wetlands, either natural, created, or restored. Nitrate removal from the water column in wetlands is performed by plant uptake, sequestration in the soils, and microbial transformation that include immobilization and denitrification. Plant uptake and microbiological immobilization result in temporary storages in the system since most nitrogen will eventually return to the wetland via plant death and decomposition. In contrast, denitrification constitutes a real nitrogen sink because in this process bacteria reduce NO_3^- to nitrogenous gases (N_2, NO, N_2O) that are emitted to the atmosphere (Clement et al., 2002). In general, NO_3^- removal by wetlands, primarily caused by microbial denitrification, varies seasonally, with highest rates during summer and lowest rates during the coldest temperatures (Mitsch et al., 2000; Spieles and Mitsch, 2000; Hernandez and Mitsch, 2007). 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 pollutant swapping, i.e., exchanging NO_3^- for N_2O is advisable.

In addition to preserving existing wetlands there are two basic approaches utilizing wetlands for reducing the N and other nutrients from reaching rivers and streams and especially vulnerable downstream coastal systems: 1) creation and restoration of ecosystems, principally wetlands and riparian forests, between farms and adjacent ditches, streams and rivers; and 2) diversion of rivers into adjacent constructed and restored wetlands all along the river courses (See Chapter 4).

At a series of workshops on restoration of the Mississippi-Ohio-Missouri (MOM) River Basin in 2003-04 (Day et al., 2005; Mitsch and Day, 2006), scientists and managers were asked to focus on needed research and chokepoint opportunities, especially for managing N in that basin. They concluded that a major, interdisciplinary research program, as a lead-in to the actual restoration of wetlands and rivers, needs to take place with sufficient funding, study sites, and time to reduce remaining uncertainties about the efficacy of wetlands to solve pollution problems related to N. Twenty to thirty full-scale, existing and new agricultural/wetland demonstration projects should be located throughout the country and instrumented to study agricultural runoff into wetlands in a variety of soil conditions. Pilot and full-scale studies are needed of diversions into riparian systems along river channels to determine their effectiveness for nutrient removal.

To give scale to the solution needed, restoration of over 2 million hectares of wetlands is needed in the MOM basin to reduce the nitrogen load to the Gulf of Mexico sufficiently to ensure a reduction in the size of the hypoxia (Mitsch et al., 2001; Mitsch and Day, 2006; see Chapter 4). If wetlands could be economically and effectively restored where croplands now exist on hydric soils within the 100-year floodplain, returning croplands that are on hydric soils may be an important NO_3^- control mechanism. Cropland on hydric soil in the floodplain occupy about 2.8 million hectare, 40% more than is needed for the restoration. If this area and its wetlands were given back to the Mississippi, over a million tons of NO_3^--N would be annually removed or prevented from reaching the Gulf of Mexico (Hey et al. 2004). A recommendation encouraging "wide spread wetland restoration and creation with strategic placement of these wetlands where reactive nitrogen is highest in ditches, streams, and rivers" is made in Recommendation 9 (see section 2.3.1.3).

3.3.4.2 Technical controls (control points) on transfer and transformations of atmospheric emissions of Nr in and between environmental systems: NO_x

A major contributor to Nr in the atmosphere is fossil fuel combustion. During the combustion process NO_x ($NO_x = NO + NO_2$) are released to the atmosphere. Globally the production of NO_x has accelerated the last few decades through, primarily the increase in fossil fuel combustion (Galloway et al., 1995; 2008). With this increase in emissions from ~5 Tg N in 1940 to ~ 25 Tg N in 2005, combustion of fossil fuels account for about 50% of the total global NO_x emissions for 1990. Of the anthropogenic sources, fossil fuel, aircraft, biomass burning, and part of the soil emission are most important (Holland et al., 1997). Although global NO_x emissions continue to increase, these emissions are declining in the US. (see section 2.2.2).

Nitrogen oxide is formed during combustion by three mechanisms:

- thermal NO_x where N and oxygen (O_2) gas, present normally in combustion air, combine at high temperatures, usually above 1600 C to form NO through the Zeldovich mechanism.
- fuel NO_x where nitrogen from a fuel, e.g., coal and biofuels, is released as some intermediate and then combines with O_2 to form NO, and

- prompt NO_x where nitrogen gas combines with radical components of the fuel, forming various compounds including hydrogen cyanide and other cyano radicals. These in turn form NO_x. Contributions of prompt NO_x are usually low as compared to fuel NO_x.

There are several ways to control NO_x. The most common controls are on coal-fired electric utility generators and those are discussed below. Following electric utility generator controls, or external combustion systems, there is a discussion on internal combustion controls.

Reduction of the temperature limits the kinetics of the N/ O_2 reaction. Temperature can be controlled by using a fuel-rich mixture versus fuel lean. In this case the reactions to take place at lower temperatures. Fuel-rich mixtures also reduce the amount of O_2 available for reaction and there are changes to the chemical mechanisms which limit the oxidation of N_2. If fuel-lean mixtures are used for temperature control, while the temperature is lower, there is a significant amount of O_2 present. Typically in external combustion systems, this is implemented by using less excess air and using staged combustion. In addition, flue-gas recirculation (FGR) is used to lower the temperature. Low-NO_x burners operate under the principle of internally staging the combustion. To reduce fuel NO_x, air and fuel staging are used to reduce the peak temperature where air and fuel are admitted in separate locations.

Chemical reduction of NO_x is also possible. These methods include: selective non-catalytic reduction (SNCR); SCR; and fuel reburning. SNCR is an add-on technology where urea or NH_3 is injected in a controlled temperature zone to allow for the reduction of NO_x. SCR is also an add-on technology where the flue gas must pass through a catalyst bed to allow for reaction between ammonia and NO_x. Care must be taken with both technologies to avoid NH_3 slip. Fuel reburning requires the injection of a fuel to create a zone where NO_x is reduced to N_2. Low NO_x burners may also use an internal fuel reburning to reduce the NO_x.

For internal combustion engines, the same mechanisms as discussed above are used but in a variety of different ways, since these systems are using high pressure and predominately have thermal NO_x versus fuel NO_x formation. Most technologies involve the need to reduce the peak temperature and duration of high temperatures of the combustion zone. For example, gas turbines utilize low NO_x burners, while spark ignition engines utilize a three-way catalyst which requires less than 0.5% O_2. In this case, additional NO_x is reduced by utilizing unburned fuel as a reagent over the catalyst for chemical reduction of NO_x. It should be noted however, that a side reaction for the three-way catalyst system produces ammonia. For diesel engines, delaying the injection of the fuel, and for spark ignited engines retarding the timing can reduce NO_x emissions. Engines also use exhaust gas recirculation (EGR) to reduce the peak temperatures. Recent road side studies have indicated high efficiency (~90%) for NO_x removal from the American light-duty fleet (Bishop and Stedman, 2008).

3.4 Risk reduction recommendations

3.4.1 Overarching recommendations

Human activities have significantly increased the introduction of Nr into the U.S. environment and, through radical alterations of land use, have eliminated many of the natural features that once may have provided pollutant treatment. While there have been significant benefits resulting from food production, there have also been, and continue to be, major risks to the health of both ecosystems and people due to the introduction of Nr into the nitrogen cascade. To optimize the benefits of Nr, and to minimize its impacts, will require an integrated nitrogen management strategy that not only involves EPA, but also coordination with other federal agencies, the States, the private sector, universities, and a strong public outreach program. The committee understands that there are real economic costs to the recommendations contained in this report. For each recommendation there will of necessity be tradeoffs derived from the varying cost-effectiveness of different strategies.

The committee makes three over-arching recommendations:

Recommendation A

An integrated approach to the management of Nr will likely use a combination of these four implementation mechanisms. Each mechanism must be appropriate to the nature of the problem at hand, supported by critical research on decreasing the risks of excess Nr, and reflect an integrated policy that recognizes the complexities and tradeoffs associated with the nitrogen cascade. Management efforts at one point in the cascade may be more efficient and cost effective than control or intervention at another point. This is why understanding the nature and dynamics of the N cascade is critically important.

Recommendation B

EPA should form an Intra-agency Nr Management Task Force that will build on existing Nr research and management capabilities within the Agency. This Intra-Agency Task Force should be aimed at increasing scientific understanding of: 1) Nr impacts on terrestrial and aquatic ecosystems, human health, and climate,2) Nr-relevant monitoring requirements, and 3) the most efficient and cost-effective means by which to decrease various adverse impacts of Nr loads as they cascade through the environment.

Recommendation C

EPA should join with other agencies within the US government in establishing an Inter-agency Nr Management Task Force. The members of this Inter-Agency Task Force should include at least the following federal agencies: US Department of Agriculture (USDA), US Department of Energy (DOE), US Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), US Geological Survey (USGS), US Forest Service (USFS,) and Federal Emergency Management Agency (FEMA)). This Task Force should coordinate federal programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and managing Nr in the environment.

The intra- and inter-agency Nr-Management Task Forces should take a systems approach to both scientific research and public policy by emphasizing the following research and management goals:

- Development of methods to help implement a systems approach
    - developing and evaluating proposed Nr budgets

- developing appropriate life cycle accounting methods
- developing monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads
- evaluating the critical loads approach to air and water quality management
- developing Nr indicators for excess Nr effects on human health and environment
- developing new systems-based approaches for controlling Nr releases to the environment

- Enhancing ecosystem services that lead to the denitrification of Nr in the landscape including reconnecting rivers and streams to their floodplain, creation and restoration of wetlands in agricultural landscapes, and stream and ditch enhancements that increase the surface area of potential denitrification.
- Best management practices (BMPs)
  - developing the scientific understanding required for identifying best management practices (BMPs) for specific application, including:
    - Nr applications in agriculture to ensure adequate food, feed, fiber, and bioenergy feedstock supply while also avoiding negative impacts on the environment and human health;
    - Nr applications for developed (e.g., residential and commercial) runoff mitigation and landscape maintenance;
    - planning and pollution prevention including low impact development and natural ecosystem service preservation;
    - Enhancing the appropriate matching of crops, cropping systems, and land types and capabilities for the most productive use of Nr and the reduction of excess Nr
    - development and natural ecosystem service preservation;
    - primary use of natural land features and attributes, such as wetland preservation and enhancement, natural soil profiles and buffer strips;
    - improved removal of Nr from sewage waste streams at both large-scale wastewater treatment facilities and individual subsurface (septic) systems
  - establishing proactive extension and technology transfer approaches to facilitate adoption of BMPs
- Assessment activities
  - assessing combined carbon (C) and Nr effects on terrestrial and aquatic ecosystems

  - assessing indicators/endpoints, costs, benefits and risks associated with impairment of human health and decline and restoration of ecosystem services
  - reviewing existing and proposed legislation for purposes of extending Nr regulatory authority or streamlining procedures for enacting Nr risk reduction strategies
  - evaluating economic incentives, particularly those that integrate air, aquatic, and land sources of excess Nr

- Developing new education, outreach, and communication initiatives

3.4.2 Near-term target recommendations

INC makes four recommendations that set near-term targets for the decrease of Nr entering the environment from various sources. These, and their rationale, are set forth below.

Target Goal 1. Controls on NO_x emissions from mobile and stationary sources

The Clean Air Act (1970) and its Amendment (1990) have resulted in NO_x emissions that are less than 50 percent of what they would have been without existing controls. While this is an admirable accomplishment, there is still a need to seek improvements. NO_x emissions are an order of magnitude greater than at the beginning of the 20th century. As a consequence there remain significant negative impacts on both humans and ecosystems. In 2002, coal-fired utilities generated approximately 1.3 Tg N annually. If all coal-fired plants used state-of-the-art NO_x controls, this number could be reduced by 0.6 Tg N/yr; in fact the NO_x State Implementation Plans enacted in 2003 and 2004 reduced 2002 emissions by 0.3 Tg N/yr, so in essence, half the reduction has already been accomplished. The EPA should continue to reduce NO_x emissions from major point sources, including electric generating stations and industrial sources, expanding the use of market mechanisms such as cap and trade. Under this scenario, it is likely that high efficiency, low emission power plants will be built for energy needs.

For mobile sources, emissions for highway and off-highway sources are approximately 2.2 Tg N/yr and 1.2 Tg N/yr, respectively. For on road vehicles, better controls for heavy duty diesel vehicles are needed. For off-road vehicles, which include locomotives, construction, farm, landscaping equipment, and marine vehicles, there are currently no controls, but 80-90% NO_x removal is technically achievable. Assuming a 40% reduction for these sources, there is a potential reduction of 1.4 Tg. The total reduction for both mobile and stationary sources is then approximately 2 Tg N/yr. Part of achieving such levels of compliance will require the implementation of inspection and maintenance programs or road-side monitoring.

INC cautions, however, that achieving such a goal may be inadequate for many areas to meet the new 65 ppb ozone standard recommended by the CASAC or even the 75 ppb currently promulgated. Additional measures such as increasing the role of solar- and wind-generated electricity, wider use of hybrid and electric cars, and public transit conducive to energy conservation and reduced emissions should be promoted.

Target Recommendation 1. *INC recommends that the EPA expand its NO_x control efforts from the current decreases of emissions of light duty vehicles (including passenger* *cars) and power plants to include other important unregulated mobile and stationary sources (e.g., off road vehicles) sufficient to achieve a **2.0 Tg N/yr** decrease in the generation of reactive nitrogen.*

<u>Target Goal 2.</u> Nr discharges and emissions from agricultural lands

INC finds that excess flows of Nr into streams, rivers, and coastal systems can be reduced by approximately 20% **(~1 Tg N/yr)** through improved methods of landscape management and without undue disruption to agricultural production. This would include activities such as using wetland management (e.g., USDA Wetlands Protection Program), improved tile-drainage systems and riparian buffers on crop land, and implementing storm water and nonpoint source management practices (e.g., EPA permitting and funding programs).

In addition, INC believes that crop N-uptake efficiencies can be increased by up to 25% over current levels through a combination of knowledge-based practices and advances in fertilizer technology (such as controlled release and inhibition of nitrification). Crop output can be increased while reducing total Nr by up to 20% of applied synthetic fertilizers, approximately **2.4 Tg N/yr** below current levels of Nr additions to the environment. These are appropriate targets with today's available technologies. Further progress is possible through expanded research programs.

INC is concerned about current policies and practices governing biofuel development. Acreage devoted to corn production has increased substantially for corn based ethanol production during the past several years (with nearly one-third of the crop currently devoted to bioethanol production), with fertilizer nitrogen increasing by at least 10% (an additional 0.5 Tg N/yr), largely to meet biofuel feedstock crop demand. In the absence of Nr controls and a failure to implement best practices, current biofuels policies will make it extremely difficult to reduce Nr transfers to soils, water and air (Simpson et al. 2008). Integrated management strategies will be required.

INC also notes with concern the increase of N_2O in the atmosphere. INC believes that greenhouse gas (GHG) emissions trading will provide both opportunities and challenges for mitigating Nr environmental and health impacts. Policies and regulations should consider how to reward reductions of N-related GHG. Biofuel subsidies that accurately account for Nr contributions to GHG emissions, certification of individual biofuel plants for GHG impact, and rewards for farmers who reduce N_2O emissions are examples of how an integrated strategy can reduce agricultural GHG impacts. For additional production of liquid biofuels beyond the grandfathered amount in the Energy Independence and Security Act (EISA), EPA has the power to exercise some controls on N_2O emissions through the life cycle greenhouse gas accounting requirements. In this regard, the committee endorses Section 204 of the EISA calling on the Agency to adopt a life cycle approach to the assessment of future renewable fuel standards as a positive step toward a comprehensive analysis.

Target Recommendation 2. *INC recommends a goal of decreasing livestock-derived NH_3 emissions by 30% (a decrease of **0.5 Tg N/yr**) by a combination of BMPs and engineered solutions. This is expected to decrease $PM_{2.5}$ by ~0.3 µg/m^3 (2.5%), and improve health of ecosystems by achieving progress towards critical load recommendations.*

*Additionally we recommend decreasing NH_3 emissions derived from fertilizer applications by 20% (decrease by **~0.2 Tg N/yr**), through the use of NH_3 treatment systems and BMPs.*

<u>Target Goal 3.</u> Ammonia emissions from livestock management and manure handling

In spite of gains made over the last several decades in decreasing the amount of NOx emitted from stationary and mobile combustion sources, the total amount of Nr released into the atmosphere has remained relatively constant. This is related to the essentially unregulated release of ammonia from livestock operations. At the present time, fewer livestock are required to produce more animal products than in the past. For example, since 1975 milk production has increased linearly at the rate of ~ 180 kg milk per cow /yr while milk cow herd population decreased at the rate of ~69,000 head per yr, i.e. the 60% greater amount of milk produced in 2006 compared to 1970 required 25% fewer cows. Animal inventories declined by 10% for beef brood cows from 36 million head in 1970 to 33 million head in 2006, and the inventory of breeder pigs and market hogs declined 8% from 673 million head to 625 million head in the same period, even with similar or greater annual meat production. These trends resulted from greater growth rates of animals producing more meat in a shorter amount of time. In 1970, broilers were slaughtered after 80 days on feed at 1.7 kg live weight, but by 2006 the average weight was 2.5 kg after only 44 days on feed. These trends are in requiring fewer animals to produce more animal food products through improved diet and increased production efficiency will continue.

Implementation of improved methods of livestock management and manure handling and treatment to decrease NH_3 emissions that have been developed since 1990 and will further decrease ammonia and other gases and odor emissions. For example, sawdust litter helps decrease NH_3 emissions from pig manure with 44-74% of manure N converted to N_2. Storage covers for slurry storage tanks, anaerobic lagoons, and earthen slurry pits decrease emissions from those containments. Anaerobic digestion in closed containment has been studied for many types of applications. Recent research demonstrates reduction in NH_3 emissions after a permeable cover was installed, e.g a polyethylene cover decreased NH_3 emissions by ~80%. A well managed swine lagoon can denitrify approximately 50% of the excreted N to N_2. Recently engineered developments utilizing closed loop systems (Aneja et al. 2008) substantially reduce atmospheric emissions of ammonia (> 95%) and odor at hog facilities.

<u>Table 27: Estimates for potential decreases in NH_3 emissions from livestock manure in the U.S. (estimate is based on livestock emissions of 1.6 Tg from Table 1).</u>

NH_3 Source	% of Total NH_3	Tg NH_3-N/yr emitted	Estimated Decrease of NH_3 %	Estimated Decrease of NH_3 Tg N/yr
Dairy	23.1	0.37	10	0.040
Beef	27.1	0.44	10	0.040
Poultry	27.5	0.44	50	0.220
Swine	17.5	0.28	50	0.140
Goat/sheep	1.6	0.03	10	0.003
Horse	2.9	0.05	10	0.005
Total	**100.0**	**1.61**		**0.450**

Target recommendation 3. *INC recommends that excess flows of Nr into streams, rivers, and coastal systems be decreased by approximately 20%* ***(~1 Tg N/yr)*** *through improved landscape management and without undue disruption to agricultural production. This would include activities such as using large-scale wetland creation and restoration to provide needed ecosystem services of Nr retention and conversion as well as matching cropping systems and intensity of Nr use to land characteristics.. Improved tile-drainage systems and riparian buffers on crop land, and implementing storm water and non-point source management practices (e.g., EPA permitting and funding programs) are other alternatives that are less proven. In addition, the committee recommends that crop N-uptake efficiencies be increased by up to 25% over current practices through a combination of knowledge-based practices and advances in fertilizer technology (such as controlled release and inhibition of nitrification). Crop output can be increased while decreasing total Nr by up to 20% of applied artificial Nr, amounting to ~**2.4 Tg N/yr** below current amounts of Nr additions to the environment. These are appropriate targets with today's available technologies and further progress is possible.*

<u>Target Goal 4.</u> Discharge of Nr from developed lands and point sources

National loadings of Nr to the environment from public and private wastewater point sources are relatively modest in comparison with global Nr releases, however they can be important local sources with associated impacts. Further, in the last decade there has been increasing point source permitting requirements for storm water under the NPDES program focusing in on uban runoff, industrial sites and new construction. While the stormwater permits cover much of the developed land runoff, there are also developed areas that do not exceed threshold conditions for the NPDES program and, although functionally indistinguishable from permitted stormwater runoff, runoff from those areas is still considered "nonpoint source" according to the CWA.

There are two funding sources of significance authorized in the CWA that are used to fund projects relevant to the control of Nr. Section 319 establishes state nonpoint source management programs to plan for and implement management measures that abate sources of nonpoint pollution from eight source categories, including both urban and agricultural sources; however, the CWA disallows use of 319 funds for NPDES permit requirements, so urban areas with stormwater permits do not qualify for Section 319 funding. Over the years section 319 has made available, through 60% matching funds, over $1.6 billion in assistance. The much larger source of funding comes under Title VI of the CWA, which has provided over $24 billion (federal) for the construction of treatment facilities for point sources of wastewater over the past twenty years, although only a fraction of this amount has been dedicated to denitrification processes. Title VI "state revolving" load funds can be used for stormwater management, as well as other water pollution management activities, but not all states have chosen to use funds beyond traditional sewage treatment plant infrastructure needs because of the large backlog of demand for those purposes.

Target Recommendation 4. *INC recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA. This will decrease Nr emissions by between* ***0.5 and 0.8 Tg N/yr****.*

3.4.3 Summary statement

The committee's recommendations, if implemented, would reduce total Nr loadings to the environment in the U.S. by approximately 25% below current levels. The committee believes that these represent realistic near-term targets based on current technology, however further reductions are needed for many N-sensitive ecosystems and to ensure that health-related standards are maintained. Achieving and going beyond these recommended Nr reduction targets are critical given the growing demand for food- and fiber-production and energy use from population pressure and economic growth.

Appendix 1: Key to chemical abbreviations

AFO – Animal feeding operations

C - Carbon

CFC – Chlorofluorocarbon

DIN – Dissolved inorganic nitrogen

DO – Dissolved Oxygen

Fe - Iron

HNO_3 – Nitric Acid

HONO –Nitrous Acid

N – Nitrogen

N_2 –Diatomic nitrogen

N_2O – Nitrous oxide,

N_2O_5 – Dinitrogen Pentoxide

NH_3 – Ammonia

NH_4^+ – Ammonium

NH_x – $NH_3 + NH_4^+$

NO – Nitric Oxide

NO_2 – Nitrogen Dioxide

NO_3^- — Nitrate ion

NO_3 – Nitrate radical,

Norg – Organic Nitrogen

NO_x – Nitrogen Oxides ($NO + NO_2$)

NO_y– (NO, NO_2, NO_3, N_2O_5, HONO, HNO_3, NO_3^-, PAN and other organo-nitrates, $RONO_2$)

Nr – Reactive Nitrogen

O_2 – Oxygen

OH – Hydroxyl radical

P – Phosphorus

PAN– Polyacrylonitrile

PM – Particulate Matter

$PM_{2.5}$ – Particulate Matter less than 2.5 microns in diameter

PM_{10} – Particulate Matter less than 10 microns in diameter

$RONO_2$ – Organic Nitrates

Si – Silicon

SO_2 – Sulfur dioxide

SO_4^{2-} – Sulfate

TAN – Total ammonical nitrogen

Appendix 2: Acronyms and abbreviations

AAPFCO – Association of American Plant Food Control Officials)

AFO – Animal Feeding Operation

AIRMON – Atmospheric and Integrated Research Monitoring Network

AOB – Ammonia Oxidizing Bacteria

BL – Boundary layer

BMP – Best Management Practice

BNF– Biological Nitrogen Fixation

BNR – Biological Nutrient (or Nitrogen) Removal

CAA – Clean Air Act

CAFO – Concentrated Animal Feeding Operation

CAIR – Clean Air Interstate Rule (

CALM – Consolidated Assessment and Listing Methodology

CAST – Council for Agricultural Science and Technology

CASTNET – Clean Air Standards and Trends Network

C-BNF – Cultivation-induced biological nitrogen fixation ()

CFC – Chlorofluorocarbon

CFR – Code of Federal Regulations

CL – Critical load

CLAD – Critical Loads Ad-Hoc Committee

CMAQ – Community Multiscale Air Quality

DRP – Conservation Reserve Program

CSO – Combined sewer overflow

CTM – Chemical Transport Models

CWA – Clean Water Act

DIN – Dissolved Inorganic Nitrogen

DO – Dissolved Oxygen

DOE – U.S. Department of Energy

DOT – U.S. Department of Transportation

ECU – Electricity generating units

EFD – Essential Facilities Doctrine

EGR – Exhaust gas recirculation

EISA – Energy Independence and Security Act

EPA – United States Environmental Protection Agency

EQIP - Environmental Quality Incentives

EU – European Union

FAO – Food and Agricultural Organization of the United Nations

FAOSTAT – Food and Agricultural Organization Statistical Database

FGR – Flue-gas recirculation (FGR)

ha – Hectare

GHG – Greenhouse Gas

GPS – Geographic Positioning System

HAB – Harmful Algal Bloom

IPCC – Intergovernmental Panel on Climate Change

ISA – Integrated Science Assessments

ITQ - Individual Transferable Quota

kg – Kilogram

L - Liter

LA – Load Allocation

LISS – Long Island Sound Study

mg - Milligrams

MGD – Million Gallons per Day

Mmt – Million metric tons

MT - metric tons

MOM – Mississippi-Ohio-Missouri

MRB – Mississippi River Basin

MS4 – Municipal Separate Storm Sewer System

NAAQS – National Ambient Air Quality Standards

NADP – National Atmospheric Deposition Program

NASS – National Agricultural Statistics Service Information

NCA – National Coastal Assessment

NCE – Nitrogen Credit Exchange

NCCR – National Coastal Condition Report

NEEA – National Estuarine Eutrophication Assessment

NESCAUM – Northeast States for Coordinated Air Use Management

NFUE - Nitrogen Fertilizer Use Efficiency

NMP – Nutrient Management Plan

NOAA – National Oceanic and Atmospheric Administration

NPS – Nonpoint Source

NRC – National Research Council

NRCS – Natural Resources Conservation Service

NRD – Natural Resource District

NRI –National Resources Inventory

NTN – National Trends Network

NUE – Nitrogen Use Efficiency. NUE is defined as the kg grain produced per kg of total N used by the crop, where total N includes N from fertilizer, biological N fixation and soil organic matter mineralization

OTAG – Ozone Transport Assessment Group

OTC – Ozone Transport Commission

PE – Physiological Efficiency with which the N taken up by the crop is used to produce economic yield such as grain or fruit

PFP – Partial Factor Productivity

POTW – Publicly Owned Treatment Works

PSD – Prevention of Significant Deterioration

RE –Recovery Efficiency (kg N uptake per kg N applied)]

SAV – Submerged Aquatic Vegetation

SNCR – Selective non-catalytic

SCR – Selective Catalytic Reduction

SIP – State Implementation Plan

SOM – Soil organic matter

SPAtially Referenced Regressions On Watershed Attributes Model – SPARROW

STP – Sewage Treatment Plant

SW – Storm Water

SWAT – Storm Water Assessment Tool

SWPPP – Stormwater Pollution Prevention Plan

T – Temperature

Tg – Teragram (million metric tons or 10^{12} grams)

TMDL – Total Maximum Daily Load

UFTRS – Uniform Fertilizer Tonnage Reporting System

UNECE – United Nations Economic Commission for Europe

US – United States of America

USDA – U.S. Department of Agriculture

USGS – U.S. Geological Survey

USEPA – United States Environmental Protection Agency

WHO – World Health Organization

WLA – Wasteload Allocation

WPCA – Water Pollution Control Authorities

WRI – World Resources Institute

WRP – Wetland Reserve Program

WSA – Wadeable Stream Assessment

Appendix 3: Findings and Recommendations of the Integrated Nitrogen Committee

1. Introduction

This appendix contains a compilation of all the Findings and Recommendations of the Integrated Nitrogen Committee. Following a listing of the four overarching recommendations, the more specific recommendations are listed with appropriate section headings.

2. Overarching Recommendations

Recommendation A

An integrated approach to the management of Nr will likely use a combination of these four implementation mechanisms. Each mechanism must be appropriate to the nature of the problem at hand, supported by critical research on decreasing the risks of excess Nr, and reflect an integrated policy that recognizes the complexities and tradeoffs associated with the nitrogen cascade. Management efforts at one point in the cascade may be more efficient and cost effective than control or intervention at another point. This is why understanding the nature and dynamics of the N cascade is critically important.

Recommendation B

EPA should form an Intra-agency Nr Management Task Force that will build on existing Nr research and management capabilities within the Agency. This Intra-Agency Task Force should be aimed at increasing scientific understanding of: 1) Nr impacts on terrestrial and aquatic ecosystems, human health, and climate,2) Nr-relevant monitoring requirements, and 3) the most efficient and cost-effective means by which to decrease various adverse impacts of Nr loads as they cascade through the environment.

Recommendation C

EPA should join with other agencies within the US government in establishing an Inter-agency Nr Management Task Force. The members of this Inter-Agency Task Force should include at least the following federal agencies: US Department of Agriculture (USDA), US Department of Energy (DOE), US Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), US Geological Survey (USGS), US Forest Service (USFS,) and Federal Emergency Management Agency (FEMA)). This Task Force should coordinate federal programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and managing Nr in the environment.

The intra- and inter-agency Nr-Management Task Forces should take a systems approach to research, monitoring, and evaluation in the following areas to inform public policy related to Nr management:

- Development of methods to help implement a systems approach
  - developing and evaluating proposed Nr budgets
  - developing appropriate life cycle accounting methods
  - developing monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads
  - evaluating the critical loads approach to air and water quality management
  - developing Nr indicators for excess Nr effects on human health and environment
  - developing new systems-based approaches for controlling Nr releases to the environment
- Enhancing ecosystem services that lead to the denitrification of Nr in the landscape including reconnecting rivers and streams to their floodplain, creation and restoration of wetlands in agricultural landscapes, and stream and ditch enhancements that increase the surface area of potential denitrification.
- Best management practices (BMPs)
  - developing the scientific understanding required for identifying best management practices (BMPs) for specific application, including:
    - Nr applications in agriculture to ensure adequate food, feed, fiber, and bioenergy feedstock supply while also avoiding negative impacts on the environment and human health;
    - Nr applications for developed (e.g., residential and commercial) runoff mitigation and landscape maintenance;
    - planning and pollution prevention including low impact development and natural ecosystem service preservation;
    - Enhancing the appropriate matching of crops, cropping systems, and land types and capabilities for the most productive use of Nr and the reduction of excess Nr
    - development and natural ecosystem service preservation;
    - primary use of natural land features and attributes, such as wetland preservation and enhancement, natural soil profiles and buffer strips;
    - improved removal of Nr from sewage waste streams at both large-scale wastewater treatment facilities and individual subsurface (septic) systems

  - establishing proactive extension and technology transfer approaches to facilitate adoption of BMPs

- Assessment activities
  - assessing combined carbon (C) and Nr effects on terrestrial and aquatic ecosystems
  - assessing indicators/endpoints, costs, benefits and risks associated with impairment of human health and decline and restoration of ecosystem services
  - reviewing existing and proposed legislation for purposes of extending Nr regulatory authority or streamlining procedures for enacting Nr risk reduction strategies
  - evaluating economic incentives, particularly those that integrate air, aquatic, and land sources of excess Nr
- Developing new education, outreach, and communication initiatives

3. Near-term target recommendations

In addition, INC makes four recommendations that set near-term targets for the decrease of Nr entering the environment from various sources.

1. INC recommends that the EPA expand its NO_x control efforts from the current decreases of emissions of light duty vehicles (including passenger cars) and power plants to include other important unregulated mobile and stationary sources (e.g., off road vehicles) sufficient to achieve a **2.0 Tg N/yr** decrease in the generation of reactive nitrogen.
2. INC recommends a goal of decreasing livestock-derived NH_3 emissions by 30% (a decrease of **0.5 Tg N/yr**) by a combination of BMPs and engineered solutions. This is expected to decrease $PM_{2.5}$ by ~0.3 $\mu g/m^3$ (2.5%), and improve health of ecosystems by achieving progress towards critical load recommendations. Additionally we recommend decreasing NH_3 emissions derived from fertilizer applications by 20% (decrease by **~0.2 Tg N/yr**), through the use of NH_3 treatment systems and BMPs.
3. INC recommends that excess flows of Nr into streams, rivers, and coastal systems be decreased by approximately 20% **(~1 Tg N/yr)** through improved landscape management and without undue disruption to agricultural production. This would include activities such as using large-scale wetland creation and restoration to provide needed ecosystem services of Nr retention and conversion

as well as matching cropping systems and intensity of Nr use to land characteristics.. Improved tile-drainage systems and riparian buffers on crop land, and implementing storm water and non-point source management practices (e.g., EPA permitting and funding programs) are other alternatives that are less proven. In addition, the committee recommends that crop N-uptake efficiencies be increased by up to 25% over current practices through a combination of knowledge-based practices and advances in fertilizer technology (such as controlled release and inhibition of nitrification). Crop output can be increased while decreasing total Nr by up to 20% of applied artificial Nr, amounting to ~**2.4 Tg N/yr** below current amounts of Nr additions to the environment. These are appropriate targets with today's available technologies and further progress is possible.

4. INC recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA. This will decrease Nr emissions by between **0.5 and 0.8 Tg N/yr**.

Implementing these recommendations would decrease the introduction of Nr into the U.S. by about 25%, which would then decrease the amount of Nr lost to the atmosphere, soils and waters.

4. Specific findings and recommendations

Finding 21

Crop agriculture receives 63% of US annual new Nr inputs from anthropogenic sources (9.8 Tg from N fertilizer, 7.7 from crop BNF versus 29 Tg total) and accounts for 58% (7.6 Tg, see Table 1) of total US Nr transfers from terrestrial systems to air and aquatic ecosystems, yet current monitoring of fertilizer use statistics by federal agencies is inadequate to accurately track trends in quantities and fate of N applied to major crops and the geospatial pattern by major watersheds.

Recommendation 1: *Increase the specificity and regularity of data acquisition for fertilizer application to major agricultural crops in terms of timing and at a sufficiently small application scale (and also for urban residential and recreational turf) by county (or watershed) to better inform decision-making about policies and mitigation options for reducing Nr load in these systems, and to facilitate monitoring and evaluation of impact from implemented policies and mitigation efforts.*

Finding 22

Nr inputs to crop systems are critical to sustain crop productivity and soil quality. Moreover, given limited land and water resources, global population growth and rapid economic development in the world's most populous countries, the challenge is to accelerate increases in crop yields on existing farm land while also achieving a substantial increase in N fertilizer uptake efficiency. This process is called "ecological intensification" because it recognizes the need to meet future food, feed, fiber and energy demand of a growing human population while also protecting environmental quality and ecosystem services for future generations (Cassman, 1999). More diverse cropping systems with decreased Nr fertilizer input may also provide an option on a large scale if the decrease in Nr losses per unit of crop production in these diverse systems can be achieved without a decrease in total food production, which would trigger indirect land use change to replace the lost production and negate the benefits.

Recommendation 2:

a) *Data on NFUE and N mass balance, based on direct measurements from production-scale fields, are required for the major crops to identify which cropping systems and regions are of greatest concern with regard to mitigation of Nr load and to better focus research investments, policy development, and prioritization of risk mitigation strategies.*
b) *Promote efforts at USDA and land grant universities to: (i) investigate means to increase the rate of gain in crop yields on existing farm land while increasing N fertilizer uptake efficiency and (ii) explore the potential for more diverse cropping systems with lower N fertilizer input requirements so long as large-scale adoption of such systems would not cause indirect land use change.*
c) *EPA should work closely with the US Department of Agriculture (USDA), Department of Energy (DOE), and the National Science Foundation (NSF), and land grant universities to help identify research and education priorities efficient use and mitigation of Nr applied to agricultural systems.*

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Finding 23

Nitrous oxide emissions from the Nr inputs to cropland from fertilizer, manure, and legume fixation represent a large proportion of agriculture's contribution to greenhouse gas emissions, and the importance of this source of anthropogenic greenhouse gas will likely increase unless NFUE is markedly improved in crop production systems. Despite its importance, there is considerable uncertainty in the estimates of nitrous oxide emissions from fertilizer and research should focus on reducing this uncertainty.

Recommendation 3: *The committee recommends that EPA ensure that the uncertainty in estimates of nitrous oxide emissions from crop agriculture be greatly reduced through the conduct of EPA research and through coordination of research efforts more generally with other agencies such as USDA, DOE, NSF and with research conducted at universities.*

Finding 24

Rapid expansion of biofuel production is changing the cost-benefit ratio of N fertilizer use in crop production and also changing the nutrient profile of livestock diets with consequences for effective management of Nr.

Recommendation 4: Rapid expansion of biofuel production has the potential to increase N fertilizer use through expansion of corn production area and associated N fertilizer inputs, and from extending cultivation of cellulosic materials that will also need N inputs. Distiller's grains are changing animal diets and affecting N recycling in livestock. Both have important consequences for the effective future management of Nr.

Finding 25

There are no nationwide monitoring networks in the US to quantify agricultural emissions of greenhouse gases, NO, N_2O, reduced sulfur compounds, VOCs, and NH_3. In contrast there is a large network in place to assess the changes in the chemical climate of the US associated with fossil fuel energy production, ie the National Atmospheric Deposition Program/National Trends Network (NADP/NTN), which has been monitoring the wet deposition of sulfate (SO_4^{2-}), NO_3^-, and NH_4^+ since 1978.

Recommendation 5: *The status and trends of gases and particulate matter emitted from agricultural emissions, e.g.,* NO_3^- *and* NH_4^+ *should be monitored and assessed utilizing a nationwide network of monitoring stations.*

Finding 26

Farm-level improvements in manure management can substantially reduce Nr load and transfer. There are currently no incentives or regulations to decrease these transfer and loads despite the existence of management options to mitigate.

Recommendation 6: *Policy, regulatory, and incentive framework is needed to improve manure management to reduce Nr load and ammonia transfer, taking into account phosphorus load issues.*

Finding 7

Synthetic N fertilizer application to urban gardens and lawns amounts to approximately 10% of the total annual synthetic N fertilizer used in the U.S. Even though this N is a large part of N fertilizer used little attention is paid to how efficiently it is used.

Recommendation 7a: *To ensure that urban fertilizer is used as efficiently as possible, the committee recommends that EPA work with other agencies such as USDA as well as state and local extension organizations to coordinate research and promote awareness of the issue.*

Recommendation 7b: *Through outreach and education, supported by research, improved turf management practices should be promoted, including improved fertilizer application and formulation technologies and maintenance techniques that minimize supplemental Nr needs and losses, use of alternative turf varieties that require less fertilization, alternative ground covers in place of turf, and use of naturalistic landscaping that focuses on native species.*

Finding 27

Scientific uncertainty about the origins, transport, chemistry, sinks, and export of Nr remains high, but evidence is strong that atmospheric deposition of Nr to the Earth's surface as well as emissions from the surface to the atmosphere contribute substantially to environmental and health problems. Nitrogen dioxide, NO_2, is often a small component of NOy, the total of oxidized nitrogen in the atmosphere. The current NAAQS for NO_2, as an indicator of the criteria pollutant "oxides of nitrogen," is inadequate to protect health and welfare. Serious consideration should be given to replacing or supplementing the NO_2 measurements and standard with NOy. Atmospheric emissions and concentrations of Nr from agricultural practices (primarily in the form of NH_3) have not been well monitored, but NH_4^+ ion concentration and wet deposition (as determined by NADP and NTN) appear to be increasing. This suggests that NH_4^+ emissions are increasing. Both wet and dry deposition contribute substantially to NH_x removal, but only wet deposition is known with much scientific certainty. Thus consideration should be given to adding these chemically reduced and organic forms of Nr to the list of Criteria Pollutants.

Recommendation 8a. *Increase the scope and spatial coverage of the Nr concentration and flux monitoring networks (such as the National Atmospheric Deposition Program and the Clean Air Status and Trends Network) and appoint an oversight review panel for these two networks.*

Recommendation 8b. *Monitor NH_3, NH_x, NO_y, NO_2, NO, and PAN concentrations, measure or infer deposition, and support the development of new measurement and monitoring methods.*

Recommendation 8c. *Measure deposition directly both at the CASTNET sites and in nearby locations with non-uniform surfaces such as forest edges.*

Recommendation 8d. *EPA should continue and support research into convective venting of the Planetary Boundary Layer and long range transport.*

Recommendation 8e. *Develop and support analytical techniques and observations of atmospheric organic N compounds in vapor, particulate, and aqueous phases.*

Recommendation 8f. *Increase the quality and spatial coverage of measurements of the NH_3 flux to the atmosphere from major sources especially agricultural practices.*

Recommendation 8g. *Improve numerical models of NOy and NHx especially with regard to chemical transformations, surface deposition, and off shore export; develop linked ocean-land-atmosphere models of Nr.*

Finding 28

Although total N budgets within all terrestrial systems are highly uncertain, Nr transfers from grasslands and forests (vegetated) and urban (populated) portions of the N Cascade appear to be higher, on a per cent of input basis, than from agricultural lands. The relative amount of these transfers ascribed to leaching, runoff and denitrification, are as uncertain as the N budgets themselves.

Recommendation 9: *EPA should join with USDA, DOE, and universities should work together in efforts to ensure that the N budgets of terrestrial systems are properly quantified and that the magnitudes of at least the major transfer vectors are known.*

Finding 29

Over the past 25 years, there has been a growing recognition of eutrophication as a serious problem in coastal estuaries (NRC, 2000). The last comprehensive national NCCR was published in 2004 (EPA, 2004) included an overall rating of “fair” for estuaries, including the Great Lakes, based on evaluation of over 2000 sites. The water quality index, which incorporates nutrient effects primarily as chlorophyll-a and dissolved oxygen impacts, was also rated “fair” nationally. Forty percent of the sites were rated “good” for overall water quality, while 11% were “poor” and 49% “fair”.

Recommendation 10: *The committee recommends that EPA consider a range of scales reflecting ecosystem, watershed, and regional levels that include all inputs, e.g. atmospheric and riverine, of marine eutrophication dynamics and management.*

Finding 30

Denitrification of Nr in terrestrial and aquatic systems is one of the most uncertain parts of the nitrogen cycle. Denitrification is generally considered to be a dominant N loss pathway in both terrestrial and aquatic systems, but it is poorly quantified

Recommendation 11: *EPA, USDA, DOE, and universities should work together to ensure that denitrification in soils and aquatic systems is properly quantified, by funding appropriate research.*

Finding 31

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. Since 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 12. *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 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, with parallel interests in advancing the necessary underlying science of Nr creation, transport and transformation, impacts, and management.*

Finding 32

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 13. *It is, therefore, recommended that the EPA 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.*

Finding 33

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.

Recommendation 14. *To better address Nr runoff and discharges from the peopled landscape the committee recommends that EPA*:

14a. *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 with special attention to the need to regulate nonpoint source and related land use practices.*

14b. *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.*

14c. *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.*

14d. *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.*

14e. *Use ecosystem-based management approaches that balance natural and anthropogenic needs and presence in the landscape.*

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Finding 34

Meeting Nr management goals for estuaries, when a balance should be struck between economic, societal and environmental needs, under current federal law seems unlikely. Enforceable authorities over nonpoint source, stormwater, air (in terms of critical loads), and land use are not adequate to support necessary Nr controls. Funding programs are presently inadequate to meet existing pollution control needs. Furthermore, new technologies and management approaches are required to meet ambitious Nr control needs aimed at restoring national water quality.

Recommendation 15. *INC recommends that EPA reevaluate water quality management approaches to ensure Nr management goals are attainable, enforceable, and affordable and that monitoring and research are adequate to problem definition and resolution, particularly in the development of nitrogen removal technologies. This may require changes in the way EPA sets water quality criteria and some compromises in ecosystem goals to accommodate human uses of the air, land and water.*

Finding 35

The committee has determined that an integrated approach to monitoring that includes multimedia (air, land and water) components and considers a suite of environmental and human concerns (e.g., Nr effects, climate change, human health) would be most useful and efficient. Some of the phenomena that we present in this report simply need more definition and verification but, more importantly, as control is brought to bear on Nr, improvements need to be measured (i.e. monitored) to validate the success of one control or another. If the desired improvements are not realized as shown by the collected data, corrective measures will be required. The pool of data would be used to formulate new management procedures. The process of monitoring and control revisions is termed adaptive management—a process that INC supports as it does not delay actions that can be taken immediately, but acknowledges the likelihood that management programs will be altered (adapted) as scientific and management understanding improve**.**

Recommendation 16. *The committee recommends that EPA initiate discussions and take action to develop a national, multimedia monitoring program that monitors sources, transport and transition, effects using indicators where possible, and sinks of Nr in keeping with the nitrogen cascade concept. This comprehensive program should build upon existing EPA and state initiatives as well as monitoring networks already underway in other federal agencies such as the U.S. Geological Survey programs and the NADP effort.*

Finding 36

In this connection, the INC strongly commends EPA for its recently increased willingness to think more holistically – and in more fully integrated ways – about both the policy-relevant science and the practical arts of air quality management aimed at protection of both aquatic and terrestrial ecosystems from adverse effects of Nr. These shifts in both emphasis and approach have included:

1) Increased emphasis in the NAAQS review processes on scientific questions that are as directly relevant as possible to well-defined policy questions of concern to EPA;
2) More frequent discussion about both public-welfare and public-health impacts of mixtures of air pollutants;
3) More frequent discussion about the critical loads concept as an alternative or complement to the more familiar NAAQS Standards;
4) Separation of the preparation and review of documentation for a Secondary (public-welfare-based) NAAQS from the (previously always dominating) Primary (public-health-based) NAAQS review processes;

5) The decision by the Science Advisory Board of EPA to establish this special Integrated Nitrogen Committee (INC); and
6) The unprecedented decision to undertake an integrated (simultaneous) review/ /of the Secondary NAAQS for two Criteria Pollutants at the same time [Oxides of Nitrogen (NO_x) and Oxides of Sulfur (SO_x)].

Especially notable evidence for EPA's "increased willingness to think more holistically – and in more fully integrated ways" is the following statement of Conclusion in the Executive Summary of the December 2008 Integrated Science Assessment for Oxides of Nitrogen and Sulfur (EPA, 2008):

> The main effects of N and S pollution assessed in the ISA are acidification, N enrichment, and Hg methylation. Acidification of ecosystems is driven primarily by deposition resulting from SO_x, NO_x, and NH_x pollution. Acidification from the deposition resulting from current emission levels causes a cascade of effects that harm susceptible aquatic and terrestrial ecosystems, including slower growth and injury to forests and localized extinction of fishes and other aquatic species. In addition to acidification, atmospheric deposition of reactive N resulting from current NO_x and NH_x emissions along with other non-atmospheric sources (e.g., fertilizers and wastewater), causes a suite of ecological changes within sensitive ecosystems. These include increased primary productivity in most N-limited ecosystems, biodiversity losses, changes in C cycling, and eutrophication and harmful algal blooms in freshwater, estuarine, and ocean ecosystems.

In addition, 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 17. *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.*

<u>Finding 37</u>

Current EPA policy (EPA 2007e) discourages states from controlling ammonia emissions as part of their plan for reducing $PM_{2.5}$ concentrations. Ammonia is a substantial component of $PM_{2.5}$ in most polluted areas of the U.S. 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 US. Lower NH_3 emissions will lower $PM_{2.5}$ concentrations. Such reductions in $PM_{2.5}$ concentrations have been linked to reductions in morbidity and mortality.

Recommendation 18. *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.*

Finding 38

The committee notes that the effective management of Nr in the environment must recognize the existence of tradeoffs across impact categories involving the cycling of other elements, particularly C.

Recommendation 19. *The committee recommends that the integrated strategies for Nr management outlined in this report be developed in cognizance of these interrelations and tradeoffs.*

Finding 39

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 20: *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 INC 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 or 2100.*
2. *Down-scale statistics or nest regional climate models within each of these global scenarios to generate meteorological and chemical fields (e.g., T, RH, winds, precipitation, CO_2) for a few years around 2050 and 2100.*
3. *Run several independent biogeochemical Nr models (Earth System models that include air/water/land) for N 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.*

Appendix 4: Technical Annexes

A. Production of N_2 and N_2O via gas-phase reactions

Atmospheric conversion of NO_x and NH_x to less reactive N_2 or N_2O appears to play a minor role in the global N budget, but currently is not well quantified. The gas-phase reactions in the troposphere that convert NH_3 and NO_x to N_2 and N_2O, start with attack of NH_3 by OH:

$$NH_3 + OH\cdot \rightarrow NH_2\cdot + H_2O \quad (1)$$

Several potentially interesting fates await the NH_2 radical:

$$NH_2\cdot + O_3 \rightarrow NH, NHO, NO \quad (2)$$

$$NH_2\cdot + NO_2 \rightarrow N_2O + H_2O \quad (3)$$

$$NH_2\cdot + NO \rightarrow N_2 + H_2O \quad (4)$$

$$k_{O3} = 1.9x10^{-13}\ cm^3\ s^{-1}$$

$$k_{NO2} = 1.8x10^{-11}\ cm^3\ s^{-1}$$

$$k_{NO} = 1.8x10^{-11}\ cm^3\ s^{-1}$$

The first step, attack by OH, is slow. The rate constant for the Reaction 1 is 1.6 x 10^{-13} $cm^3\ s^{-1}$ and the lifetime of NH_3 for a typical concentration of 10^6 OH cm^{-3} is about 70 d. In most areas of the world where concentrations of NH_3 are high, concentrations of sulfates are also high, and NH_3 is removed by conversion to condensed phase ammonium sulfate or bisulfate on time scales much faster than 70 d. The mean lifetime of these aerosols with respect to wet deposition is about 10 d.

There are some areas of the world, notably California and South Asia, where NH_3 and NO_x are emitted in large quantities, but SO_2 is not, and there gas-phase conversion can take place. In general, $[O_3] >> [NO_x]$, and Reaction 2 represents an unimportant source of NOx, but Reactions 3 and 4 may be atmospherically noteworthy. As an upper limit to current N_2O production, we can assume that each of these regions covers an area of 10^6

km^2 and that they contain ammonia at a concentration of 10 μg N m^{-3} in a layer 1000 m deep. The annual production of N_2 and/or N_2O would then be on the order of 0.1 Tg N, a minor but nontrivial contribution to denitrification and about 1% of the anthropogenic N_2O production. If NH_3-rich air is lofted out of the boundary layer into the upper troposphere where deposition is impeded, it will have an atmospheric residence time on the order of months, and the probability of reaction to form N_2O or N_2 becomes greater. This possibility has not been investigated extensively. It is also possible than Europe and North America will continue to reduce S emissions without reducing NH_3 emissions and the atmospheric source of N_2O will grow in importance.

In the stratosphere, N_2O photolysis leads to loss of Nr via

$$N_2O + h\mu \rightarrow N_2 + O \quad (5)$$

While reaction with an electronically excited oxygen atom $O(^1D)$ leads to production of NO via

$$N_2O + O(^1D) \rightarrow 2NO \qquad (6)$$

Photolysis (Reaction 5) dominates, but a large enough fraction of the N_2O reacts with $O(^1D)$ that this is the main source of NO_x in the stratosphere. The fate of this oxidized nitrogen (NOy) is transport back into the troposphere where it is removed by wet deposition. Downward transport of the odd N from the oxidation of N_2O is a minor (~1%) source of NO_y in the troposphere. Most of the N_2O released into the atmosphere is eventually converted to N_2 – the problem is that it destroys stratospheric ozone in the process.

In summary, our current understanding of the chemistry of atmospheric ammonia suggests that *in situ* conversion to N_2 and N_2O plays a minor (~1%) role in global N budgets, but if assumptions about kinetics or concentrations are in error these mechanisms could become important.

B. SPARROW Model for Estimating Watershed Nr

Estimates of Nr transfers in aquatic ecosystems are difficult to quantify at the national scale, given the need to extrapolate information from sparse monitoring data in specific watersheds to the geographic boundaries of the nation. One excellent tool for estimating

Nr loads at regional scales is the spatially referenced regression on watershed attributes (SPARROW) modeling technique. The SPARROW model has been employed to quantify nutrient delivery from point and diffuse sources to streams, lakes, and watershed outlets at the national scale (Smith et al. 1997). The model infrastructure operates in a geographic framework, making use of spatial data to describe sources of pollutants (e.g., atmospheric deposition, croplands, fertilizers) and characteristics of the landscape that affect pollutant transport (e.g., climate, topography, vegetation, soils, geology, and water routing). Though empirical in nature, the SPARROW modeling approach uses mechanistic formulations (e.g., surface-water flow paths, first-order loss functions), imposes mass balance constraints, and provides a formal parameter estimation structure to statistically estimate sources and fate of nutrients in terrestrial and aquatic ecosystems. The spatial referencing of stream monitoring stations, nutrient sources, and the climatic and hydrogeologic properties of watersheds to stream networks explicitly separates landscape and surface-water features in the model. This allows nutrient supply and attenuation to be tracked during water transport through streams and reservoirs, and accounts for nonlinear interactions between nutrient sources and watershed properties during transport. The model structure and supporting equations are described in detail elsewhere (Smith et al. 1997, Alexander et al. 2000, Alexander et al. 2008). Table 1 provides an estimate of contemporary Nr loading in surface waters of the US, representing long-term average hydrological conditions (over the past 3 decades). There are hot spots of high Nr yields to rivers associated with land use and watershed characteristics, and SPARROW allows considerations of the fate of these Nr inputs to streams and rivers as they flow downstream to coastal receiving waters (Alexander et al. 2008).

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