# 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 active, chemically reactive, and radiatively active nitrogen compounds[1]. At the global scale, human activities now create approximately two-fold more Nr than natural terrestrial ecosystems produce. Human activities include the production of Nr through the Haber-Bosch process that generates ammonia ($NH_3$) for synthetic fertilizer and industrial feedstock, the enhancement of biological nitrogen fixation (BNF) by crop cultivation (e.g., legumes), and the combustion of fossil fuels. The first two activities form Nr on purpose; the last one forms Nr as a byproduct, ~~by accident.~~

Anthropogenic creation of additional Nr results in large-scale impacts. The first and foremost, humans rely heavily on anthropogenic Nr creation for food production. For a number of reasons, however, essentially all of the Nr created by humans is lost to the environment where it circulates between and accumulates in environmental systems (e.g.,the atmosphere; aquatic and terrestrial ecosystems). Once lost to the environment, the Nr contributes to a number of adverse environmental effects, including photochemical smog, increased levels of nitrogen (N)-containing aerosols, decreased atmospheric visibility, acid deposition, coastal eutrophication, greenhouse effects, and stratospheric ozone depletion. These effects contribute to declines in human health (e.g., respiratory diseases) and ecosystem health (e.g., biodiversity loss). The effects are magnified because any one atom of Nr in the environment can contribute to each effect (positive and negative) in sequence, as the Nr moves through environmental reservoirs; this characteristic of Nr is termed the nitrogen cascade.

To assist EPA in its management of nitrogen-related issues, this SAB committee was charged with addressing the following objectives:

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

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[1] The term reactive nitrogen (Nr) as used in this paper includes all biologically active, chemically reactive,and radiatively active nitrogen (N) compounds in the atmosphere and biosphere of Earth. Thus, Nr includes inorganic reduced forms of N [e.g., ammonia ($NH_3$) and ammonium ($NH_4^+$)], inorganic oxidized forms [e.g., nitrogen oxides ($NO_x$), nitric acid ($HNO_3$), nitrous oxide ($N_2O$), and nitrate ion ($NO_3^-$), and organic compounds (e.g., urea, amines, and proteins), by contrast to unreactive N2 gas.

2. Evaluate the contribution an integrated nitrogen management strategy[2] 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.

The rest of this summary gives an overview of the actions taken towards achieving the INC's charges. It then summarizes the Nr inputs to the United States (US), and the fate of the Nr in the US. The chapter then addresses how impacts are, and could be, assessed, and then concludes with the four overarching recommendations for both research and management that should be followed to help the EPA develop an integrated nitrogen management strategy, and five specific recommendations that will decrease by 25% the amount of Nr lost to the US environment.

The Integrated Nitrogen Committee (INC) of EPA's Science Advisory Board worked for two years gathering material on Nr inputs into the US, the flows of Nr through US environmental systems, and the influence that human activities have had on those flows. The INC also summarized the extensive literature that exists on the consequences on the atmosphere, terrestrial ecosystems and aquatic ecosystems. This very necessary material provided the context for INC to address the following fundamental questions:

- How should Nr be managed and how should management priorities be determined?
- To what degree can the US decrease Nr losses to the environment using existing technology?
- What further decreases are needed?
- What can EPA do to more develop an integrated management strategy for Nr?

In the past, other studies have examined the biogeochemical flows of Nr in the US, the impacts of alterations in those flows, and developed policies for addressing specific consequences of those alterations. What is different about this report is that it is the first time that this has been attempted in an integrated manner. Much more needs to be done in this regard and EPA has a powerful role to play, as is outlined in the report's findings and recommendations.

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[2] An integrated nitrogen management strategy takes an 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 reactive nitrogen, while limiting overall adverse effects.

## Overview

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

To aid the committee in its deliberations, four initial charges were identified. This section of the Executive Summary contains an overview of what was accomplished under each of the charges.

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

To address this aspect of the charge, INC first determined the major sources of newly created Nr for the US. Then the committee determined examined the flows of Nr within the food, fiber, feed and biofuel production systems for the US, paying special attention to identifying locations in the food production system where Nr is lost to the environment. The committee did the same thing for energy production, but since all the Nr formed during this process is lost to the environment, the committee identified the important energy producing sectors that contributed to Nr formation (Section 2.2).

The committee then examined the fate of the Nr lost to the environment, determined the amount stored in different systems (e.g., forest soils) and tracked the Nr as it was transferred from one environmental system (e.g., the atmosphere) to another (e.g., terrestrial ecosystems) (Section 2.3). The product of these two activities is summarized in Figures 1 and 2.

These two activities set the stage for addressing the problems N presents in the environment and the links among them. Using the nitrogen cascade, the committee identified the impacts that Nr has on people and ecosystems as it moves through environmental 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). Section 2.4 of the report covers this aspect of the first charge.

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

An integrated management strategy takes into account the contributions of all Nr sources, and all Nr chemical species, to negative impacts on humans and ecosystems. Further, the strategy should ensure that solving one problem related to Nr does not exacerbate another problem or decrease the ability to produce food. In short, the strategy should seek to maximize the benefits of Nr, while limiting overall adverse effects.

To address this aspect of the charge, the committee identified several examples of where actions in managing Nr in one environmental system caused unintended consequences in another. In addition, it provided examples of types of management actions that could be taken that would be ‘integrated’ in nature.

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

INC has identified four major actions which, collectively, would limit new Nr by about 25%, recognizing that closing off Nr at these points will result in further declines in Nr-related impacts throughout the nitrogen cascade. In addition, INC has suggested several ways in which these goals could be attained that include conservation measures, additional regulatory steps, application of modern technologies, and end-of-pipe treatment. These are initial actions; others should be taken once the recommended actions in the report that focus on a better understanding of N dynamics and impacts in the US are completed.

Charge #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 each finding is one or more "Recommendations". There is a total of 28 findings with associated recommendations. They address specific issues identified through out the report, and include suggestions for action where EPA, together with other agencies, could decrease the amount of Nr lost to the US environment, using current technologies. In addition, there are four over-arching recommendations that, if implemented, would provide EPA with the framework in which to improve N research in the support of risk reduction.

The remaining sections of the executive summary cover the points made above in more detail.

**Behavior of reactive nitrogen in the environment: transfer and transformations, impacts and integrated risk reduction strategies**

*Sources of Nr*

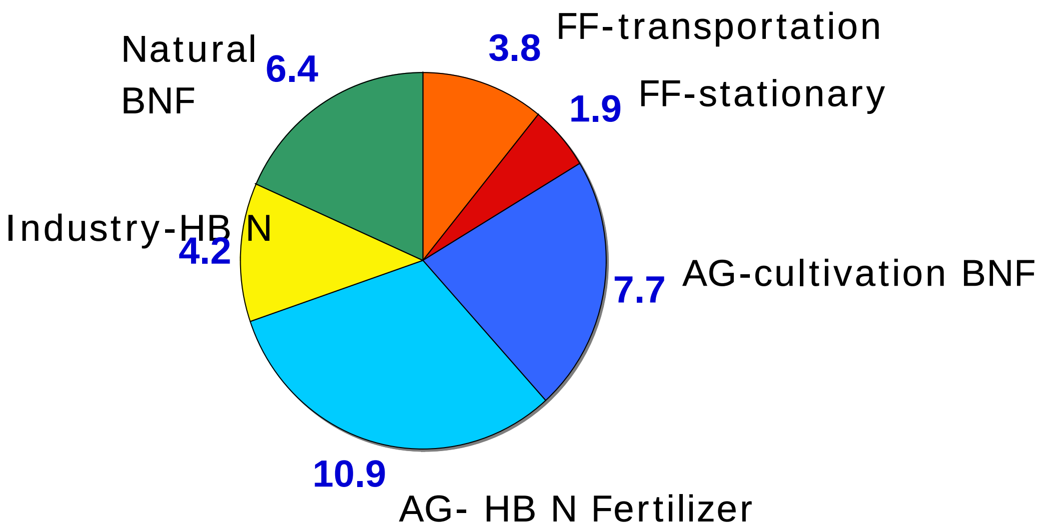
At the global scale, humans introduce approximately two-fold more Nr than do natural terrestrial processes; in the US it is approximately five-fold. Natural ecosystems in the US introduce about Nr measured as 6.4 teragrams (Tg) of nitrogen (N) per year (hereafter expressed as Tg N/yr). In contrast, human activity results in the introduction of about 29 Tg N/yr (Figure 1).

The first major source of Nr to the US is the Haber-Bosch process, which introduces 15.2 Tg N/yr into the US; 9.4 Tg N/yr from internal production and 5.8 Tg N/yr from imports. This total amount is used in three ways: 9.9 Tg N/yr is used to produce crops; 1.1 Tg N/yr is used to produce turf; and 4.2 Tg N/yr is used as an industrial feedstock (e.g., nylon and explosives production).

Fossil fuel combustion is the second major source. It introduces approximately 5.7 Tg N/yr into the environment, 3.8 Tg N/yr from transportation sources and 1.9 Tg N/yr from stationary (utility and industry) sources, almost entirely as $NO_x$.[3]

[3] Combustion of wood and other forms of biomass generally occurs at temperatures too low to convert $N_2$ to Nr

1 Figure 1: New Nr introduced into the US, 2002, Tg N



A third source of Nr introduced into the US is enhancement of biological nitrogen fixation (BNF) by crop cultivation, which introduces 7.7 Tg N/yr. A small amount of Nr is also imported in grain and meat; in 2002 it was approximately 0.2 Tg N/yr.

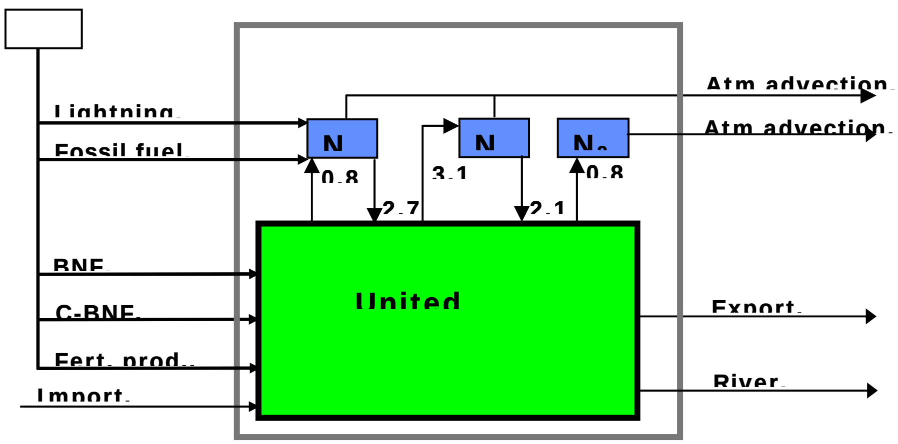
In summary, food production and fossil fuel combustion contribute the greatest amounts of Nr compared to other human sources in the United States. Although fossil fuel combustion is widely recognized within EPA and society in general to be a major source of nitrogen, sulfur, and carbon pollutants and resulting environmental quality concerns, in fact, feed and food production and subsequent consumption by animals and humans are much larger (about four times larger!) sources of reactive nitrogen than fossil fuel combustion.

**Reactive nitrogen transfer and transformations in and between environmental systems**

There are several possible fates for the approximately 35 Tg N/yr introduced into the US. Emission of $N_2O$ removes 0.8 Tg N/yr into the global atmosphere. Of the 6.3 Tg Tg N/yr of $NO_x$ emissions, 2.7 Tg N/yr are deposited back to the US, and by difference we estimate that 3.6 Tg N/yr per year are advected out of the US atmosphere. Similarly, of the 3.1 Tg N/yr of $NH_3$ that are emitted to the US atmosphere, 2.1 Tg N/yr are re-deposited, and 1 Tg N/yr is advected via the atmosphere. This value results from model consistency among media; a more through estimate of atmospheric advection of Nr can be found in Section 2.3. Riverine injection of Nr to the coastal zone accounts 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. All total, these losses sum to 14 Tg N/yr, leaving 21 Tg N/yr unaccounted for. Of this amount, we estimate that 5 Tg N/yr year are stored in soils/vegetation and groundwater, and, by difference, we estimate that 16 Tg N/yr are denitrified to $N_2$ (Figure 2). There are substantial uncertainties (+/- 50%) for some of these terms, especially those that involve $NH_x$ emission and deposition and terms that are arrived at by difference (e.g., atmospheric advection; denitrification). These uncertainties drive the first tier of recommendations of this report.

**Figure 2: Nr inputs into the US, exchanges with the atmosphere, and losses from the US via exports and riverine and atmospheric transport, 2002US nitrogen budget for 2002**

**Tg N /yr**




**Nr Inputs: 35 Tg N**

**Nr Outputs: 14 Tg N**

**Nr Missing: 21 Tg N**

**Nr Storage: 5 Tg N**

**~ 2 Tg soils&vegetation**

**~ 3 Tg groundwater**

**Nr Denitrified to $N_2$:**

**21 Tg N - 5 Tg N = 16 Tg N**

## Consequences, impacts and metrics for Nr

~~The best and most~~ important consequence of Nr is contributions to food production and global food security. In many 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.

There are however, numerous negative consequences from anthropogenic Nr, including photochemical smog, atmospheric particulate loading, ecosystem fertilization, ~~acidification~~, and/or eutrophication, greenhouse effect and stratospheric ozone depletion. Human activities have not only increased the supply but enhanced the global movement of various forms of nitrogen 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 (Figure 2). Nitrogen is a dynamic element easily transformed from one species to another and is transported rapidly through and between ecosystem reservoirs. These characteristics make it an especially challenging element to control.

Because nitrogen is both a critical resource and also a contributor to a number of environmental problems, it is imperative to understand how to reduce the risks to society while also providing the materials, food and energy required by society.

Various approaches can be used to prevent, eliminate, reduce, or otherwise manage risk. Understanding the environmental impacts of Nr can inform decisions on how best to manage nitrogen risks. There are two main approaches to characterizing these impacts: traditional impacts and ecosystem services.

Historically, EPA environmental protection programs have addressed impacts of Nr such as global warming, eutrophication, ecotoxicity, human health (cancer and non-cancer), acidification, smog formation, and ozone depletion, among others. Sometimes these impacts can be expressed in common metrics. Common metrics have the considerable advantage of defining a straightforward framework within which environmental standards can be derived that are protective of human health and the environment, the principal mission of EPA. Such metrics also encourage evaluation of damage from collective 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 sulfur dioxide ($SO_2$) and $NO_x$ on the acidification of watersheds can be expressed as a common metric.

The ecosystem services approach complements traditional impact characterizations by assessing causative contaminant emissions. It considers how a specific service provided by one or more ecosystems or the corresponding causative functions (e.g. categories such as climate change, nutrient cycling, and food production) is impaired. The attractiveness of this approach is its recognition that the health of humans and the environment are inextricably linked. Less clear, in some cases, are ways in which to measure and monitor these impacts.

Both ways of expressing nitrogen impacts have value. Historical approaches (i.e., effects based) provide a readily adaptable framework for regulation. Function-based categories (i.e., services based) provide a richer context for the complex connections among Nr inputs and transformations. Further, their impacts on human well-being and dollar-based impacts can identify those effects that have the greatest damage costs to society. Using multiple metrics may provide a clearer picture of priorities for action, identify effective control points for reducing Nr impacts, and provide insights into more effective regulatory strategy.

*Tradeoffs Among Nr Risk Reduction Options are Complex*

Once the foreseeable impacts are understood and the suite of benefits associated with various risk reduction options 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 reduce risks from Nr applied to agro-ecosystems may reduce yields and increase commodity prices, which in turn may result in expansion of crop production area at the expense of natural wetlands, grasslands, and forests.

*Measurement of Nitrogen in the Environment*

What ~~you~~ measure determines both what ~~you do~~ and how ~~you gauge~~ success or failure. Most regulations set limits or specify control technologies for specific forms of nitrogen without regard to the ways in which nitrogen is transformed once introduced into the environment. Normally, regulations also require some form of monitoring to document compliance. Monitoring of the specific forms of nitrogen 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 form of Nr can be quickly converted to other forms.

The impacts of reactive nitrogen often can be expressed as the dollar 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 reactive nitrogen 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. Similarly if human mortality and morbidity are the metrics used, priorities for Nr releases could be very different.

In order to determine the extent of damage caused by excess nitrogen[4] in environmental reservoirs, one needs to know the Nr concentration or loading within a reservoir and the threshold at which negative impacts are manifested. This threshold then provides a target that can be used to guide strategies to decrease the Nr in the reservoir. The thresholds for impacts are better known for some impacts relative to 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 $NO_x$, 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 loading to impact. On the other hand, the impact of Nr deposition on ecosystems is only generally known. There is strong scientific evidence to show that N 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 N deposition in that range, it is necessary to better define the link between N deposition and ecosystem response. Further, and related to the previous section, our knowledge of N deposition is uncertain, especially for the reduced inorganic and organic N species. This knowledge needs to be improved to better link deposition to ecosystem response (see Recommendation xx)

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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.

## Integrated Risk Reduction Strategies for Nr

Typically, quantitative risk assessment; technical feasibility; economic, social and legal factors; and additional benefits of the various control strategies contribute to the development of a suite of risk reduction strategies from which managers select an approach.

*Control Strategies for Nr*

There are several ways in which the release and control of Nr in the environment are 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 (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 lower dependency on Nr (e.g. switchgrass instead of corn grain as a feedstock for ethanol)
3. Transformation—in which one form of nitrogen is converted to another form (e.g. nitrification of wastewater, denitrification),
4. Source limitation—in which the amount of Nr introduced into the environment is lowered (e.g. lower fertilizer application rates, 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 reduced $NO_x$ from more efficient energy sources)

Effective management of Nr requires combinations of these approaches; no one approach is a perfect alternative for controlling Nr in the environment.

*Management of Nr in the Environment*

Generally speaking, US environmental policies employ four mechanisms for the management of contaminants in the environment:

1. Command-and-Control—in which permitted limitations on emissions, as promulgated under various statutes, are issued. Violations may result in the assessment of penalties.

2. Government-based programs for affecting 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 cap and trade markets 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.

An integrated approach to the management of Nr must use a combination of mechanisms. Each mechanism must be appropriate to the nature of the problem at hand, supported by critical research on reducing the risks of Nr, and reflect an integrated policy that recognizes the complexities and tradeoffs associated with the nitrogen cascade. Control 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.

## Findings and Recommendations

The committee's recommendations can be loosely organized into several tiers. These include recommendations that:

1. address deficiencies in knowledge about Nr flows and fates.
2. concern ecological and human impacts of Nr.
3. address specific actions that can be taken to decrease Nr in the environment.
4. address how EPA could develop an integrated N management strategy, in cooperation with other agencies.

There is over-lap among these tiers, as some recommendations call for both research and immediate action. But collectively they represent a view of what is needed to develop an integrated Nr management strategy based on sound science. This integrated strategy keeps in mind the nutritional demands of the US and its trading partners. Specific recommendations for implementing an integrated strategy appear in the subsequent chapters of this report. In addition to providing these specific recommendations, the report also makes four over-arching recommendations:

### Recommendation A

*EPA should pursue an integrated approach to develop the understanding necessary for science-based policies, regulations, and incentives to avoid and remediate the impacts of excess Nr on the environmental, human health, and climate. Such integration must cut across media (air, land, and water), Nr form (oxidized and reduced), federal agencies, and existing legislative statutes [e.g., EISA – Energy Independence and Security Act (EISA), the Clean Air Act, and the Clean Water Act (CWA)].*

### Recommendation B

*EPA should form an Intra-agency Task Force on Managing Nr that builds upon existing Nr efforts within the Agency. The task force would identify the most cost-effective approaches to avoid the negative impacts of Nr loads cascading through the environment. These loads pose a significant threat to human health and environmental quality and directly affect climate change.*

**Recommendation C**

*The federal government should form an Inter-agency Task Force on Managing Nr be formed, with EPA as the lead agency that includes at a minimum U.S. Department of Agriculture (USDA), U.S. Department of Energy (DOE), U.S. Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), and U.S. Geological Survey (USGS). This Task Force would coordinate federal programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and regulating Nr in the environment.*

The task forces should take a systems approach to science and research by:

- evaluating critical loads
- Nr budgets and life cycle accounting
- monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads
- development and use of systemic models for Nr management; new technologies; fertilizer and nutrient best management practices (BMPs)
- development of Nr indicators necessary for the assessment of effects related to excess Nr on human health and the environment
- assessing combined carbon (C) and N effects
- addressing indicators/endpoints, costs, benefits and risks associated with the impairment of human health and decline and restoration of ecosystem services
- investigating the need for new regulations
- new education, outreach, and communication initiatives
- implementing economic incentives, particularly those that integrate air, aquatic, and land sources of Nr
- new infrastructures for managing Nr releases to the environment
- review of enabling legislation for purposes of extending regulatory authority or streamlining procedures for enacting Nr risk reduction strategies.

In addition to these three overarching recommendations, the committee has 25 more specific recommendations that are located in the main body of the report at the appropriate locations, and are collated in Appendix 3. Four of these recommendations propose specific actions EPA and other agencies could initially take to decrease Nr losses to the US. These four recommendations collectively would result in ~25% decrease in the amount of Nr lost to the US environment. Given the issues of Nr accumulation in the US and global environment, these initial steps will not be enough to reverse the collective Nr damages to environmental and human health. Other steps will need to be taken after attention is paid to all the recommendations of this report.

# 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 19$^{th}$ 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 20$^{th}$ 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 the movement of Nr between reservoirs and potential losses 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 losses during food production, the challenge is how to minimize those losses 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. All 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[5] or the "nitrogen cascade" (Figure 3). Unlike other element-based pollution

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[5] 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.

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).

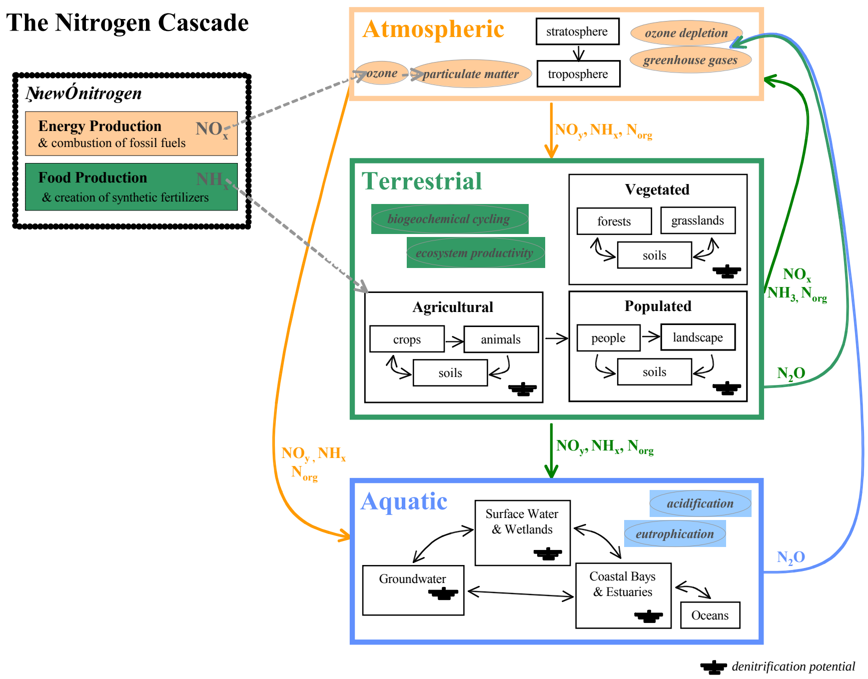
The "biogeochemical" dimension of the nitrogen cascade involves Nr creation from $N_2$ as a consequence of chemical, food and energy production, Nr use in food and chemical production, Nr losses to the environment, changes in Nr species residence times in environmental reservoirs, Nr transfers among 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.

**<u>Figure 3: The Nitrogen Cascade</u>**




*The popular 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 theUS. 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 important process of denitrification is the only mechanism by which Nr is converted to chemically inert $N_2$, 'closing' the continuous cycle.*

*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. Losses 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 losses (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. Losses 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.*

## 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.

## <u>1.3. The need for integration</u>

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 <u>Charge and scope of this report</u>

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

Although N is a major required nutrient that governs growth and reproduction of living organisms, Nr losses from human and natural sources have a profound effect on air, water and soil quality. Human consumption of energy to sustain economic development results in emissions of $NO_x$ to the atmosphere via fossil fuel combustion. Consumption of food to meet nutritional requirements of a growing population results in agricultural emissions of $NH_3$, urban and industrial emissions of $NO_x$, and $N_2O$ as well as losses of $NO_3^-$ and other N compounds to water bodies due to leaching and runoff. Once released into the atmosphere by either human or natural processes, these Nr compounds undergo transformation through atmospheric reactions (e.g. gas-to-particle conversion), transport associated with wind, and finally wet and dry deposition. Reactive nitrogen lost from agricultural and ~~peopled~~ systems can enter groundwater, streams, lakes, estuaries, and coastal waters where the Nr can also undergo transformation mediated by a wide range of biotic and abiotic processes. The introduction of Nr into agroecosystems provides much of the world's food. The loss of Nr to the environment throughout the food production process and during fossil fuel combustion contributes to many of ~~the major environmental problems of today.~~

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.

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

* Newly created reactive N

| **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 ag 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 | |

<u>Table 1 Notes</u>

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.

<u>Table 1 Data Sources</u>:

- 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[6]. 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

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

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).

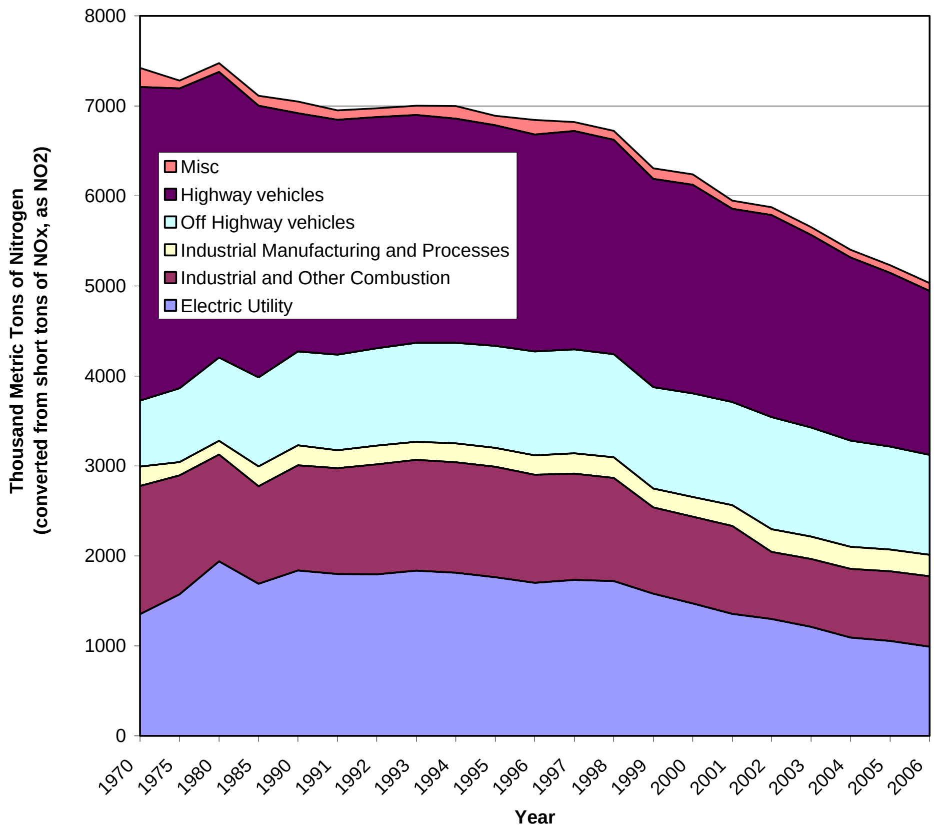
Thus in 2002, anthropogenic activities introduced a total of 29 Tg N into the US, mostly in support of food production, although turf production, industrial uses and fossil fuel combustion were also important sources. Natural sources of Nr in the US are BNF in unmanaged landscapes, and lightning. The 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.

Losses 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 losses from the various components of both energy and food production.

### 2.2.2 Nr formation and losses 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 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.

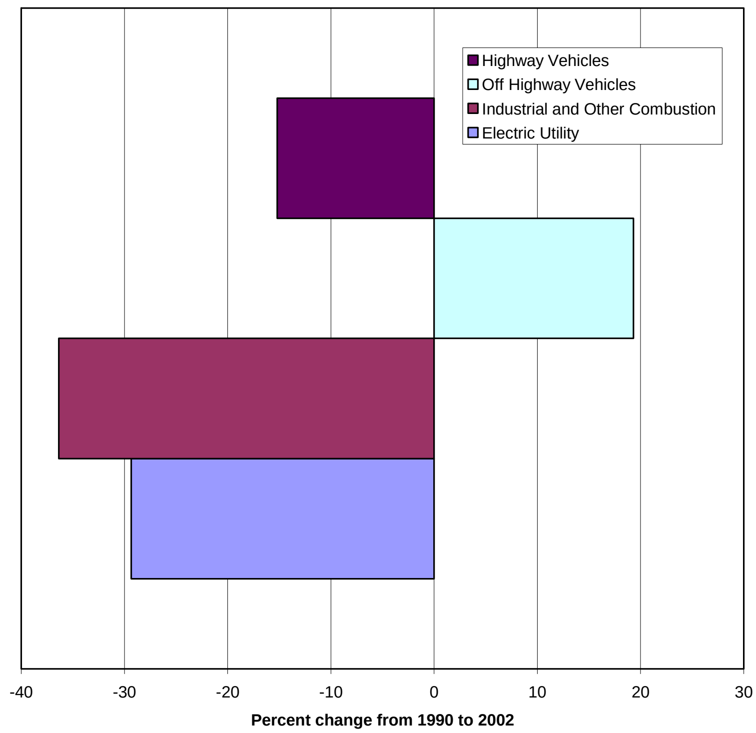
**Figure 4: US $NO_x$ emission trends, 1970-2006. Data are reported as thousand of metric tons of N converted from $NO_x$ as $NO_2$**




(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% (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.)**




Texas, California, Florida, Ohio, and Illinois emissions (in metric tons of nitrogen as converted from tons of $NO_x$ or $NO_2$) with the processes listed (Table 2), illustrate the fact that individual state emission scenarios are quite different.

**Table 2: Top 5 Emitters of N in metric tons (2001 data; based on 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** |

As seen in Table 2, 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%). Louisiana and Texas have high emissions due to industry because of the chemical and oil industries located in these states. These results illustrate that many sources contribute to the $NO_x$ emitted from energy sources and the number of automobiles is a factor. The number of automobiles is related to the population. 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.

### 2.2.3 Nr inputs and losses from crop agriculture

Agriculture uses more Nr and accounts for more Nr losses 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 $NO_3^-$ losses from leaching, runoff and erosion and gaseous emissions via volatilization of $NH_3$ and NOx and nitrification/denitrification. Similar loss 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 must 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 losses 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 U.S. 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)

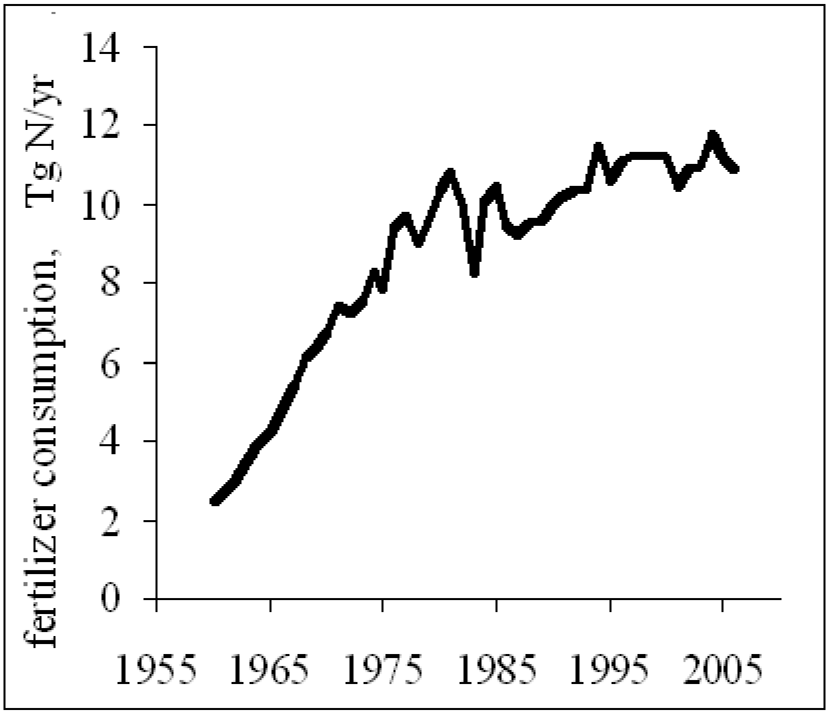


**Table 3: Sources and amount of nitrogen fertilizers used in the U.S. in 2002.**

[Data from Terry et al. (2006)]

| Synthetic Nitrogen Fertilizers | Tg/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) of total US Nr losses 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 of N applied to major crops and the geospatial pattern by major watersheds.

**Recommendation 1**: *Improve detail and regularity of data acquisition for fertilizer use by major crop (and for urban residential and recreational turf) and county (or watershed) to better inform decision-making about policies and mitigation options for nitrogen in these systems, and to allow monitoring of impact from implemented policies and mitigation efforts.*

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 NUE as the grain yield per unit of applied N. 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.

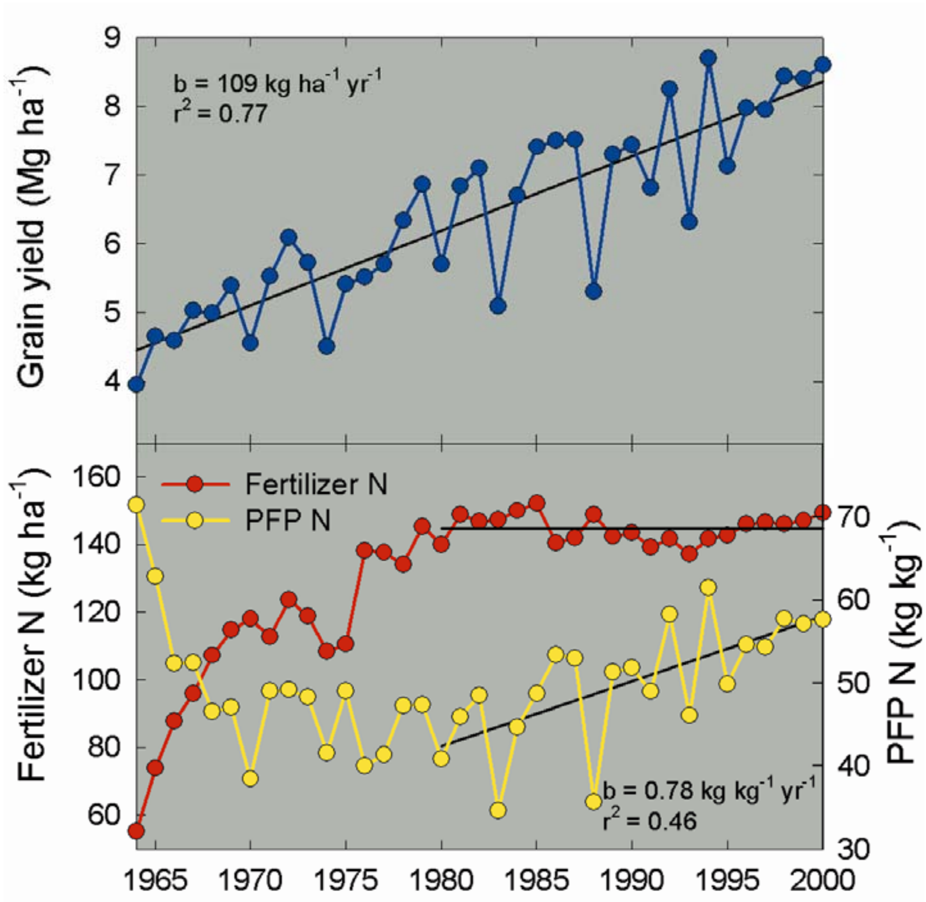
A recent review of N-use efficiency for cereals from research field studies around the world, mostly conducted on "small-plot" experiments at research stations, reported mean single year fertilizer N recovery efficiencies for maize, wheat and rice of 65%, 57% and 46%, respectively (Ladha et al., 2005). However, crop fertilizer N recoveries based on actual measurements in production-scale fields are seldom greater than 50% and often less than 33%. For example, a review of N fertilizer recovery in different cropping systems, (Cassman et al., 2002) estimated average recoveries of 37% for maize in the north central U.S.

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 NUE is fundamental for prioritization of research and education investments, both in the public and private sectors. While management can substantially improve NFUE 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 losses 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 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, there has been a 38% increase in NFUE since 1980 (Figure 7).[7] Similar improvements have been documented for rice production in Japan and for overall crop production in Canada.

[7] 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 US corn yields, N fertilizer rates applied to corn, and N fertilizer efficiency quantified by the partial factor productivity (PFP) for applied N (kg grain produced per kg N fertilizer applied) (Cassman et al., 2002).**




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. 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 "smart" 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 USrequired 0.45 kg of applied N to produce 19.1 kg of grain yield in 1980, whereas that same amount of N produced 26.5 kg of grain in 2000 (units converted 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 nitrogen fertilizer efficiency achieved in 2000, 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's strong positive impacts on yields in crops (e.g. corn) creates a strong economic incentive for its use. Nitrogen costs have increased dramatically since Hurricane Katrina, while corn prices have also increased dramatically with the increase in corn-based ethanol plants. However, the critical factor is the corn-to-fertilizer price ratio. 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 losses, 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 et al, 2002; Cassman, 1999). 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 losses 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 losses compared to conventional cropping systems that require less land to produce the same amount of crop output.

**Finding 2**

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, and fiber demand of a growing human population while also protecting environmental quality and ecosystem services for future generations (Cassman, 1999). More diverse cropping systems with reduced N fertilizer input may also provide an option if there is a reduction in Nr losses per unit of crop production, which is required to avoid indirect land use change from expansion of crop production area to replace the loss in production.

**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 to better focus research investments, policy development, and prioritization of risk mitigation strategies.*
b) *Research is needed with an explicit focus on the challenge of both accelerating the rate of gain in crop yields on existing farm land while substantially increasing N fertilizer uptake efficiency and also on quantifying whether widespread adoption of lower-yielding more diverse cropping systems[8] with lower N fertilizer input requirements can reduce regional Nr load when the impact of indirect land use change is considered.*
c) *EPA should work closely with the U.S. Department of Agriculture (USDA), Department of Energy (DOE), and the National Science Foundation (NSF) to help identify research and education priorities for prevention and mitigation of Nr applied to agricultural systems.*

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[8] Greater diversity in a cropping system is achieved by increasing the number of different crop species used in the rotation (temporal diversity) or as a polycrop or intercrop (spatial diversity).

*2.2.3.2. 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 from Boyer et al. 2002.

*2.2.3.3. Emissions factors and losses 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, 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 $N_2O$ losses 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 $N_2O$ losses 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).

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., 2007), 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 NUE 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 must 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, and NSF.*

*2.2.3.4. Impact of biofuel production capacity on Nr flux in agriculture*

The current transportation system in the US requires enormous amounts of liquid motor fuels at a time when petroleum use exceeds petroleum discovery. Hence, the price of petroleum has increased substantially during the past 10 years, and most of the world's petroleum reserves are located in politically unstable countries. This situation provides strong motivation for investment in biofuels made from crops,[9] and a number of countries have enacted favorable policies and incentives to foster expansion of biofuel production capacity. In the US, ethanol production from corn has more than doubled to 41 billion liters/yr since 2005, and the renewable fuel standard established in the 2007 EISA will support expansion of this capacity by another 16 billion liters/yr by 2015. Brazil is rapidly expanding its production of sugarcane ethanol, Europe and Canada are expanding biodiesel production from canola oil, and Indonesia and Malaysia have plans to increase biodiesel production from palm oil.

When petroleum prices are high, corn has greatest value as feedstock for biofuel rather than for human food or livestock feed (CAST, 2006). Because of the steady rise in petroleum prices from mid-2005 until mid-2008 and the 2007 EISA mandate,, the amount of corn used for ethanol has increased rapidly; about 30% of US corn production will be used for ethanol in 2008, which represents about 10% of global corn supply. This increased demand puts a floor under both corn prices and ethanol prices, which have risen substantially since the first half of 2005. Higher corn prices send powerful signals to corn producers to increase production. Farmers have responded by increasing corn acreage by millions of acres since 2006, and they may be motivated to increase N fertilizer rates to boost yields. However, N fertilizer prices have also risen so the net impact of expanded biofuel production on actual N rates used by crop producers is uncertain. 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).

[9] In addition to crops, biofuels can be made from algae and forest wood waste products.

**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:** *There exists a critical need to understand and predict these changes in terms of maximizing the N efficiency of both crop and livestock production systems and to develop strategies for avoiding increased Nr load in the environment as a result of current and future expansion of biofuel production from corn and other "second generation*" biofuel feedstock crops.

### 2.2.4. Nr inputs and losses from animal agriculture

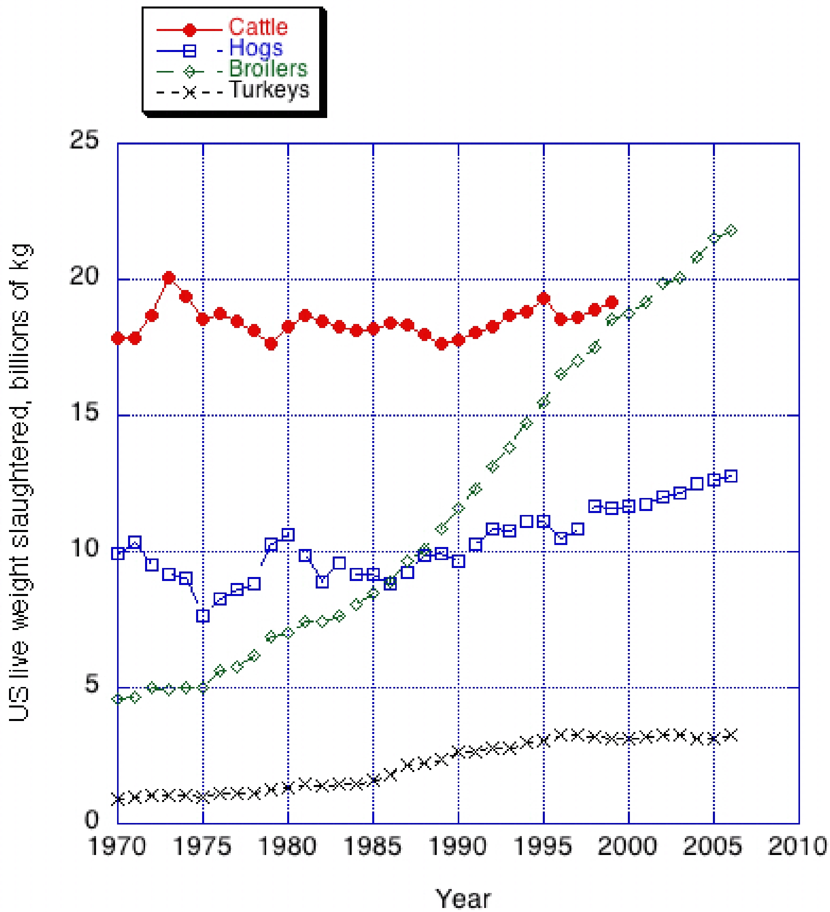
In the US, domestic animals produce 6.0 Tg N/yr in manure and are the largest source of atmospheric $NH_3$-N (1.6 Tg N/yr) (Table 1). Livestock also contribute to $N_2O$-N emissions, though in much smaller proportions (~4% of total US $N_2O$-N emissions).

#### *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).

1 **Figure 8: Meat production from 1970 to 2006. Source: USDA-NASS, Census Reports**

2 (Data on cattle not taken after 1999)

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4 Another trend in animal production has been for fewer animals to produce more animal products. For
5 example, the 60% greater amount of milk produced in 2006 compared to 1970 required 25% fewer cows
6 (Figures 9 and 10). Animal inventories declined by 10% for beef brood cows from 36 million head in
7 1970 to 33 million head in 2006, and the inventory of breeder pigs and market hogs declined 8% from
8 673 million head to 625 million head in the same period, even with similar or greater annual meat
9 production. This trend resulted from greater growth rates of animals producing more meat in a shorter
10 amount of time. In 1970, broilers were slaughtered after 80 days on feed at 1.7 kg live weight, but by
11 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.**

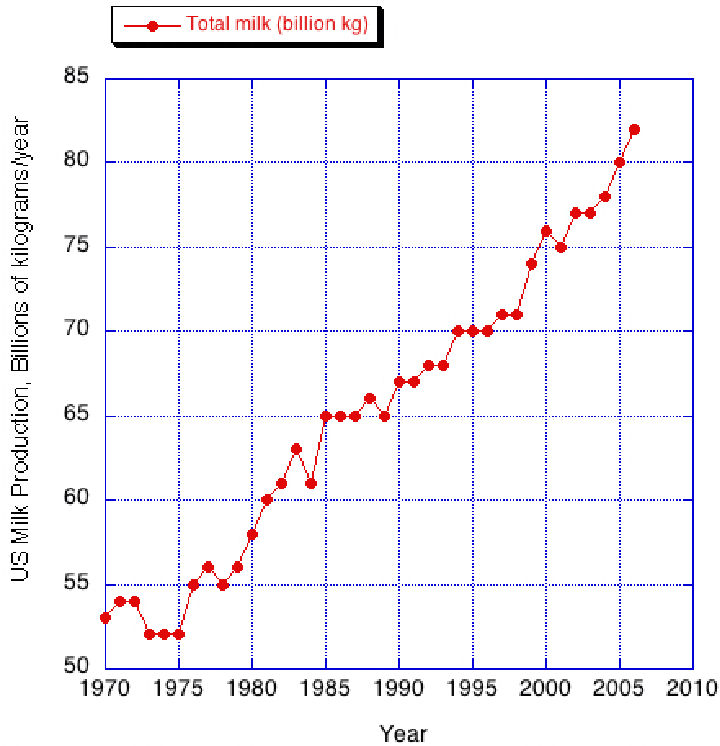
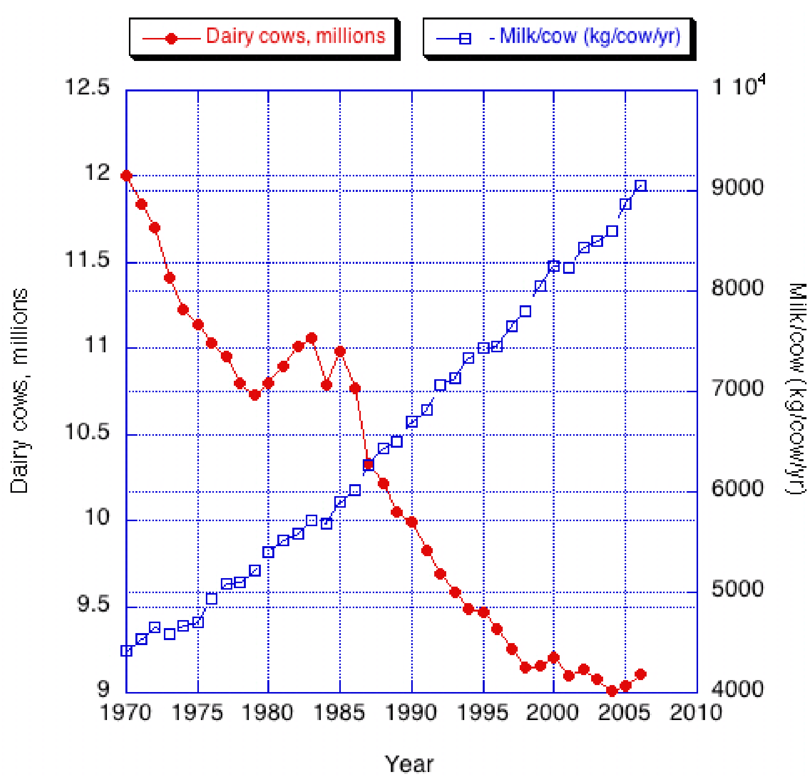
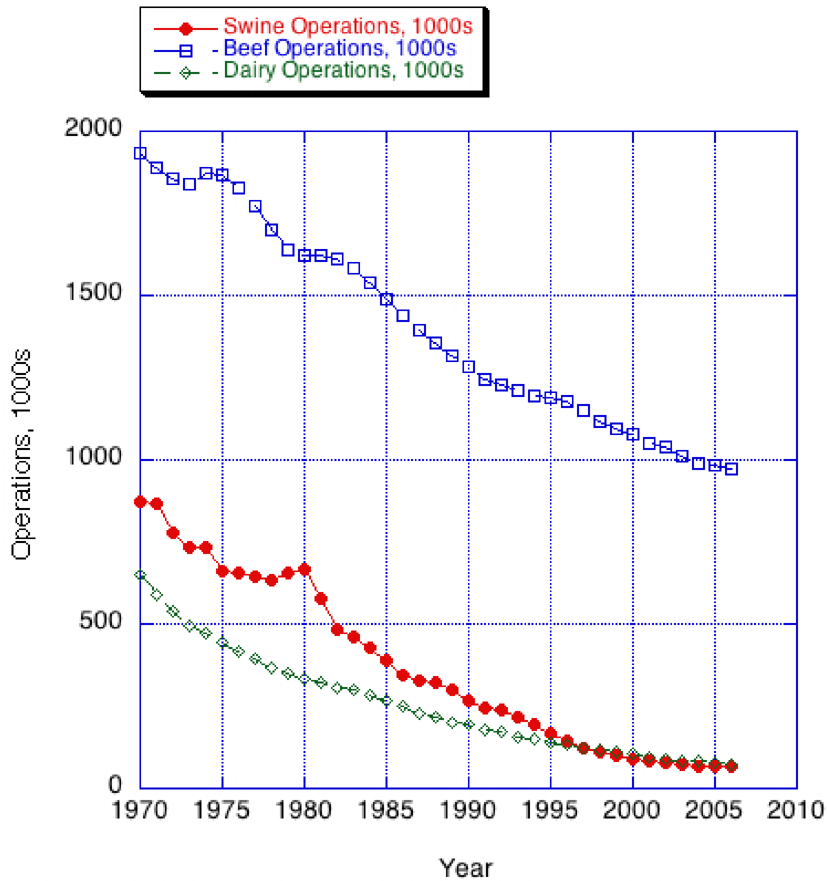



**Figure 10: US Inventory of mature dairy cows and milk production per cow from 1970 to 2006. Source: USDA-NASS, Census Reports.**



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). 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 aerosol formation, soil acidification, eutrophication, loss of biodiversity, and the neutralization of acids produced by sulfur and nitrogen oxides. 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 $NO_3^-$ 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.

**<u>Finding 5</u>**

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 nationwide by a 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 NRC 2996 recommendations Surprisingly, NRC 1996 recommended greater total crude protein compared to NRC 1976 despite formulating for rumen degraded and undegraded 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. Nitrogen excretion*

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.

**Table 6: Manure N excreted 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. Levels of pH greater than 7.0 allow $NH_3$ to undergo volatilization. Otherwise, $NH_3$ is in the form of $NH_4^+$ and therefore cannot be volatized (Arogo et al., 2006; James, 2008).

EPA estimates annual manure N excreted in livestock production in the US. for the "Inventory of U.S. 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 lost 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 lost 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 losses. There are currently no incentives or regulations to decrease these losses 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 losses, 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 loss 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. Volatile losses of Nr 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.**

| | Area (Million ha) | N rate kg/ha* | Total N Used Tg N |
|---|---|---|---|
| **Type of turf fertilized** | 4.7 | 73.2 | 0.35 |
| **Nominal fertilization** | 0.93 | 296 (195-488) | 0.27 |
| **Professional lawn care high maintenance areas (golf/sports)** | 1.26 | 390 | 0.49 |
| **Total** | **6.89** | **--** | **1.11** |

*10,000 m$^2$/ha, used values of 0.73, 2.92 and 3.89 kg N/100 m$^2$ for nominal fertilization, professional lawn care, and high maintenance areas, respectively.

## 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 U.S. 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.

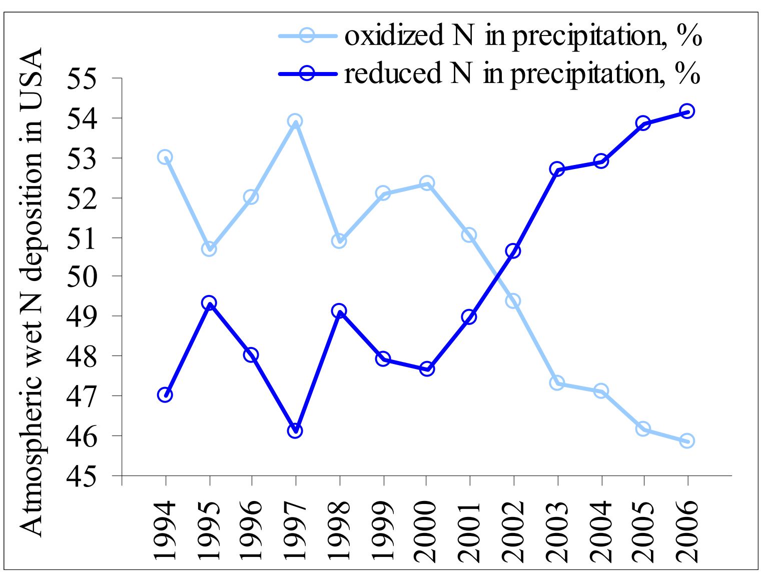
*Review of Nr wet deposition.* Substantial progress has been made in monitoring wet deposition, as is summarized by the National Atmospheric Deposition Program/National Trends Network (NADP), established in 1979, which monitors precipitation composition at over 250 sites in the US and its territories (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 |

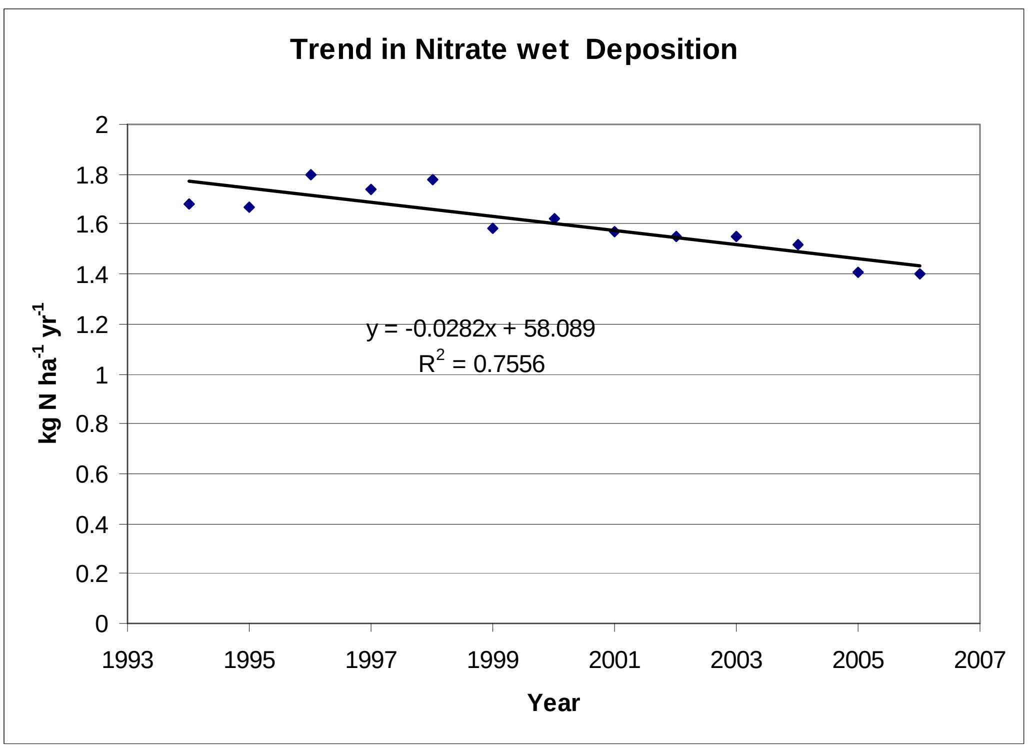
**Figure 12: Percent change in relative contribution of oxidized ($NO_3^-$) and reduced ($NH_4^+$) nitrogen wet deposition from 1994 to 2006. As emissions of $NO_x$ have decreased, the relative importance of $NH_x$ has increased.**



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 n Recommendation D.

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

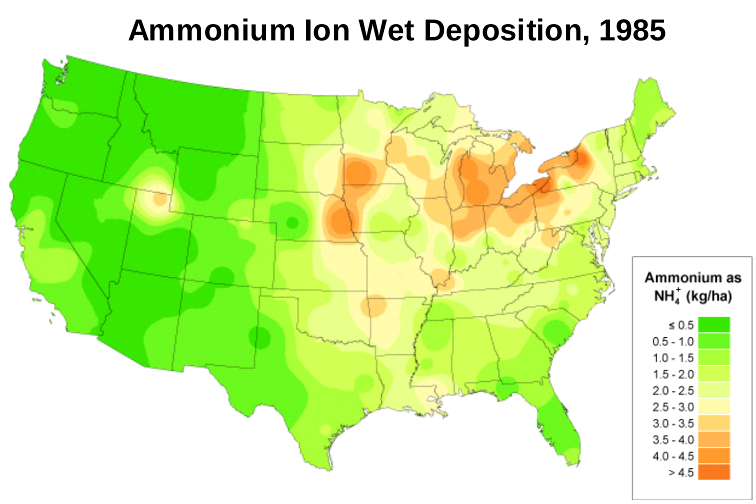
Note the sampling methods and locations have not been tested for temporal or spatial bias.

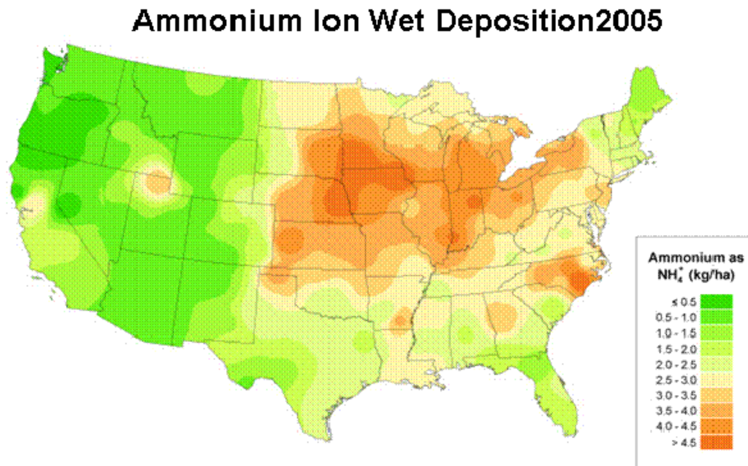



*How is Nr deposition related to emissions?* The relationship between emissions of Nr and observed deposition is critical for understanding the efficacy of abatement strategies as well as for partitioning local and large-scale effects of emissions. Only a few studies covering several individual sites 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 NO is weak, dry deposition of $NO_3^-$ accounts for 39 % of the total $NO_3^-$ 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 $\mu g/m^3$ in the Southeast and 2.40 $\mu g/m^3$ 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 $\mu g\ N/m^3$ 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 $\mu g\ N/m^3$ 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 spatially sparse. 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.

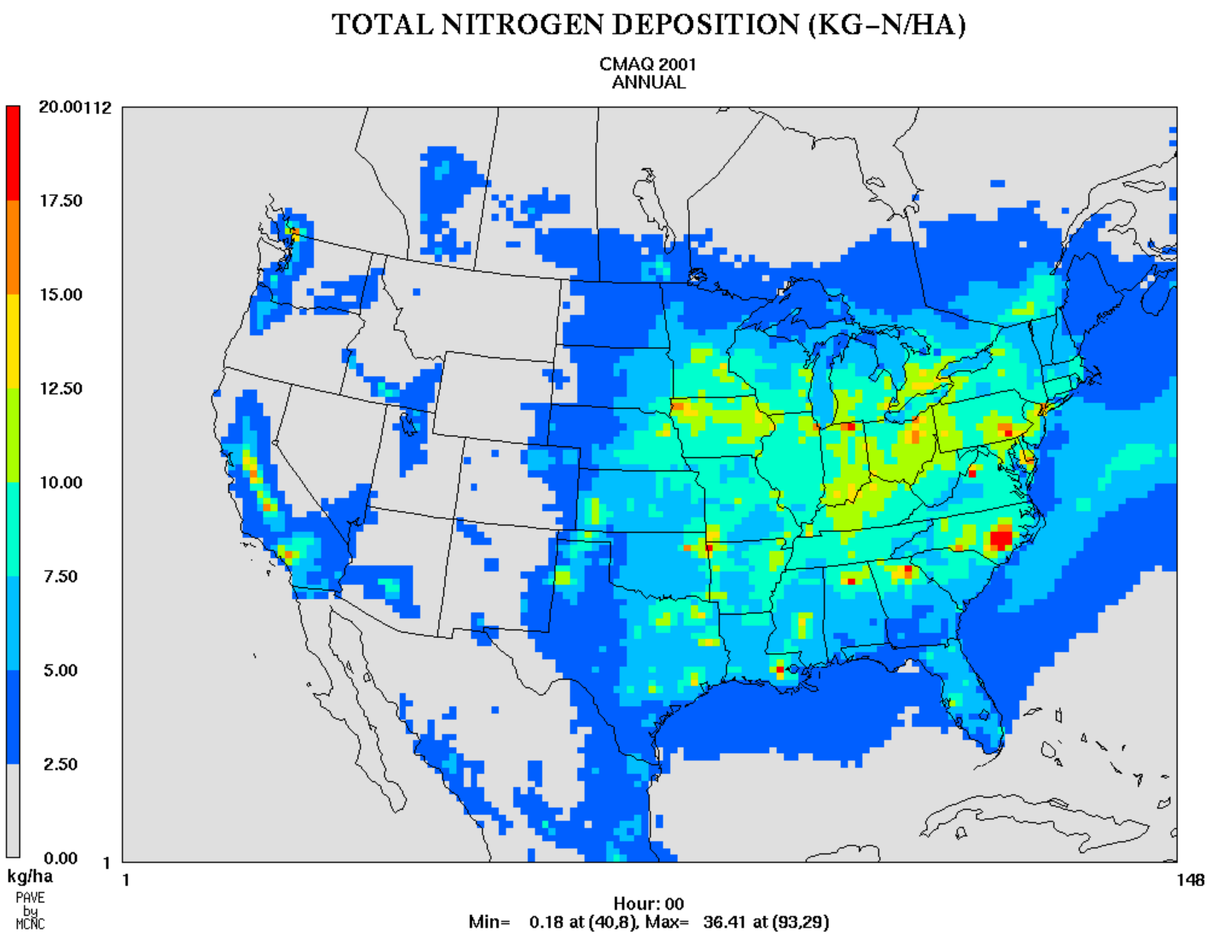
**Table 12: Results from CMAQ for total deposition in 2002 to the 48 contiguous states of oxidized and reduced N.**

| | 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.

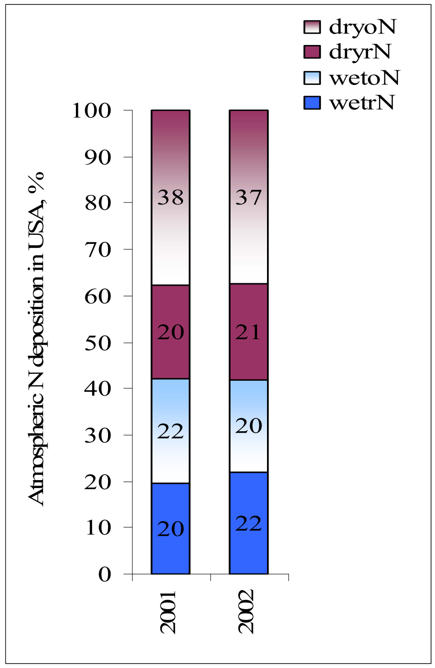
The three-year CMAQ run gives an indication of the spatial pattern of deposition and the relative importance of wet and dry deposition (Figures 15 and 16). For $NH_x$, wet and dry are equally important, but for $NO_y$, dry deposition is greater than wet. While this is not true for the eastern US it is 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 U.S. based on 3 years of differing meteorology - one dry, one wet, and one average precipitation year - across the Eastern U.S. (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.

**<u>Figure 16: Relative contributions of wet and dry deposition for reduced and oxidized N. Results from CMAQ runs for the 48 contiguous States.</u>**




For comparison purposes, a collection of <u>Chemical Transport Models</u> (CTM's) [*Dentener et 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 $NH_x$, 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 the 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; *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 within 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). Duce et al (2008) suggest that organic Nr constitutes a ~~fair~~ 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.

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 7**

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. 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. Ammonia, $NH_4^+$, and possibly organic N levels in the atmosphere are too high to protect public health and welfare, and reductions of $NH_x$ emissions are necessary.

**Recommendation 7a.** *Increase the scope and spatial coverage of the Nr concentration and flux monitoring network (such as CASTNET) for the US and appoint an oversight panel.*

**Recommendation 7b.** *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 techniques.*

**Recommendation 7c.** *The current $NO_2$ standard is inadequate to protect health and welfare, and compliance monitoring for $NO_2$ is inadequate for scientific understanding.*

**Recommendation 7d.** *Measure deposition directly both at the CASTNET sites and nearby locations with non-uniform surfaces such as forest edges.*

**Recommendation 7e.** *EPA should continue and support research into convective venting of the Planetary Boundary Layer and long range transport.*

**Recommendation 7f.** *Develop techniques and support observations of atmospheric organic N compounds in vapor, particulate, and aqueous phases.*

**Recommendation 7g.** *Improve quality and quantity of measurements of the $NH_3$ flux to the atmosphere from major sources especially agricultural practices.*

**Recommendation 7h.** *Improve numerical models of NOy and NHx especially their chemistry, surface deposition, and 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 loss 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 U.S. 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 $person^{-1}$ $day^{-1}$, 0.16 g N $g^{-1}$ $protein^{-1}$, 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 in the terrestrial portion of the nitrogen cascade within the continental US in 2002**

| Environmental System | N Input | N Storage* | Products | Loss |
|---|---|---|---|---|
| **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 |

*Estimates from section 2.3.2.

**Finding 8**

Although total N budgets within all terrestrial systems are highly uncertain, Nr losses 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 losses ascribed to leaching, runoff and denitrification, are as uncertain as the N budgets themselves.

**Recommendation 8:** *EPA, USDA, DOE, and universities should work together to ensure that the N budgets of terrestrial systems are properly quantified and that the magnitude of at least the major loss vectors are known, by funding appropriate research.*

*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 thestates 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).

~~The availability of N controls or "limits" primary production in much of the world's estuarine, near-shore coastal and open ocean waters (Dugdale 1967, Ryther and Dunstan 1971, Nixon 1995, Smetacek et al. 1991, Jørgensen and Richardson 1996; Duce et al., 2008). 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. Nitrogen inputs 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. Because marine ecosystems lose some fixed N via denitrification and burial, "new" N supply is needed to compensate for these N losses.~~ 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) ~~or microbial looping.~~

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 U.S. 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 U.S. 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 U.S. (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.

**Text Box 1: Hypoxia in the Gulf of Mexico**

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 9**

Over the past 25 years, there has been a growing recognition of ~~anthropogenic~~ 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 9:** *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 U.S. 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 U.S. land area in 2002. This equates to approximately 5.5% of the U.S. 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 US. (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 52.5 x $10^3$ Tg of C in above and below ground in US forests; soil contains 59% of total C, 9% in litter and 5% in tree roots**.** The EPA estimate for 2002 is 43.6 X $10^3$ 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** | 5.80 | 0.48 |
| **Total** | 3.7 | 0.31 |

| | | |
|---|---|---|
| **US Total C & N Storage in 2002** | 186 | 1.7[10] |

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

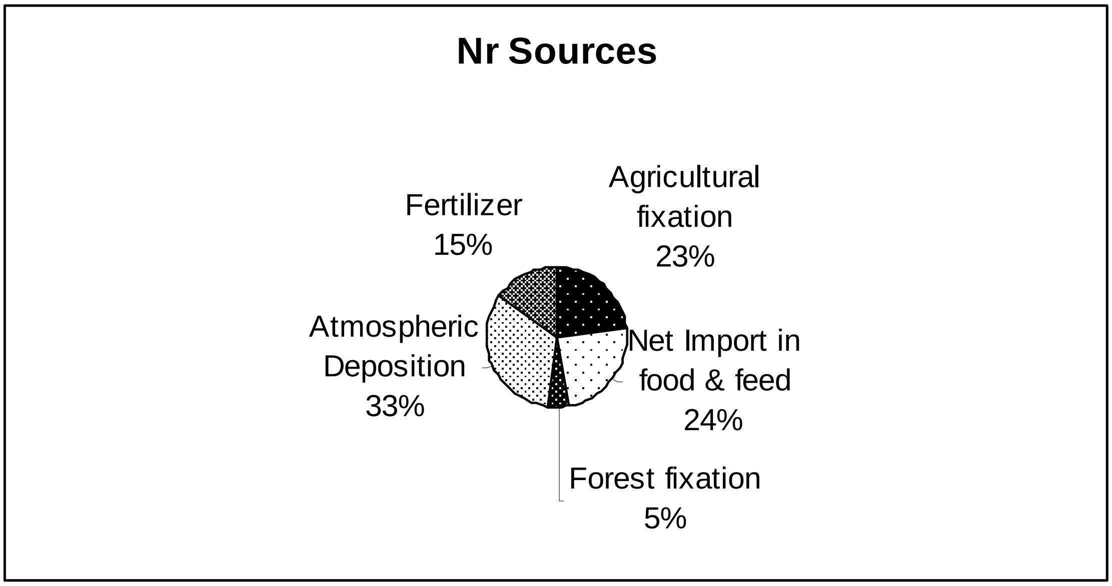
There are no comprehensive data available to assess the transfer and transformations in and between the atmosphere, terrestrial systems (agriculture, populated and vegetated systems) and aquatic systems nationally. Determining a national N budget is a priority research area. As there are no national data available, an example analysis of Nr input and fate in 16 watersheds in the northeastUS, 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).

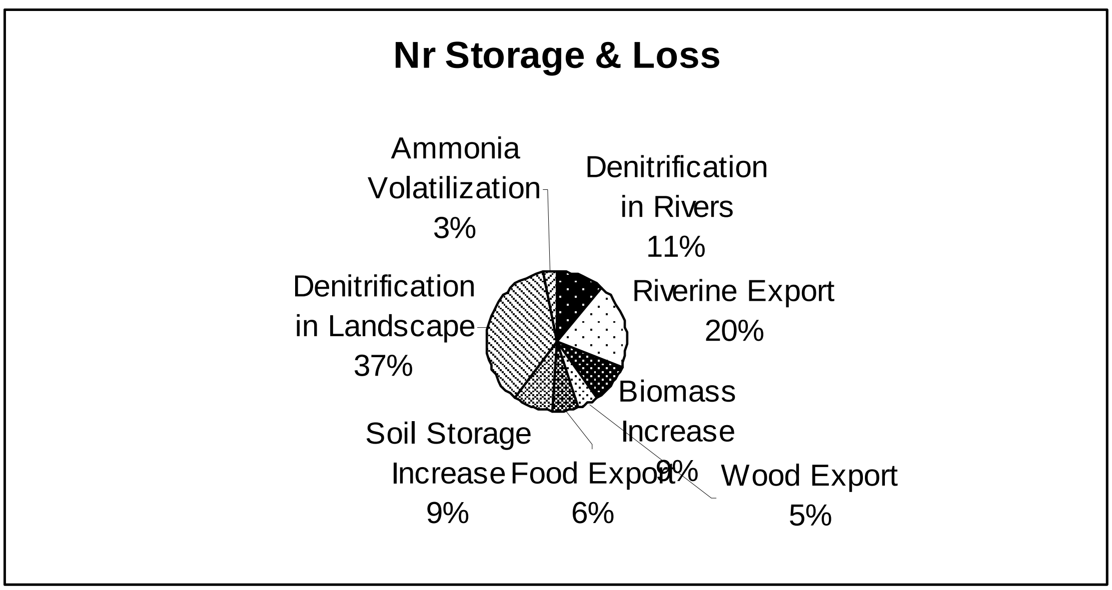
The watersheds in this study encompass a range of climatic variability, Maine to Virginia. The watersheds are a major drainage to the coast of the North Atlantic Ocean. Using data from the early 1990s, Boyer et al. (2002) reported the quantification of N inputs to each watershed from atmospheric deposition, 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 17 shows the Nr sources and the estimated fate of this Nr as a per cent of the weighted average Nr input.

[10] 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.

**Figure 17: Nr input and loss from 16 watersheds in the northeast US.**

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 10</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 10:** *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. Synopsis of areas of the US nitrogen cascade where the estimates of Nr transfer and transformations in and between environmental systems are highly uncertain.

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 that are produced and released to the atmosphere during nitrification/denitrification. These gases include $NO_x$, $N_2O$ and $N_2$.

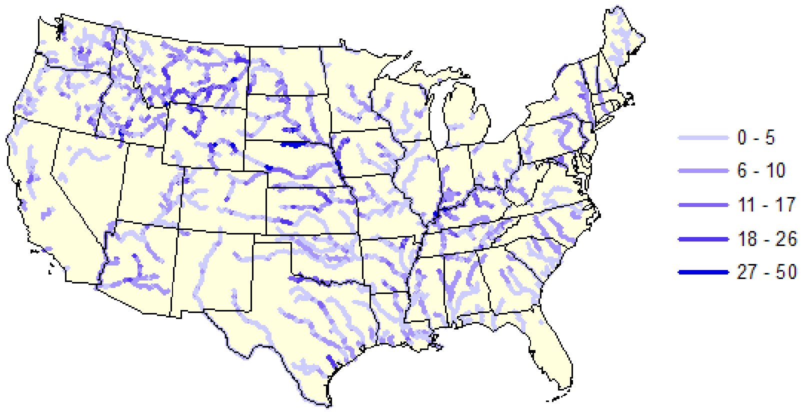
- The amount of Nr deposited in each environmental system as dry deposition needs to be quantified and monitored.

- Rates and amount of ammonia emission 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.

**Figure 18: Total Nr yields (kg/ha/yr) in large rivers of the US (Alexander et al. 2008).**




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

### 2.4.1 Measurement of N 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 $NHx_3$, or kg/ha of $NO_3^-$.

**Finding 11**

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

**Recommendation 11.** *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 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 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 the ideal 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 global warming, 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; WRI 2005). 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.

**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 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 12**

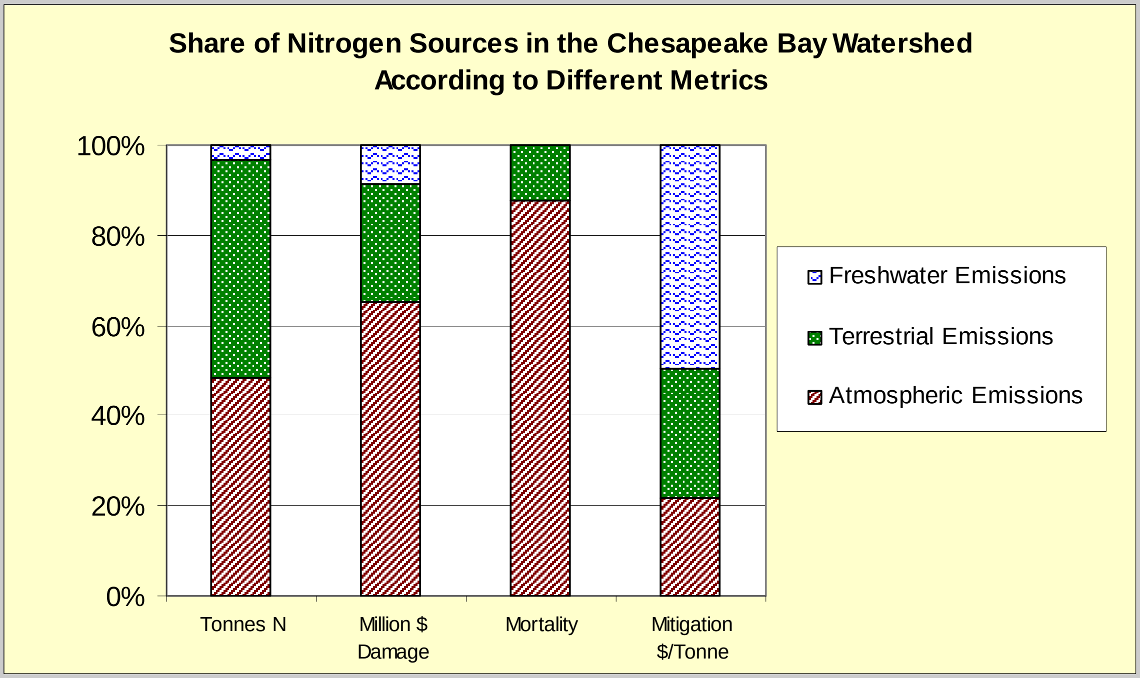
The committee finds that reliance on only one approach for categorizing the measurement of Nr is unlikely to result in the desired outcome of translating N-induced degradation into the level of understanding needed to develop support for implementing effective N management strategies.

**Recommendation 12.** *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 20 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 19: 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 18.

**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 |

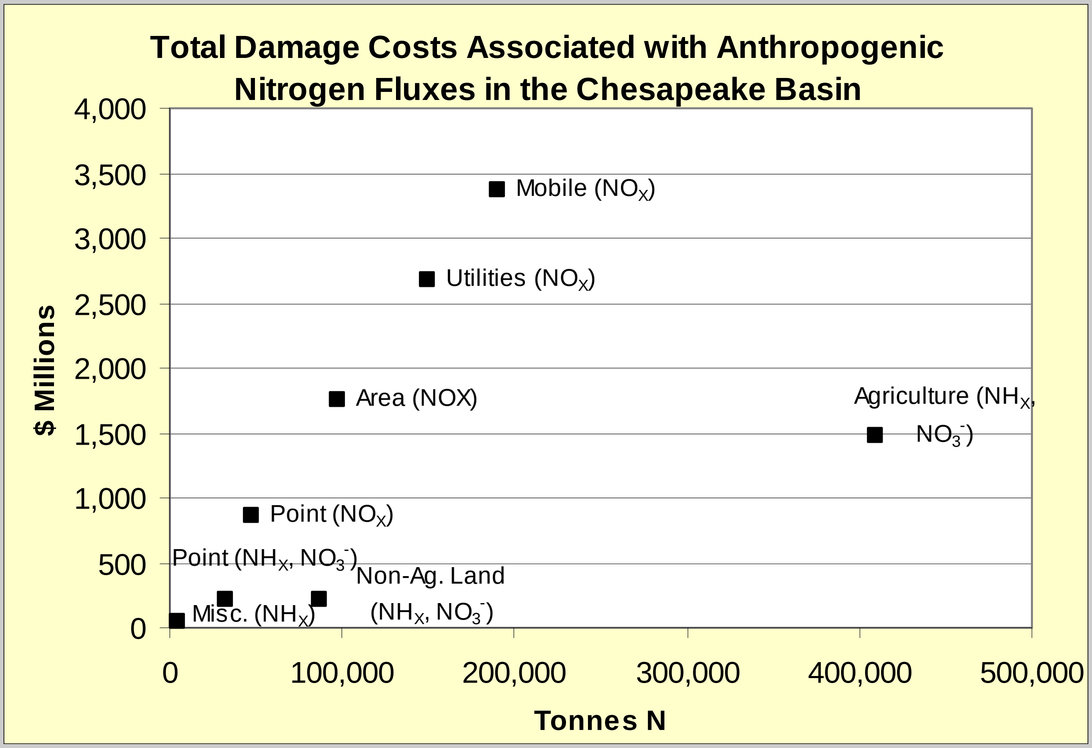
| | | | | |
|---|---|---|---|---|
| Freshwater Emissions | 32,000 | 223 | **0** | $19,000 |

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 21.

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 20: 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).

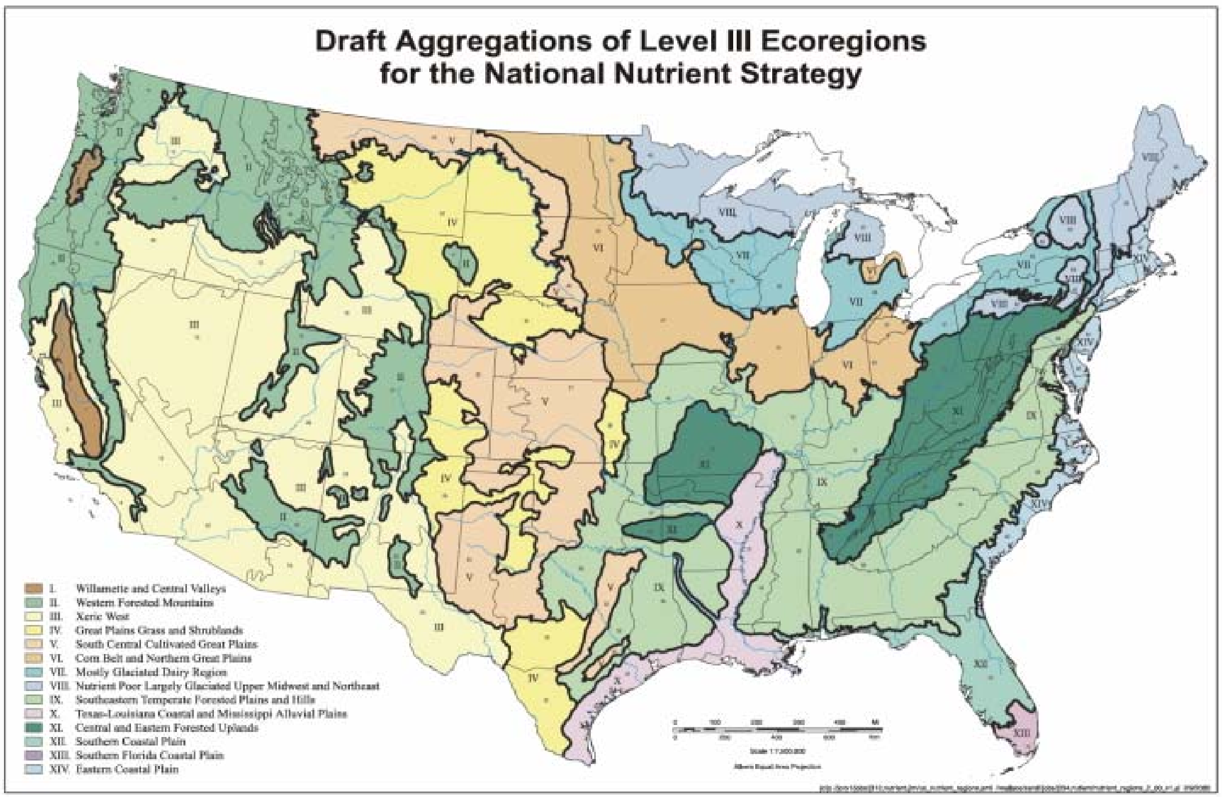
The EPA sets minimum requirements for approvable standards and criteria including: use designations; water quality criteria sufficient to protect the designated uses; and an antidegradation policy (40 CFR Sec. 131.6). Traditionally, Nr and other land, air and water pollutants are measured in terms of 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), and ecoregional guidance for lakes and reservoirs and rivers and streams (Figure 21). 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.

**Figure 21: Fourteen nutrient ecoregions as delineated by Omernik (2000)**

Ecoregions were based on geology, land use, ecosystem type, and nutrient condition, including economic and ecosystem services.



*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 (Figure 22) 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 US where deposition to surface waters falls within this range.

*2.4.3.43 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 13**

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 STPs 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 13.** *To better address Nr runoff and discharges from the peopled landscape the committee recommends that EPA*:

**13a.** *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.*

**13b**. *Review current regulatory practices for point sources, including both wastewater treatment plants and stormwater, to determine adequacy and relationship 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.*

**13c.** *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.*

**13d.** *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.*

**13e.** *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 23.

**Table 23: Estuaries with nitrogen management plans or TMDLs and target levels**

| Estuary | Nitrogen Management 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% | ? |
| **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 Nr loadings to estuaries would cumulatively be less than 25%, which is below the targets identified in Table 23. 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 disproportionate results. Despite regional efforts and commitments from all watershed states, and more funding than any other estuary program is likely to see, they are falling short of management targets and are coming off a discouraging year (2007) that saw a severe hypoxic episode. Similarly, the adoption of the Long Island Sound TMDL (See Long Island Sound Text Box) 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 14**

Meeting Nr management goals for estuaries, when a balance must be struck between economies, society and the environment, under current federal law seems unlikely. Enforceable authorities over nonpoint source, stormwater, air (in terms of critical loads), and land use – both development and agriculture are inadequate to require necessary Nr controls. Funding programs are presently inadequate to meet existing pollution control needs. Further, new technologies and management approaches are required to meet ambitious Nr control needs aimed at restoring national water quality.

**Recommendation 14.** *Therefore, the committee 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 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 sewage treatment plants (STP 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 15**

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 used to formulate new management procedures. The process of monitoring and control revisions is termed adaptive management—a process that the committee supports as it does not delay actions that can be taken now, but acknowledges the likelihood that management programs will be altered (adapted) as scientific and management understanding improve.

**Recommendation 15.** *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 USGS 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 US 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] 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 18: 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 $m^{-3}$ |
| **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 20, 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 US and both central and northern Europe
3. Nr saturation of forest soils, presently occurring mainly in high-elevation forests of the eastern US 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 US and Europe at ambient concentrations of ozone above 60 ppb. Such concentrations occur frequently throughout the eastern US 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, 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 US 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 Mediteranean climate of southern California are likely to be very different from those for direct effects on similar systems in other regions of the US -- 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 US 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.

In this connection, the INC 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 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 ($SO_x$)].

*2.4.5.7 Additional comments on Nr critical loads*

The INC recommends that the EPA give very careful consideration to adoption of the critical loads concept in determining thresholds for effects of excess Nr on terrestrial and aquatic ecosystems. The committee's considered judgment for this recommendation is based on the following important events and ideas.

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 aimed decreasing 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.

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.

**Table 19: 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.**

The relevance and link to Nr provide a prioritization for future international action to mitigate the effects of excess N. The last column summarizes existing links to international regulations and conventions.

| | 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<br><br>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<br><br>Clean Air for Europe |
| **direct toxicity of nitrite $NO_2^-$** | $NO_2^-$ conc | Y | $NO_x$ | 2 | World Health Organization<br><br>Convention on Long-range Transboundary Air Pollution<br><br>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<br><br>$NO_3^-$ (&P) conc (aq) | N | Runoff, Nr deposition | 1 | Convention for the Protection of the Marine Environment of the North-East Atlantic<br><br>Helsinki Commission<br><br>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 20: 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..**

The relevance and link to N provide a prioritization for future international action to mitigate the effects of excess nitrogen. The last column summarizes existing links to international regulations and conventions.

| | 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<br><br>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<br><br>Clean Air for Europe WFD |
| **Eutrophication of freshwaters, lakes (incl. biodiversity)** | Biological Oxygen Demand, $NO_3^-$ conc (aq)<br><br>Critical loads | Y<br><br>N | Runoff, NR deposition | 3 | Water Framework Directive |
| **Eutrophication of coastal ecosystems inducing hypoxia (incl. biodiversity)** | BOD, $NO_3^-$ conc (aq)<br><br>Critical loads | BOD, $NO_3^-$ conc (aq)<br><br>Critical load | Runoff, NR deposition | 1 | Convention for the Protection of the Marine Environment of the North-East Atlantic<br><br>Helsinki Commission<br><br>Barcelona Convention |
| **Nitrogen saturation of soils (incl. effects on GHG balance)** | Critical loads | Y | NR deposition | 1 | Convention on Long-range Transboundary Air Pollution<br><br>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<br><br>Clean Air for Europe,<br><br>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 21: 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.**

The relevance and link to N provide a prioritization for future international action to mitigate the effects of excess nitrogen. The last column summarizes existing links to international regulations and conventions.

| | **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.

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 losses 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 threshold is the limiting factor, putting caps on losses from sources. A pre-classification of regions might be useful, e.g. urban regions, remote regions, marine areas, etc.

### 2.4.6 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. Four such categories of tradeoffs are ammonia release from concentrated feed lot operations (CAFOs), human nutrition, nitrification-denitrification, and nitrogen-carbon related impacts.

*2.4.6.1 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.

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 US 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 neither that $NH_3$ reduces the toxicity of atmospheric aerosols nor 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.

**Finding 16**

The committee finds that the net benefit of $NH_3$ emissions reductions greatly outweigh any potential harm.

**Recommendation 16.** *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.*

*2.4.6.2 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 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 was decreased (Figure 23) from approximately 400,000 MT (metric tons) in 1991 to 200,000 MT in 2002. Animal manure N application decreased from 250,000 MT to approximately 240,000 MT during this time period. Nevertheless, although N input into Danish cereal crop production decreased, cereal crop yield remained relatively constant, as shown in Figure 24.

**<u>Figure 22: Synthetic and livestock manure used as N fertilizer in Denmark (IFA 2004).</u>**

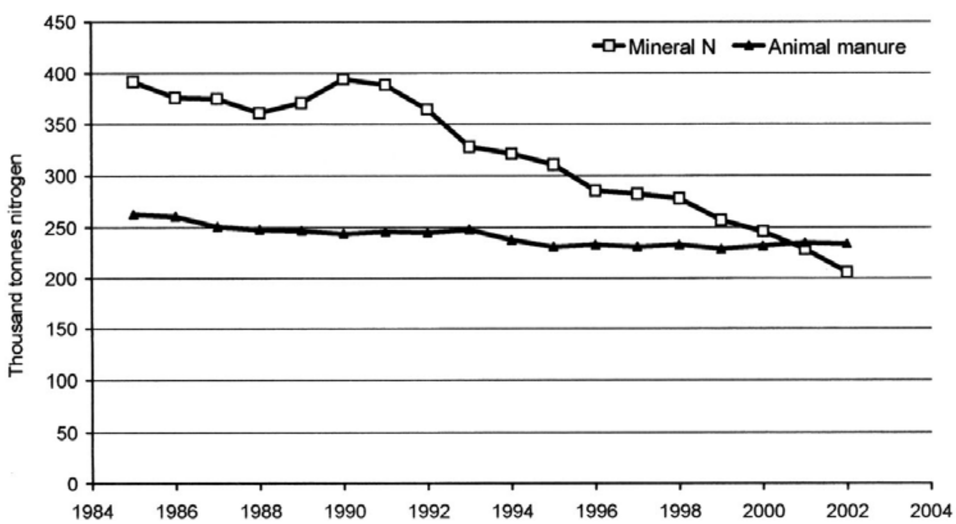
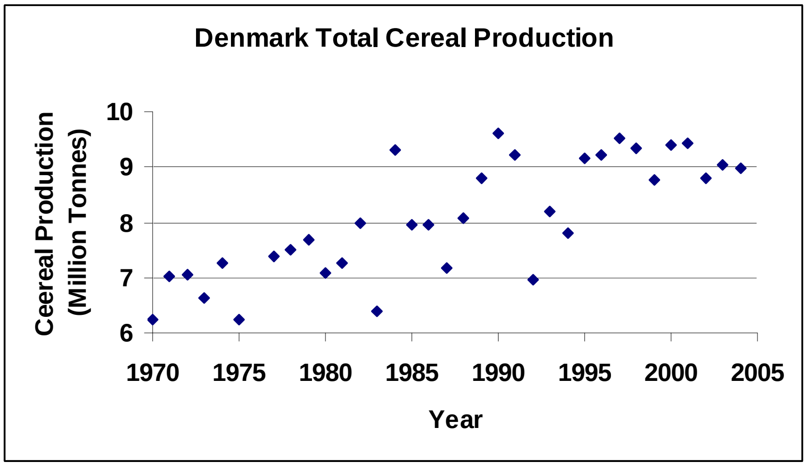


**Figure 23: Total cereal grain production in Denmark (FAOSTAT, 2007**




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 3.3). Although US fertilizer application has not declined over time, it has leveled off in recent years, as shown in Figure 25. 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 26.

**Figure 24: Consumption of N-containing Fertilizers in the US (USDA)**

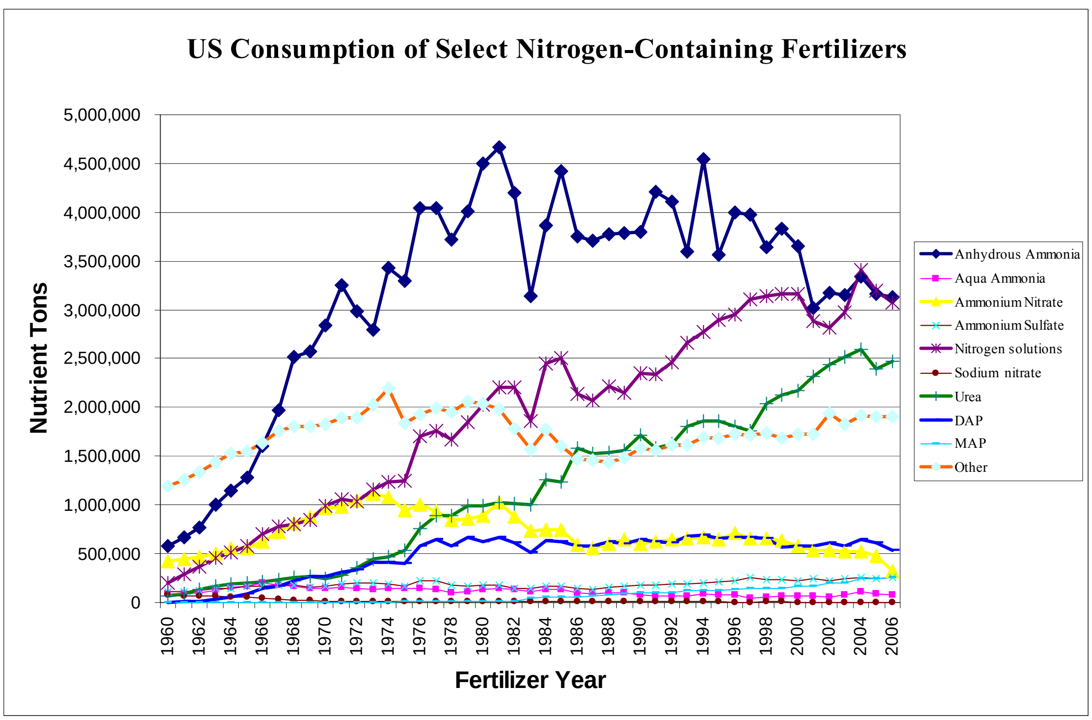
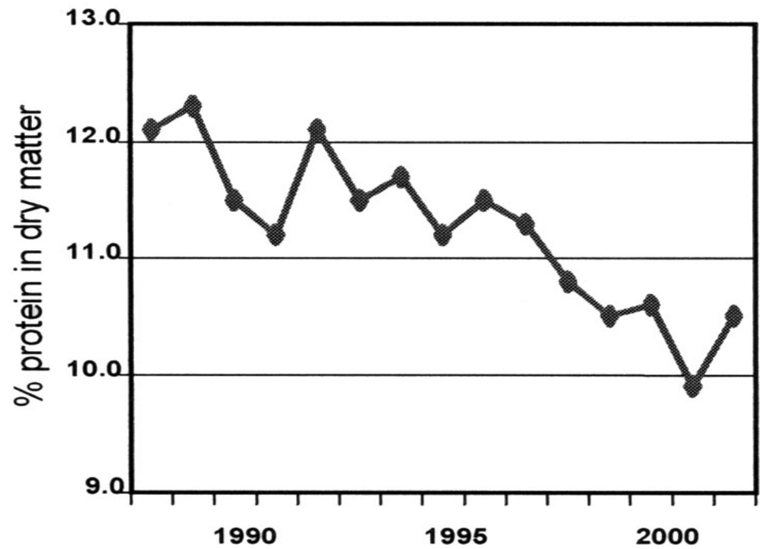


**Figure 25: Corn Grain Produced per Unit of Fertilizer N used in US (Fixen and Ford, 2002)**


The effect of decreasing N fertilizer input can be estimated based on the current level of PFP for applied N (average US 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 27 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 US 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 26: 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? 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.

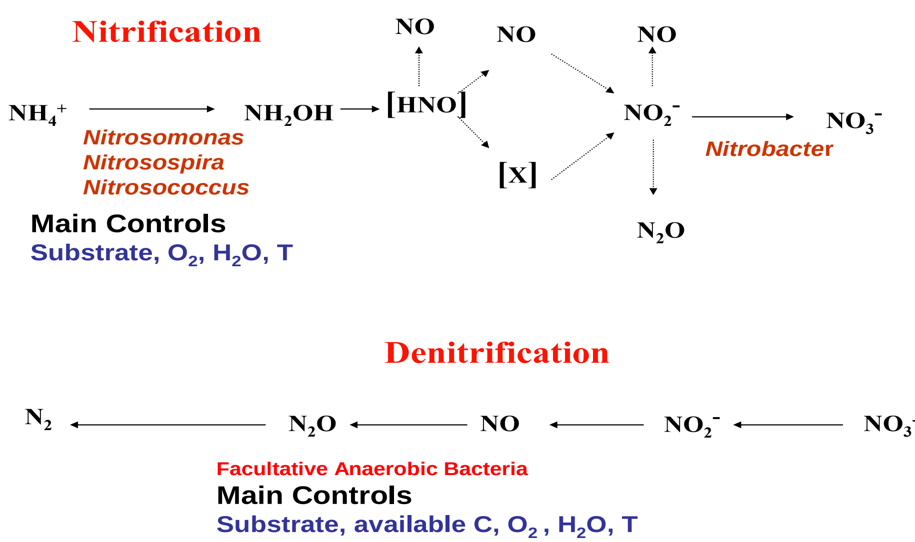
*2.4.6.3 Unintended impacts: swapping N between environmental systems*

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 28). 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 28. 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).

**<u>Figure 27: Diagram of the nitrification and denitrification processes (from Mosier and Parkin 2007)</u>**




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 US 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.

*2.4.6.4 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 29. 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 30 shows the latter case in the form of global warming 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 30 represents net C sequestration. It is difficult to make direct comparisons across disparate impact categories, however Figure 32 suggests that, in choosing among alternatives, policies that aim to minimize both sets of impacts would be preferred.

**Figure 28: Combined carbon and nitrogen global cycles (Miller et al. 2007)**

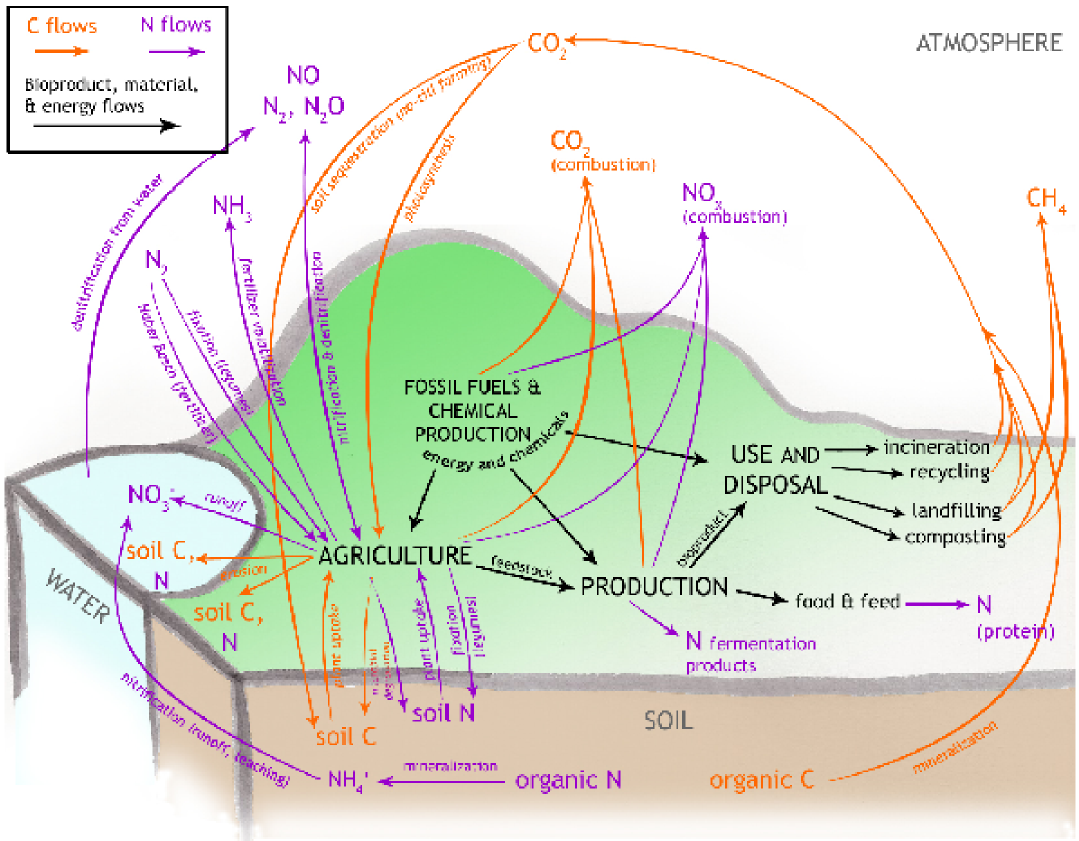
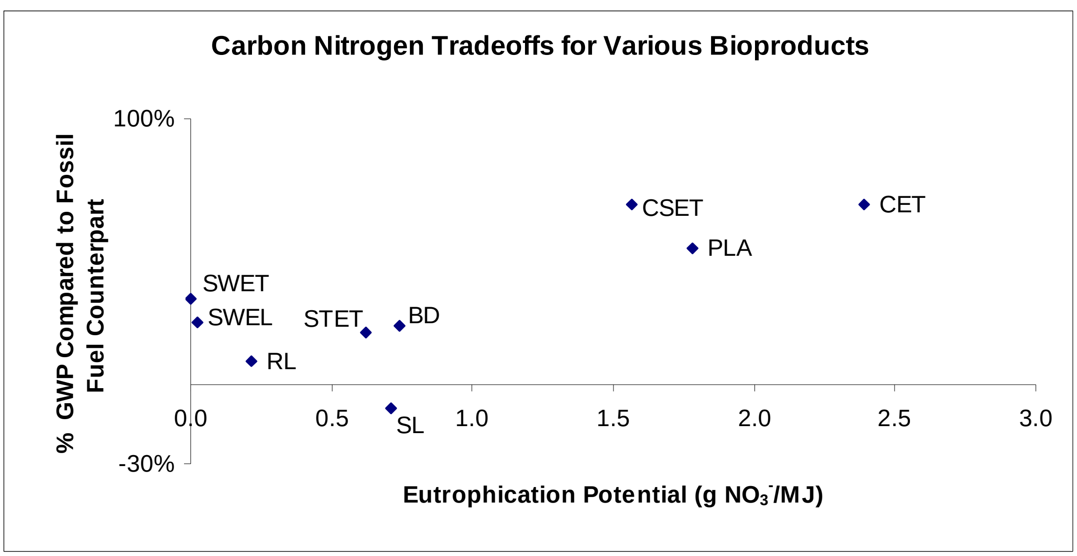



**Figure 29: Comparisons Between Global Warming and Eutrophication Impact Categories for Various Bioproducts (Miller et al. 2007).**




(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 17**

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 17.** *The committee recommends that the integrated strategies for N management specified in this report be developed in cognizance of these interrelations and tradeoffs.*

# Chapter 3: Integrated risk reduction strategies for reactive nitrogen

## 3.1 Introduction

In Chapter 2, the environmental impacts and metrics associated with the emission of the various forms of Nr were presented, and ways of organizing these into impact "categories" reviewed. 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 the 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.

## 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—in which the flux of Nr that creates an impact is lowered through better management practices (e.g. on-field agricultural practices, controlled combustion conditions, urban development and landscape management practices)
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 24 provides a summary of the pros and cons of each of these approaches.

**Table 22: Advantages and limitations of various approaches to Nr control**

| Control strategy | Advantages | Limitations |
|---|---|---|
| Improved practices, conservation | Lessens one or more impacts | Education cost |
| 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 | May contribute to other impacts |
| 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 | Residuals containing Nr must still be managed effectively |
| 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[11]

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 US 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 Total Maximum Daily Load 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.

**Finding 18**

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 18.** *The committee recommends that the Agency work toward adopting the critical loads approach in the future. 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*

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

*cooperative, adaptive, and systemic approaches that derive from a common understanding of the nitrogen cascade*.

### 3.3.2 Market based instruments for pollution control [12]

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 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:

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[12] 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.

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 (see section 4.3.3).

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 25 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 (Water Quality Trading to Meet the Long Island Sound Wasteload Allocation in Connecticut and Water Quality Trading in the Illinois River Basin).

## Text Box 4: Water Quality Trading to Meet the Long Island Sound and Wasteload Allocation in Connecticut

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 31). 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 32). 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 23: 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 30: Relative nitrogen discharge (lbs/day) from 79 POTWs**

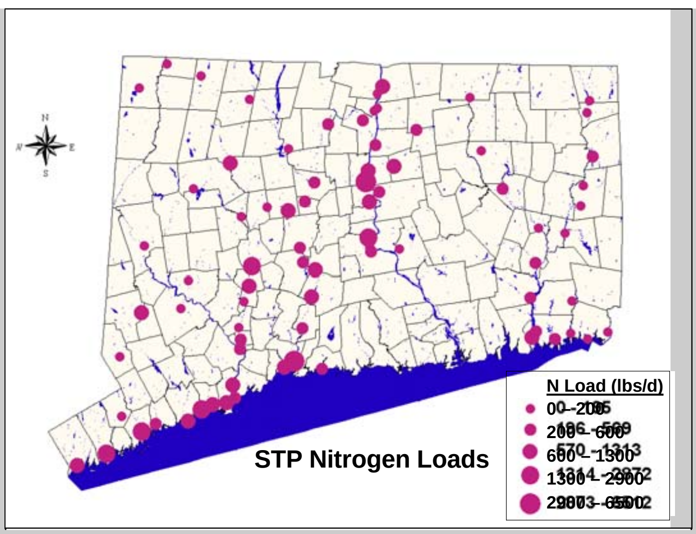
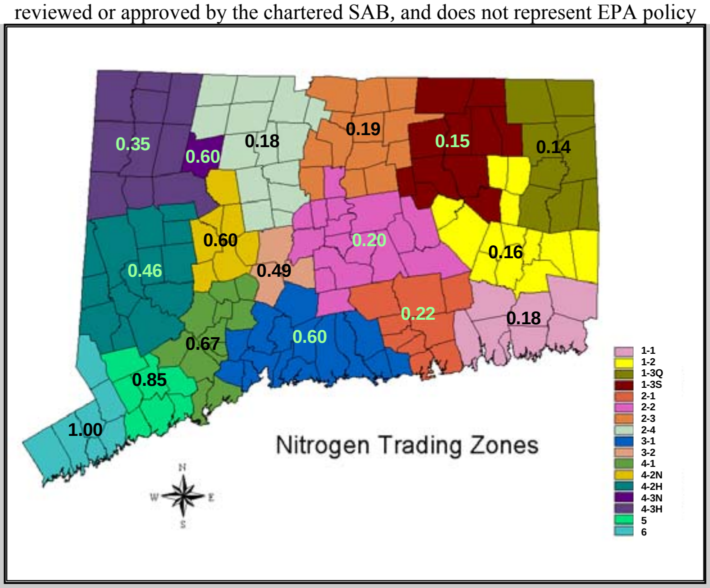



**Figure 31: Trading ratios for municipalities in Connecticut**




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.

## 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** |

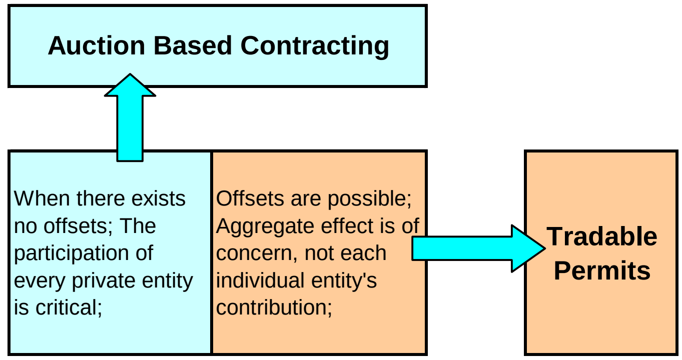


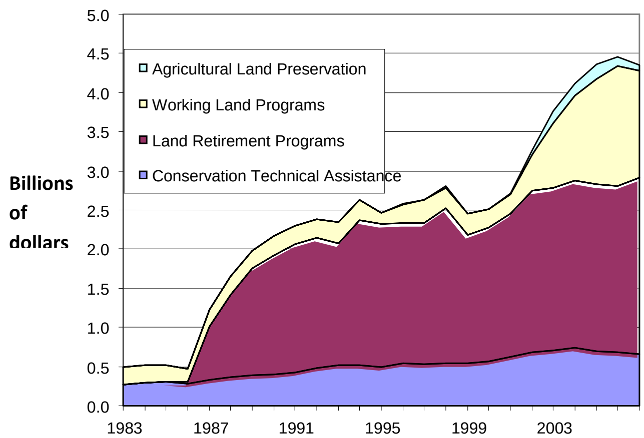

Table 25 shows pair-wise comparison between different market-based strategies. The objective and the incentive structure of the participants determine the suitability of one market based strategy over another. Each pair of cells briefly lists the most relevant set of conditions for which the respective 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: 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.

### 3.3.3 Government programs, mandates, and policy conflicts

The direct allocation of federal funds, and government policy mandates (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 US 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 US 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 rograms 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 2.3). 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 32: 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 the accompanying text box.

**Text Box 5: 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, for the WRP 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 34. The mass loading of the buyers (2,423 tons/month) is reflected in Figure 35. 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 36, 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 35. 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 38. 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 33: Distribution of municipal (> 1 MGD discharge), and industrial dischargers in the Illinois River Watershed; symbols may represent more than one discharger at that location**

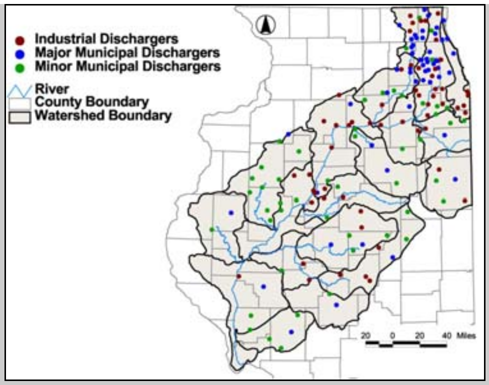



**Figure 34: Distribution of total nitrogen emissions by sub-watershed**

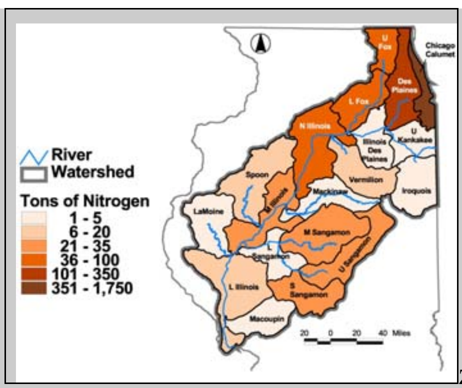


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**Figure 35: Potential land availability in the 100-year flood zone for nutrient farming in each sub-watershed in the Illinois River Watershed**

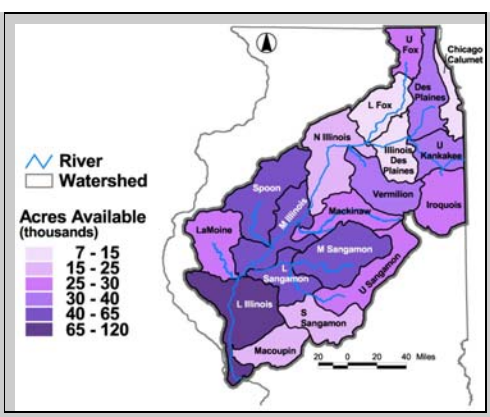


**Figure 36: Spring available total nitrogen load by sub-watershed**

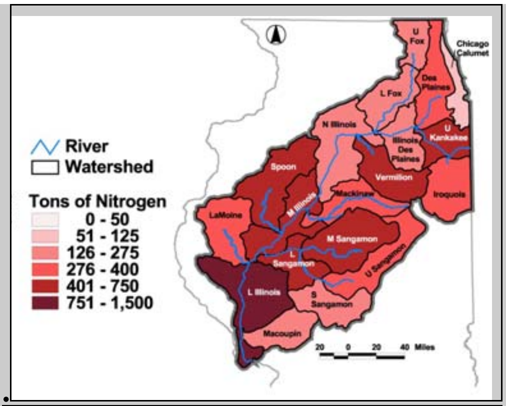



**Figure 37: Spring marginal cost (price) by watershed.**

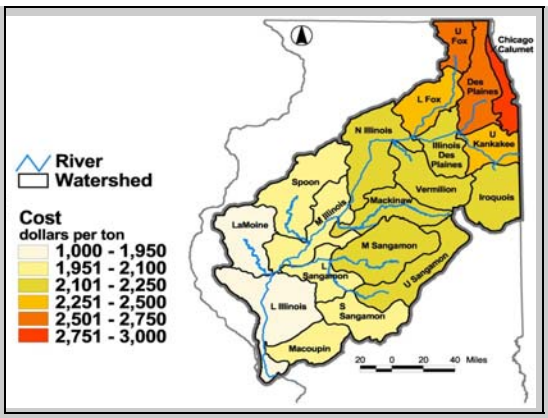
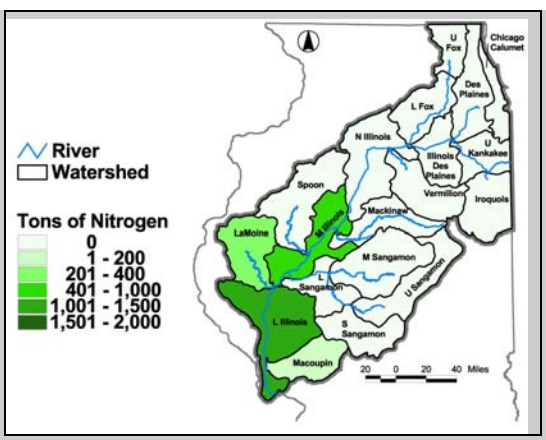



**Figure 38: Unrestricted spring credit sales (tons/month) by sub-watershed**




## 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 39); 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 X 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 27). 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 25: 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[13] | $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 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.

[13] 1Assumes all credits were sold at the cheapest cost within the Illinois River Watershed.

*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 nitrogen 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 to use 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 "choke" 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.

*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 US and western Europe, ~ 70 and ~60 g animal protein per person per day, respectively. In 2003, total protein consumption in the US 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 US and globally. Additional Nr may be conserved by decreasing the amount of food that is wasted.

*Removing croplands that are susceptible to Nr loss from crop production.* Booth and Campbell (2007)'s model 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.*
>
> *Would increasing conservation to this extent be sufficient to reduce nitrate loading and the size of the Gulf hypoxic zone? It has been estimated a 30% reduction in total nitrogen inputs would shrink the hypoxic zone by 20-60%, though still greater reductions may be required. Our model characterizes nitrate loading, not total nitrogen loading, but under our scenario, a 30% reduction in total nitrate inputs to the Gulf would require a 50% reduction in agricultural loading to aquatic systems."*

The latest Gulf of Mexico hypoxia report indicates that an even greater Nr reduction is needed to get appreciable decrease of the hypoxic zone (EPA SAB, 2007a).

*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).

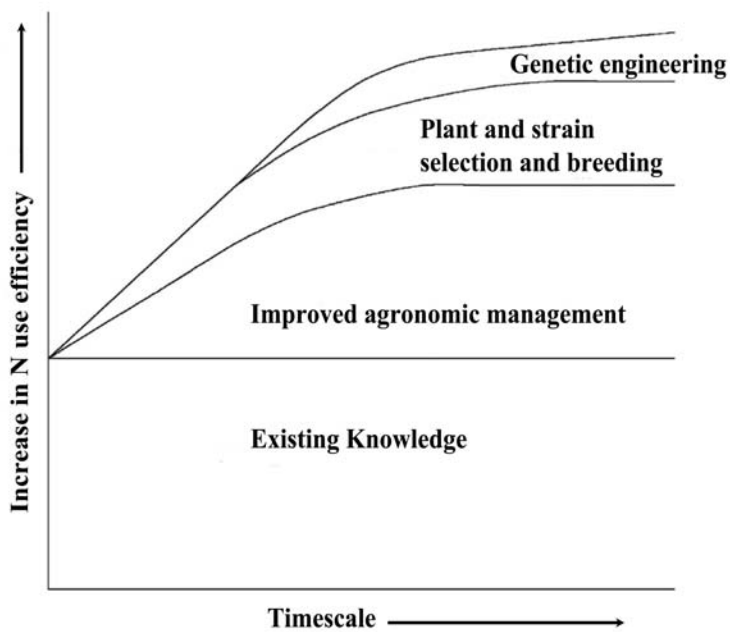
Corn yield in the US has increased from an average of 100 bu/ac in 1985 to 136 bu/ac in 2005 as a result of improved nutrient and pest management, expansion of irrigated area, conservation tillage, soil testing, and improved crop genetics (yield and pest resistance) [Council for Agricultural Science 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 US 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 US 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 loss 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 both total NUE and 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 NUE (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 NUE 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 18 indicates where expected greatest gains in NUE 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 to achieve the degree of improved synchrony needed to make substantial improvements in NUE. 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 NUE will be needed to ensure that the goals of food security, biofuel production, and protection of environmental quality are met.

**Figure 39: The likely impact of research investment in increasing N use efficiency (Giller et al. 2004)**




*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 losses 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 bemoaned 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) cattle and pig slurry tanks $NH_3$ 3.3 kg N per square meter per year until covered with straw then below detection limit. Mahimairaja et al. (1994) reported that $NH_3$ volatilization was reduced by 90-95% under anaerobic conditions. See section 3.2 for a discussion of best management practices to minimize $NH_3$ emissions from livestock waste.

*Wetlands to decrease $NO_3^-$ loading of aquatic systems.* The 40 to 60% of fertilizer N that is not used by crop production, and an appreciable portion of the N fixed by soybeans, is moved from the crop field into surface and groundwater. Del Grosso et al. (2006) estimate that nationally, 30+% as much as the N applied as fertilizer is susceptible to leaching. In soybean production, where little fertilizer N is used, nitrate leaching still poses a significant problem. Del Grosso et al. (2006) estimate that 93% as much nitrate is leached under soybean production as under corn production. Much of the nitrate leached from agricultural fields could be removed from drainage water in wetlands, either natural or reconstructed.

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 MOM (Mississippi-Ohio-Missouri) 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 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).

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. 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 (See section 2.4.7.3).

3.3.4.2. *Technical controls (control points) on transfer and transformations 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 3.3.1).

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 is 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

### Finding 19

Human activities have significantly increased the introduction of Nr into the US environment, and 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, 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.** *EPA should pursue an integrated approach to develop the understanding necessary for science-based policies, regulations, and incentives to avoid and remediate the impacts of excess Nr on the environmental, human health, and climate. Such integration must cut across media (air, land, and water), Nr form (oxidized and reduced), federal agencies, and existing legislative statutes [e.g., EISA – Energy Independence and Security Act (EISA), the Clean Air Act, and the Water Quality Act (CWA)].*

**Recommendation B.** *EPA should form an Intra-agency Task Force on Managing Nr that builds upon existing Nr efforts within the Agency. The task force would identify the most cost-effective approaches to avoid the negative impacts of Nr loads cascading through the environment. These loads pose a significant threat to human health and environmental quality and directly affects climate change*

**Recommendation C.** *The federal government should form an Inter-agency Task Force on Managing Nr be formed, with EPA as the lead agency that includes at a minimum U.S. Department of Agriculture (USDA), U.S. Department of Energy (DOE), U.S. Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), and U.S. Geological Survey (USGS). This Task Force would coordinate federal programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and regulating Nr in the environment.*

The task forces should take a systems approach to science and research by:

- evaluating critical loads
- Nr budgets and life cycle accounting
- monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads
- development and use of systemic models for Nr management; new technologies; fertilizer and nutrient best management practices (BMPs)
- development of Nr indicators necessary for the assessment of effects related to excess Nr on human health and the environment
- assessing combined carbon (C) and N effects
- addressing indicators/endpoints, costs, benefits and risks associated with the impairment of human health and decline and restoration of ecosystem services
- investigating the need for new regulations
- new education, outreach, and communication initiatives
- implementing economic incentives, particularly those that integrate air, aquatic, and land sources of Nr
- new infrastructures for managing Nr releases to the environment
- review of enabling legislation for purposes of extending regulatory authority or streamlining procedures for enacting Nr risk reduction strategies.

In addition to these overarching recommendations, the committee makes a number of other findings and recommendation related to managing the risk of Nr in the environment.

## Finding 20

Emissions of Nr from fuel combustion (in the form of $NO_x$) have been reduced substantially for some classes of emitters such as power plants and light duty vehicles. Other sources, including most off road vehicles, some industrial equipment, and some older electricity generating units, operate with little or no $NO_x$ controls. Most sources can be controlled (with well established engineering practices and at a reasonable cost) to the point of 90% reductions of $NO_x$ emissions relative to uncontrolled combustion. $NO_y$ levels in the atmosphere remain too high to protect public health and welfare, and continued reductions of $NO_x$ emissions are necessary. Most $NO_x$ sources can be controlled at the 90% level (relative to uncontrolled combustion) with existing technology and at a reasonable cost, and this should be an across the board goal. Accordingly, the Agency should continue to reduce $NO_x$ emissions from major point sources, such as EGU's, using a market mechanism such as cap and trade. Under this scenario, it is likely that high efficiency, low emission power plants will be built for energy needs.

**Recommendation 20a.** *Decrease NOx emissions from off road vehicles, some industrial equipment, and some older electricity generating units that currently operate with little or no $NO_x$ controls.*

**Recommendation 20b**. *Improve monitoring and modeling of $NO_x$ from vehicles and other mobile sources.*

**Recommendation 20c**. *The committee recommends that in implementing this approach, the Agency consider the mass of $NO_x$ emitted per unit of power provided, rather than past emissions, as the figure of merit for EGUs.*

**Recommendation 20d.** *For total $NO_x$ from all mobile sources the Agency should consider a cap that decreases with time.*

**Recommendation 20e.** *The Agency should implement and maintain programs such as inspection and maintenance or road-side monitoring to ensure that most light-duty vehicles meet emissions standards.*

**Recommendation 20f.** *The Agency should require major sources of $NO_x$ from industrial fuel combustion to implement control technologies and/or include in cap and trade programs.*

**Recommendation 20g.** *The Agency should promulgate stricter $NO_x$ emissions standards for heavy duty diesel vehicles and off road vehicles including locomotives, construction, farm, and landscaping equipment, and marine vehicles.*

**Recommendation 20h.** *The Agency should encourage, through its sustainability initiatives, replacement of EGUs powered by fossil fuels with cleaner energy sources such as wind and solar.*

**Recommendation 20i.** *The Agency should promote changes in lifestyle, urban planning, and public transit conducive to energy conservation and reduced emissions*.

**Finding 21**

The committee believes that excess flows of Nr into streams, rivers, and coastal systems be reduced by approximately 20% **(~1 Tg)** through improved 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, the committee believes that crop N-uptake efficiencies 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 artificial Nr, amounting to ~**2.4 Tg** below current levels of Nr additions to the environment. These are appropriate targets with today's available technologies and further progress is possible.

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% **(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 losses to soils, water and air (Simpson et al. 2008). Integrated management strategies will be required. In this regard, the committee endorses Section 204 of the 2007 Energy Independence and Security Act (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.

**Recommendation 21**. *The committee recommends that improved detail and regularity of data acquisition for fertilizer use by major crop (and for urban residential and recreational turf) and county (or watershed) be undertaken in order to better inform decision-making about policies and mitigation options for nitrogen in these systems.*

**Recommendation 21b.** *Then committee recommends that improved estimates of N fertilizer uptake efficiency for the major N-using crops and cropping systems based on direct measurements from a representative range of production-scale farmer's fields should be undertaken to help guide prioritization of risk mitigation strategies.*

**Recommendation 21c**. *The committee recommends that loss estimates of* $NH_3$, $NO_x$, $NO_3^-$ *leaching, and* $N_2O$*; estimates of total Nr storage (or loss in soils coupled with organic carbon); estimates of the fate of Nr goes (most losses are currently attributed to denitrification in soils* *and water) must be improved, and that loss mechanisms from soil be better identified (e.g. is denitrification the major loss mechanism or are loss estimates too large?)*

**Recommendation 21d.** *The Agency should work closely with USDA and other agencies to identify research and education priorities for prevention and mitigation of Nr applied to agricultural systems.*

**Recommendation 21e.** *The committee recommends that research on the accelerating the rate of gain in crop yields on existing farmland be increased while substantially increasing N fertilizer uptake efficiency.*

**Recommendation 21f.** *The committee recommends that the Agency develop and promote incentives for the use of advanced fertilizers and enhanced efficiency products in crop and livestock agriculture.*

**Recommendation 21g.** *The committee recommends that the Agency undertake an expanded research program for wetlands design and management focused, in part, on Nr dynamics and removal.*

**Finding 22**

$N_2O$ in the atmosphere is also increasing. For additional production of liquid biofuels beyond the grandfathered amount in EISA, EPA has the power to exercise some controls on $N_2O$ emissions through the life cycle greenhouse gas accounting requirements. In addition, greenhouse gas emissions trading will provide both opportunities and challenges with regard to mitigation of Nr environmental and health impacts.

**Recommendation 22**. *The committee therefore recommends that policies and regulations that support implementation of emissions trading consider Nr impacts on GHG emissions and reward reductions of N-related GHG. Biofuel subsidies should accurately account for Nr contributions to GHG emissions, and individual biofuel plants should be certified for GHG impact and serve as aggregators in the biofuel production life-cycle to reward reductions in $N_2O$ emissions through BMPs by farmers producing feedstock and use of co-products in livestock diets.*

**Finding 23**

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. 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. These programs have been, and continue to be, important ways of managing Nr in the urban environment. As shown in section 3.2, 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. In most cases Nr ultimately finds its way into municipal and private sewers and treatment systems where, irrespective of its initial chemical form, it is partially or completely nitrified. Subsequent engineered complete denitrification processes (including tertiary wastewater treatment, engineered or restored wetlands, and algae production for biofuels) can convert the nitrate to $N_2$. Federal and State assistance programs directed at construction of treatment plants are an important element Nr control policy in the US. The committee believes that **0.5 to 0.8 Tg N/yr** can be cut from Nr inputs to the environment.

**Recommendation 23.** *The committee recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA. The committee recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA.*

**Finding 24**

In spite of gains made over the last several decades in lowering the amount of $NO_x$ 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. 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. These trends are in requiring fewer animals to produce more 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 will further decrease ammonia and other gases and odor emissions. For example, sawdust litter helps reduce $NH_3$ 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%) at hog facilities.

**Recommendation 24**. *The committee recommends a goal of decreasing livestock-derived $NH_3$ emissions to approximately 80% of 1990 emissions, a decrease of* ***0.5 Tg N/yr*** *(by a combination of BMPs and engineered solutions). This will reduce $PM_{2.5}$ 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.*

**Finding 25**

The CAA (1970) and its Amendment (1990), have resulted in $NO_x$ emissions that are <50% of what they would have been without the controls. While this is an admirable accomplishment, there is still a need to seek improvements, as $NO_x$ emissions are still an order of magnitude greater than at the beginning of the 20$^{th}$ century, and as a consequence there are still negative impacts on both humans and ecosystems. In 2002, coal-fired utilities generated approximately 1.2 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 or 50% reduction; in fact the $NO_x$ SIP call implemented in 2003 and 2004 reduced 2002 emissions by 0.3 Tg N/yr, so in essence, half the reduction has already been accomplished. Looking at mobile sources, highway source emissions are around 2.2 Tg and off-highway around 1.2 Tg for a total of 3.4, again 2002 numbers. Assuming a 40% reduction for these sources (most off-highway mobile sources currently have no controls, but could achieve 80-90% $NO_x$ removal), there is a potential reduction of 1.4 Tg. The total reduction is then approximately 2 Tg.

The reduction of $NO_x$ emissions by 2 Tg N /yr may be inadequate for many areas to achieve 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 and wider use of hybrid or electric cars should be considered.

**Recommendation 25.** *The committee recommends that the EPA expand its $NO_x$ control efforts from the current reductions of emissions of passenger cars and power plants to include other important unregulated mobile and stationary sources sufficient to achieve a 2.0 Tg reduction in the generation of reactive nitrogen.*

*The committee's recommendations, if implemented, would reduce total Nr loadings to the environment in the US by approximately 25% below current levels. Figure 40 compares current and proposed Nr flows in the US. The committee believes that these represent realistic intermediate targets based on current technology, however further reductions are needed for many N-sensitive ecosystems (e.g., estuarine and coastal waters). Developing these* *opportunities, and going beyond these recommended Nr reduction targets, are critical given the growing demand from population and economic growth for food- and fiber-production and energy use.*

**<u>Summary statement</u>**

The committee's recommendations, if implemented, would reduce total Nr loadings to the environment in the US by approximately 25% below current levels. The committee believes that these represent realistic intermediate targets based on current technology, however further reductions are needed for many N-sensitive ecosystems (e.g., estuarine and coastal waters). Developing these opportunities, and going beyond these recommended Nr reduction targets, are critical given the growing demand from population and economic growth for food- and fiber-production and energy use.

**Figure 40: Illustration of the impact that proposed management actions will have on the introduction of Nr into the US or the loss of Nr from agricultural systems**

Red ovals and arrows indicate secondary declines in Nr throughout the nitrogen cascade.

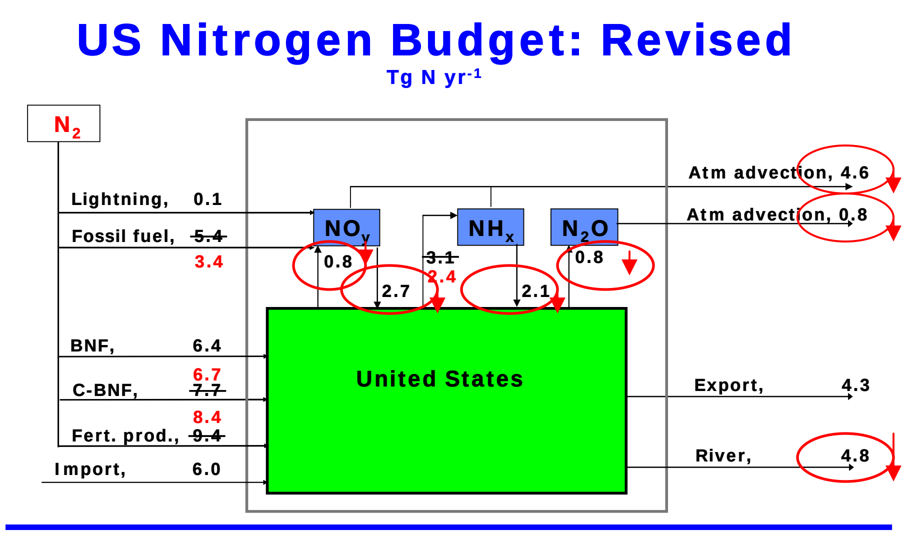



Figure 40 illustrates the impact that the proposed management actions will have on the introduction of Nr into the US, or the loss of Nr from agricultural systems. Other fluxes in Figure 40 will also be reduced due to the decrease in Nr introduction or loss, but the magnitude of such changes will require more detailed analysis.

# 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

NRC – National Research Council

$O_2$ – 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

SOM035 - ??

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

NRCS – Natural Resources Conservation Service

NRD – Natural Resource District

NRI –National Resources Inventory

NTN – National Trends Network

OTAG – Ozone Transport Assessment Group

OTC – Ozone Transport Commission

PFP – Partial Factor Productivity

POTW – Publicly Owned Treatment Works

PSD – Prevention of Significant Deterioration

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

## I. 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.

## II. Overarching Recommendations

### Recommendation A

*EPA should pursue an integrated approach to develop the understanding necessary for science-based policies, regulations, and incentives to avoid and remediate the impacts of excess Nr on the environmental, human health, and climate. Such integration must cut across media (air, land, and water), Nr form (oxidized and reduced), federal agencies, and existing legislative statutes [e.g., EISA – Energy Independence and Security Act (EISA), the Clean Air Act, and the Clean Water Act (CWA)].*

### Recommendation B

*EPA should form an Intra-agency Task Force on Managing Nr that builds upon existing Nr efforts within the Agency. The task force would identify the most cost-effective approaches to avoid the negative impacts of Nr loads cascading through the environment. These loads pose a significant threat to human health and environmental quality and directly affect climate change.*

### Recommendation C

*The federal government should form an Inter-agency Task Force on Managing Nr be formed, with EPA as the lead agency that includes at a minimum U.S. Department of Agriculture (USDA), U.S. Department of Energy (DOE), U.S. Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), and U.S. Geological Survey (USGS). This Task Force would coordinate federal programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and regulating Nr in the environment.*

The task forces should take a systems approach to science and research by:

- evaluating critical loads
- Nr budgets and life cycle accounting
- monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads

- development and use of systemic models for Nr management; new technologies; fertilizer and nutrient best management practices (BMPs)
- development of Nr indicators necessary for the assessment of effects related to excess Nr on human health and the environment
- assessing combined carbon (C) and N effects
- addressing indicators/endpoints, costs, benefits and risks associated with the impairment of human health and decline and restoration of ecosystem services
- investigating the need for new regulations
- new education, outreach, and communication initiatives
- implementing economic incentives, particularly those that integrate air, aquatic, and land sources of Nr
- new infrastructures for managing Nr releases to the environment
- review of enabling legislation for purposes of extending regulatory authority or streamlining procedures for enacting Nr risk reduction strategies.

## III. Findings and Recommendations Related to the Sources of Nr to the US

### 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) of total US Nr losses 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 of N applied to major crops and the geospatial pattern by major watersheds.

**Recommendation 1**: *Improve detail and regularity of data acquisition for fertilizer use by major crop (and for urban residential and recreational turf) and county (or watershed) to better inform decision-making about policies and mitigation options for nitrogen in these systems, and to allow monitoring of impact from implemented policies and mitigation efforts.*

### Finding 2

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, and fiber demand of a growing human population while also protecting environmental quality and ecosystem services for future generations (Cassman, 1999). More diverse cropping systems with reduced N fertilizer input may also provide an option if there is a reduction in Nr losses per unit of crop production, which is required to avoid indirect land use change from expansion of crop production area to replace the loss in production.

**Recommendation 2:**

*d) 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 to better focus research investments, policy development, and prioritization of risk mitigation strategies.*

*e) Research is needed with an explicit focus on the challenge of both accelerating the rate of gain in crop yields on existing farm land while substantially increasing N fertilizer uptake efficiency and also on quantifying whether widespread adoption of lower-yielding more diverse cropping systems[14] with lower N fertilizer input requirements can reduce regional Nr load when the impact of indirect land use change is considered.*

*f) EPA should work closely with the U.S. Department of Agriculture (USDA), Department of Energy (DOE), and the National Science Foundation (NSF) to help identify research and education priorities for prevention and mitigation of Nr applied to agricultural systems.*

**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 NUE 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 must 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, and NSF.*

**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.

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[14] Greater diversity in a cropping system is achieved by increasing the number of different crop species used in the rotation (temporal diversity) or as a polycrop or intercrop (spatial diversity).

**Recommendation 4:** *There exists a critical need to understand and predict these changes in terms of maximizing the N efficiency of both crop and livestock production systems and to develop strategies for avoiding increased Nr load in the environment as a result of current and future expansion of biofuel production from corn and other "second generation*" biofuel feedstock crops.

**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 nationwide by a network of monitoring stations.*

**Finding 6**

Farm-level improvements in manure management can substantially reduce Nr load and losses. There are currently no incentives or regulations to decrease these losses 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 losses, taking into account phosphorus load issues.*

## IV. Findings and Recommendations Related to Nr Transfers and Transformations in and between environmental systems

**Finding 7**

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. 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. Ammonia, $NH_4^+$, and possibly organic N levels in the atmosphere are too high to protect public health and welfare, and reductions of $NH_x$ emissions are necessary.

**Recommendation 7a.** *Increase the scope and spatial coverage of the Nr concentration and flux monitoring network (such as CASTNET) for the US and appoint an oversight panel.*

**Recommendation 7b.** *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 techniques.*

**Recommendation 7c.** *The current $NO_2$ standard is inadequate to protect health and welfare, and compliance monitoring for $NO_2$ is inadequate for scientific understanding.*

**Recommendation 7d.** *Measure deposition directly both at the CASTNET sites and nearby locations with non-uniform surfaces such as forest edges.*

**Recommendation 7e.** *EPA should continue and support research into convective venting of the Planetary Boundary Layer and long range transport.*

**Recommendation 7f.** *Develop techniques and support observations of atmospheric organic N compounds in vapor, particulate, and aqueous phases.*

**Recommendation 7g.** *Improve quality and quantity of measurements of the $NH_3$ flux to the atmosphere from major sources especially agricultural practices.*

**Recommendation 7h.** *Improve numerical models of NOy and NHx especially their chemistry, surface deposition, and export. Develop linked ocean-land-atmosphere models of Nr.*

**Finding 8**

Although total N budgets within all terrestrial systems are highly uncertain, Nr losses 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 losses ascribed to leaching, runoff and denitrification, are as uncertain as the N budgets themselves.

**Recommendation 8:** *EPA, USDA, DOE, and universities should work together to ensure that the N budgets of terrestrial systems are properly quantified and that the magnitude of at least the major loss vectors are known, by funding appropriate research.*

**Finding 9**

Over the past 25 years, there has been a growing recognition of anthropogenic 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 9:** *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 26**

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

## V. Findings and Recommendations Related to the Impacts and Metrics for Nr

**Finding 11**

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

**Recommendation 11.** *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 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 with parallel interests in advancing the necessary underlying science of Nr creation, transport and transformation, impacts, and management.*

**Finding 12**

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 N management strategies.

**Recommendation 12.** *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 13**

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 STPs 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 13.** *To better address Nr runoff and discharges from the peopled landscape the committee recommends that EPA*:

**13a.** *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.*

**13b**. *Review current regulatory practices for point sources, including both wastewater treatment plants and stormwater, to determine adequacy and relationship 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.*

**13c.** *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.*

**13d.** *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.*

**13e.** *Use ecosystem-based management approaches that balance* natural and anthropogenic needs and presence in the landscape.

**Finding 14**

Meeting Nr management goals for estuaries, when a balance must be struck between economies, society and the environment, under current federal law seems unlikely. Enforceable authorities over nonpoint source, stormwater, air (in terms of critical loads), and land use – both development and agriculture are inadequate to require necessary Nr controls. Funding programs are presently inadequate to meet existing pollution control needs. Further, new technologies and management approaches are required to meet ambitious Nr control needs aimed at restoring national water quality.

**Recommendation 14.** *Therefore, the committee 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 criteria and some compromises in ecosystem goals to accommodate human uses of the air, land and water.*

**Finding 15**

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 used to formulate new management procedures. The process of monitoring and control revisions is termed adaptive management—a process that the committee supports as it does not delay actions that can be taken now, but acknowledges the likelihood that management programs will be altered (adapted) as scientific and management understanding improve**.**

**Recommendation 15.** *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 USGS programs and the NADP effort.*

**Finding 16**

The committee finds that the net benefit of $NH_3$ emissions reductions greatly outweigh any potential harm.

**Recommendation 16.** *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 17**

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 17.** *The committee recommends that the integrated strategies for N management specified in this report be developed in cognizance of these interrelations and tradeoffs*

## VI. Findings and Recommendations Related to Integrated Risk Reduction Strategies for Nr

**Finding 18**

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 18.** *The committee recommends that the Agency work toward adopting the critical loads approach in the future. 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*.

**Finding 27**

Human activities have significantly increased the introduction of Nr into the US environment, and 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, 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.** *EPA should pursue an integrated approach to develop the understanding necessary for science-based policies, regulations, and incentives to avoid and remediate the impacts of excess Nr on the environmental, human health, and climate. Such integration must cut across media (air, land, and water), Nr form (oxidized and reduced), federal agencies, and existing legislative statutes [e.g., EISA – Energy Independence and Security Act (EISA), the Clean Air Act, and the Water Quality Act (CWA)].*

**Recommendation B.** *EPA should form an Intra-agency Task Force on Managing Nr that builds upon existing Nr efforts within the Agency. The task force would identify the most cost-effective approaches to avoid the negative impacts of Nr loads cascading through the environment. These loads pose a significant threat to human health and environmental quality and directly affects climate change*

**Recommendation C.** *The federal government should form an Inter-agency Task Force on Managing Nr be formed, with EPA as the lead agency that includes at a minimum U.S. Department of Agriculture (USDA), U.S. Department of Energy (DOE), U.S. Department of Transportation (DOT), National Oceanic and Atmospheric Administration (NOAA), and U.S. Geological Survey (USGS). This Task Force would coordinate federal* *programs that address Nr concerns and help ensure clear responsibilities for monitoring, modeling, researching, and regulating Nr in the environment.*

The task forces should take a systems approach to science and research by:

- evaluating critical loads
- Nr budgets and life cycle accounting
- monitoring as the basis for informed policies, regulations, and incentive frameworks for addressing excess Nr loads
- development and use of systemic models for Nr management; new technologies; fertilizer and nutrient best management practices (BMPs)
- development of Nr indicators necessary for the assessment of effects related to excess Nr on human health and the environment
- assessing combined carbon (C) and N effects
- addressing indicators/endpoints, costs, benefits and risks associated with the impairment of human health and decline and restoration of ecosystem services
- investigating the need for new regulations
- new education, outreach, and communication initiatives
- implementing economic incentives, particularly those that integrate air, aquatic, and land sources of Nr
- new infrastructures for managing Nr releases to the environment
- review of enabling legislation for purposes of extending regulatory authority or streamlining procedures for enacting Nr risk reduction strategies.

**Finding 28**

Emissions of Nr from fuel combustion (in the form of $NO_x$) have been reduced substantially for some classes of emitters such as power plants and light duty vehicles. Other sources, including most off road vehicles, some industrial equipment, and some older electricity generating units, operate with little or no $NO_x$ controls. Most sources can be controlled (with well established engineering practices and at a reasonable cost) to the point of 90% reductions of $NO_x$ emissions relative to uncontrolled combustion. $NO_y$ levels in the atmosphere remain too high to protect public health and welfare, and continued reductions of $NO_x$ emissions are necessary. Most $NO_x$ sources can be controlled at the 90% level (relative to uncontrolled combustion) with existing technology and at a reasonable cost, and this should be an across the board goal. Accordingly, the Agency should continue to reduce $NO_x$ emissions from major point sources, such as EGU's, using a market mechanism such as cap and trade. Under this scenario, it is likely that high efficiency, low emission power plants will be built for energy needs.

**Recommendation 20a.** *Decrease NOx emissions from off road vehicles, some industrial equipment, and some older electricity generating units that currently operate with little or no $NO_x$ controls.*

**Recommendation 20b**. *Improve monitoring and modeling of $NO_x$ from vehicles and other mobile sources.*

**Recommendation 20c**. *The committee recommends that in implementing this approach, the Agency consider the mass of $NO_x$ emitted per unit of power provided, rather than past emissions, as the figure of merit for EGUs.*

**Recommendation 20d.** *For total $NO_x$ from all mobile sources the Agency should consider a cap that decreases with time.*

**Recommendation 20e.** *The Agency should implement and maintain programs such as inspection and maintenance or road-side monitoring to ensure that most light-duty vehicles meet emissions standards.*

**Recommendation 20f.** *The Agency should require major sources of $NO_x$ from industrial fuel combustion to implement control technologies and/or include in cap and trade programs.*

**Recommendation 20g.** *The Agency should promulgate stricter $NO_x$ emissions standards for heavy duty diesel vehicles and off road vehicles including locomotives, construction, farm, and landscaping equipment, and marine vehicles*.

**Recommendation 20h.** *The Agency should encourage, through its sustainability initiatives, replacement of EGUs powered by fossil fuels with cleaner energy sources such as wind and solar.*

**Recommendation 20i.** *The Agency should promote changes in lifestyle, urban planning, and public transit conducive to energy conservation and reduced emissions*.

**<u>Finding 29</u>**

The committee believes that excess flows of Nr into streams, rivers, and coastal systems be reduced by approximately 20% **(~1 Tg)** through improved 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, the committee believes that crop N-uptake efficiencies 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 artificial Nr, amounting to ~**2.4 Tg** below current levels of Nr additions to the environment. These are appropriate targets with today's available technologies and further progress is possible.

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% **(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 losses to soils, water and air (Simpson et al. 2008). Integrated management strategies will be required. In this regard, the committee endorses Section 204 of the 2007 Energy Independence and Security Act (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.

**Recommendation 21**. *The committee recommends that improved detail and regularity of data acquisition for fertilizer use by major crop (and for urban residential and recreational turf) and county (or watershed) be undertaken in order to better inform decision-making about policies and mitigation options for nitrogen in these systems.*

**Recommendation 21b.** *Then committee recommends that improved estimates of N fertilizer uptake efficiency for the major N-using crops and cropping systems based on direct measurements from a representative range of production-scale farmer's fields should be undertaken to help guide prioritization of risk mitigation strategies.*

**Recommendation 21c**. *The committee recommends that loss estimates of $NH_3$, $NO_x$, $NO_3^-$ leaching, and $N_2O$; estimates of total Nr storage (or loss in soils coupled with organic carbon); estimates of the fate of Nr goes (most losses are currently attributed to denitrification in soils and water) must be improved, and that loss mechanisms from soil be better identified (e.g. is denitrification the major loss mechanism or are loss estimates too large?)*

**Recommendation 21d.** *The Agency should work closely with USDA and other agencies to identify research and education priorities for prevention and mitigation of Nr applied to agricultural systems.*

**Recommendation 21e.** *The committee recommends that research on the accelerating the rate of gain in crop yields on existing farmland be increased while substantially increasing N fertilizer uptake efficiency.*

**Recommendation 21f.** *The committee recommends that the Agency develop and promote incentives for the use of advanced fertilizers and enhanced efficiency products in crop and livestock agriculture.*

**Recommendation 21g.** *The committee recommends that the Agency undertake an expanded research program for wetlands design and management focused, in part, on Nr dynamics and removal.*

## Finding 30

$N_2O$ in the atmosphere is also increasing. For additional production of liquid biofuels beyond the grandfathered amount in EISA, EPA has the power to exercise some controls on $N_2O$ emissions through the life cycle greenhouse gas accounting requirements. In addition, greenhouse gas emissions trading will provide both opportunities and challenges with regard to mitigation of Nr environmental and health impacts.

**Recommendation 22**. *The committee therefore recommends that policies and regulations that support implementation of emissions trading consider Nr impacts on GHG emissions and reward reductions of N-related GHG. Biofuel subsidies should accurately account for Nr contributions to GHG emissions, and individual biofuel plants should be certified for GHG impact and serve as aggregators in the biofuel production life-cycle to reward reductions in $N_2O$ emissions through BMPs by farmers producing feedstock and use of co-products in livestock diets.*

## Finding 31

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. 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. These programs have been, and continue to be, important ways of managing Nr in the urban environment. As shown in section 3.2, 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. In most cases Nr ultimately finds its way into municipal and private sewers and treatment systems where, irrespective of its initial chemical form, it is partially or completely nitrified. Subsequent engineered complete denitrification processes (including tertiary wastewater treatment, engineered or restored wetlands, and algae production for biofuels) can convert the nitrate to $N_2$. Federal and State assistance programs directed at construction of treatment plants are an important element Nr control policy in the US. The committee believes that **0.5 to 0.8 Tg N/yr** can be cut from Nr inputs to the environment.

**Recommendation 23.** *The committee recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA.* *The committee recommends that a high priority be assigned to nutrient management through a targeted construction grants program under the CWA.*

**Finding 32**

In spite of gains made over the last several decades in lowering the amount of $NO_x$ 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. 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. These trends are in requiring fewer animals to produce more 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 will further decrease ammonia and other gases and odor emissions. For example, sawdust litter helps reduce $NH_3$ 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%) at hog facilities.

**Recommendation 24**. *The committee recommends a goal of decreasing livestock-derived $NH_3$ emissions to approximately 80% of 1990 emissions, a decrease of* ***0.5 Tg***

***N/yr** (by a combination of BMPs and engineered solutions). This will reduce $PM_{2.5}$ 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.*

## Finding 33

The CAA (1970) and its Amendment (1990), have resulted in $NO_x$ emissions that are <50% of what they would have been without the controls. While this is an admirable accomplishment, there is still a need to seek improvements, as $NO_x$ emissions are still an order of magnitude greater than at the beginning of the 20$^{th}$ century, and as a consequence there are still negative impacts on both humans and ecosystems. In 2002, coal-fired utilities generated approximately 1.2 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 or 50% reduction; in fact the $NO_x$ SIP call implemented in 2003 and 2004 reduced 2002 emissions by 0.3 Tg N/yr, so in essence, half the reduction has already been accomplished. Looking at mobile sources, highway source emissions are around 2.2 Tg and off-highway around 1.2 Tg for a total of 3.4, again 2002 numbers. Assuming a 40% reduction for these sources (most off-highway mobile sources currently have no controls, but could achieve 80-90% $NO_x$ removal), there is a potential reduction of 1.4 Tg. The total reduction is then approximately 2 Tg.

The reduction of $NO_x$ emissions by 2 Tg N /yr may be inadequate for many areas to achieve 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 and wider use of hybrid or electric cars should be considered.

**Recommendation 25.** *The committee recommends that the EPA expand its $NO_x$ control efforts from the current reductions of emissions of passenger cars and power plants to include other important unregulated mobile and stationary sources sufficient to achieve a 2.0 Tg reduction in the generation of reactive nitrogen.*

*The committee's recommendations, if implemented, would reduce total Nr loadings to the environment in the US by approximately 25% below current levels. Figure 40 compares current and proposed Nr flows in the US. The committee believes that these represent realistic intermediate targets based on current technology, however further reductions are needed for many N-sensitive ecosystems (e.g., estuarine and coastal waters). Developing these opportunities, and going beyond these recommended Nr reduction targets, are critical given the growing demand from population and economic growth for food- and fiber-production and energy use.*

# Technical Appendices

## 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:

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

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

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

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

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

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

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 g 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 (Figure 19) 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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